PRELIMINARY EVALUATION OF SELECTED PESTICIDES IN THE NICOMEKL RIVER WATERSHED, LOWER FRASER VALLEY, B.C., AND PROPOSED APPROACHES FOR ASSESSING TOXICITY TO AQUATIC BIOTA

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Environmental Services Ltd.

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SUMMARY

A multi-year project to assess the potential toxic effects of agricultural crop pesticides on aquatic biota within the Lower Fraser Valley, B.C., was commenced in late 1988. This report describes the Phase I tasks undertaken and completed to initiate the project.

The overall objective of Phase I was to develop a site-specific approach for predicting and monitoring the potential toxic effects on aquatic biota of pesticides entering a receiving-water body by surface water runoff from commercial farms. Initially, those herbicides and insecticides registered in Canada and marketed to farmers in the Lower Fraser Valley were examined with respect to their perceived regional use and hazard to the freshwater environment. Based on this appraisal, dinoseb (herbicide) and endosulfan (insecticide) were selected for investigation of their presence and potential toxicity to aquatic life frequenting the study area. The international scientific literature dealing with the freshwater fate and effect of these two pesticides was subsequently searched, retrieved, and reviewed with emphasis on their toxic effects (lethal and sublethal, acute and chronic).

Prospective study sites were visited and a decision made to confine the project to a portion of the Nicomekl River (Lower Mainland, B.C.) and adjacent drainage/irrigation ditches. These ditches discharge seasonal runoff from adjoining commercial farms directly to the Nicomekl River. Available information regarding sources of wastewater discharges to the Nicomekl River and its tributaries was reviewed together with data on riverwater quality and water uses. An understanding of the ditch configurations and flow-control devices was obtained from regional engineers.

On one occasion (March 1989; prior to seasonal application of pesticides), samples of water and sediment were collected from various sites along the Nicomekl River and from certain ditches near their confluence with the river. These samples were analysed for residual pesticide (dinoseb and endosulfan I and II) concentrations using solvent extractions followed by Florisil column chromatography and electron-capture gas chromatography. Appropriate quality control was included as part of the analyses. For all water samples (ditch and river), concentrations of dinoseb and endosulfan were below detection limits. Concentrations of dinoseb in sediment samples ranged from undetectable or trace amounts (most river sediments) to 49 ug/kg (dry weight basis). Except for a single ditch sediment containing 428 ug/kg endosulfan, concentrations of this insecticide in ditch and river sediments were undetectable or at trace levels only. The relevance of these findings was appraised and discussed with respect to the study area, published findings for other sites, available water quality criteria for these pesticides, and the known water quality of the Nicomekl River.

Based on the foregoing, an approach for future site-specific toxicity assessments of the study area was presented. Recommended biological testing of selected ditch and riverwater samples included several series of acute lethal/sublethal toxicity tests using rainbow trout to determine effect/no effect concentrations. Chronic toxicity tests of certain water samples using Ceriodaphnia sp. (freshwater microcrustacean) and fathead minnows were recommended, as were algal toxicity tests. Field toxicity tests and surveys for biological effects were proposed for future study. Chemical analyses (for pesticide residues and other contaminants) of toxic test waters were integrated in the design. The proposed approach included suggested sampling stations and frequencies of monitoring.

ACKNOWLEDGEMENTS

Sponsorship of this project was provided by Environment Canada, with fiscal coordination by Supply and Services Canada. Messrs. Stephen W. Sheehan and Fred Mah (Water Quality Branch, Inland Waters, Pacific & Yukon Region) of Environment Canada were responsible for project conception, acted as Scientific Authorities, and provided valuable input and guidance throughout the study.

Mr. Eric McGreer (Coastline Environmental Services Ltd.) was program manager and responsible for site selection, sampling and overall coordination of the project. B.C. Research Corporation (Drs. Carl Alleyne and Jim McKinley) performed the pesticide analyses under sub-contract to Coastline. Ms. Angela Wong and supporting librarian staff within B.C. Research Corp. also provided the computer-assisted search and retrieval of technical documents. Dr. Don McLeay (McLeay Associates Ltd.) authored the report.

The individuals and agencies listed in Appendix I all provided their kind cooperation, technical advice and assistance during this undertaking. This help was sincerely appreciated and they are acknowledged with thanks.

TABLE OF CONTENTS

LIST OF TABLES, FIGURES AND APPENDICES

1 INTRODUCTION

1.1 Background

Much of the arable flatland of the Lower Fraser Valley is used for the commercial production of vegetables. Predominant crops include potatoes, peas, beans, corn, lettuce, cabbage, broccoli, cauliflower, corn, cucumbers, beets, carrots and tomatoes. To control weed and insect pests, a variety of herbicides and insecticides are applied seasonally by the regional vegetable farmers (Moody 1989).

Although applied directly to crops and cropland, agricultural-use pesticides can enter the aquatic environment through a number of routes including aerial spray drift, surface water runoff, soil leaching and associated groundwater infusion. Studies on the movement of pesticides from agricultural land to ditches, streams, rivers, lakes and coastal waters have received little attention in British Columbia until recently (Wan 1989). Regional investigation of the fate and effect of agricultural-use pesticides entering the aquatic environment has also been lacking.

Environment Canada scientists recently identified a need to develop and implement an approach for determining the potential toxic effects of agricultural pesticides on aquatic biota inhabiting or frequenting the Fraser River Estuary Management Program study area (FREMP 1986). During the third quarter of fiscal 1988/89, a contract was awarded to COASTLINE Environmental Services Ltd. to undertake the initial (1988/89) phase of an approach to study regional effects of farm-use pesticides on aquatic biota. The sequence of steps taken in developing an

approach appropriate for the study objectives is delineated in the present report. Also included is the presentation and appraisal of pesticide analytical data derived for certain samples of surface water and sediment collected from the region selected for phase two (fiscal 1989/90) investigations.

1.2 Project Scope

The scope of activities undertaken during this project included the following:

- frequent meetings and discussions with the Scientific Authorities (S. Sheehan, F. Mah).
- contacts (personal or by phone or writing) with persons and organizations knowledgeable with respect to the nature of the undertaking (see Appendix I).
- search, retrieval, review and appraisal of published information regarding the aquatic fate and effect of proposed pesticides to be monitored.
- chemical analyses of concentrations of selected pesticides in samples of surface water and sediment collected within the region chosen for investigation.
- assessment of approaches currently in use or being recommended for predicting and monitoring the biological impact of contaminants within the aquatic environment.

Based upon the study site selected for fiscal 1989/90 investigations, it was decided to restrict the review and appraisal of aquatic toxicity data and approaches to those appropriate for freshwater biota. Budget constraints for fiscal year 1988/89 and anticipated for the subsequent year(s) further restricted the study scope and intensity of effort available for this undertaking, as well as the breadth of chemical and biological tests considered to be appropriate for fiscal 1989/90 activities.

1.3 Objectives

- To review the agricultural pesticides commonly used by farmers in the Lower Fraser Valley with respect to their potential toxic threat to aquatic life.

- To select one or more pesticides for analysis of concentrations in samples of surface water and sediment from within the FREMP region.

- To review and appraise the technical literature reporting the aquatic fate and toxicity of the pesticide(s) selected for investigation.

- To choose a field site appropriate for the study.

- To undertake a field sampling program at the study site for analysis of pesticide residues in surface water and sediment samples.

- To analyse the collected samples for concentrations of the selected pesticide(s) and to assess the relevance of the values obtained.

- To develop and recommend an approach (including test methods) for predicting and monitoring the potential toxic effects of selected agricultural-use pesticides toward aquatic life inhabiting or frequenting the study area.

2 OVERVIEW OF STUDY TASKS

2.1 General

The overall objective of this project was to develop a site-specific approach for predicting and monitoring the potential toxic effects of farm pesticides on exposed aquatic biota. Pesticides under investigation were among those used by commercial farmers in the Lower Fraser Valley of British Columbia. The approach recommended was developed for possible implementation at the selected study site during fiscal 1989/90.

To achieve this objective, a number of study tasks were performed as outlined here. Each task represents a phase of activity required to enable the design of a worthwhile site-specific approach for evaluating toxic impact of in-use pesticides on the receiving environment.

2.2 Selection of Pesticides for Evaluation

This task involved a review of the herbicides and insecticides registered and marketed for crop use in Canada. Attention was directed towards the agricultural pesticides presently under re-evaluation by Agriculture Canada as well as those of concern to Environment Canada and Fisheries & Oceans Canada. Efforts were made to ascertain the agricultural pesticides known or believed to be used commonly by vegetable farmers within the Lower Fraser Valley. Particular consideration was given to the known aquatic persistence and toxicity of those pesticides in high regional use.

The pesticide selection process was facilitated by meetings and discussions with regional Agricultural Canada (M. Edwards), Environment Canada (D. Wilson, M. Wan), Fisheries & Oceans Canada (S. Samis), B.C. Ministry of Environment (B. Vance) and B.C. Ministry of Agriculture & Fisheries (M. Waring) representatives (see Appendix I). A number of vegetable farmers within the Lower Fraser Valley were also contacted and interviewed for pesticide-use information. Details regarding the pesticides selected for investigation and their properties and uses are provided in section 3.

2.3 Literature Review

The international scientific literature dealing with the aquatic fate and effect of the pesticides chosen for residue analysis and appraisal was searched, retrieved and reviewed. Thirteen separate computerized data bases, spanning the period 1968 to 1989, were searched (see Appendix II). The focus of attention for this review was the toxicity of the selected pesticides to freshwater biota, although consideration was also given to the environmental persistence of these pesticides and their ability to bioaccumulate in aquatic life.

The key words searched are indicated in Appendix II. Threshold chemical concentrations reported to be harmful to freshwater biota exposed experimentally to these pesticides are summarized in section 4 together with information concerning their aquatic persistence and ability to bioaccumulate. Approximately 100 articles pertinent to the aquatic fate and effect of the pesticides selected for investigation were retrieved in hard copy or on microfische.

2.4 Selection of Study Site

Prospective study sites were restricted to surface waters draining Lower Fraser Valley farmland within the Fraser River Estuary Management Program (FREMP) area of interest. This included drainage ditches and streams leading to the Lower Fraser, Nicomekl, Serpentine and Campbell Rivers.

The process of site selection was based upon initial assessments of topographical and land-use maps for these regions. Additionally, the permeability to groundwater flow for various sectors of arable farmland within the FREMP area was reviewed with assistance from H. Liebscher (Regional Hydrogeologist, Environment Canada). Potential study sites were confined to those regions where soil water was relatively impervious to groundwater infusion and drainage was thought to be predominantly by surface water runoff.

Visits were made to a number of promising sites during January and February 1989. Consideration was given to the extent of vegetable farms at each site, the size, configuration and flow of drainage ditches and receiving waters, and the availability of suitable reference station(s) within the receiving waterbody where pesticide levels would be expected to be low to non-detectable. The accessability of appropriate sampling stations and their suitability for in-situ aquatic toxicity tests were also considered. Sites where residues of the selected agricultural pesticides had been identified previously in ditchwater and/or ditch sediment by Environment Canada investigators (Wan 1989) were included in this field survey. A description of the site chosen for analysis of pesticide residues and subsequent (fiscal 1989/90) aquatic toxicity evaluations is provided in section 5.

2.5 Pesticide Residue Analyses

On one occasion only (March 16, 1989), samples of ditchwater, riverwater and sediment were taken from a number of stations within the chosen study area for analysis of concentrations of the pesticides under investigation. These samples were taken to determine pesticide levels persistent in surface waters and sediment within the study area, prior to seasonal applications.

The samples of sediment and water were analysed for pesticide residues by chemists at B.C. Research Corporation (Vancouver, B.C.). Method development and confirmation (i.e. percentage recovery of pesticide in spiked samples) were included as part of the analytical program. Procedures used for sample collection and analysis are detailed in section 6 and Appendix III, together with the analytical results.

2.6 Appraisal of Pesticide Residue Data

Pesticide levels found in the sediment and water samples taken from the study area during March 1989 were assessed with respect to the nature of the region under investigation (including regional land use adjacent to the drainage ditches sampled) and the presumed seasonal use of these chemicals. The analytical values obtained were also considered with respect to those reported to be harmful to freshwater life (section 4). Additionally, these value were compared with pesticide residues found previously by Environment Canada researchers in water and sediment samples taken from agricultural-use ditches in the Lower Fraser Valley (including one of the ditches sampled in the present study). Observations regarding this appraisal are presented in section 7.

2.7 Recommended Approach for Toxicity Assessments

A site-specific approach was formulated for predicting and monitoring the potential toxic impact of agricultural-use pesticides toward sensitive aquatic life inhabiting or frequenting the receiving water under investigation. This approach was developed in consideration of the characteristics of the study area, presumed seasonal use of certain pesticides within this region, the known aquatic persistence and toxicity of the pesticides examined in this report, laboratory and in-situ aquatic toxicity tests appropriate for future assessments at this site, and an understanding of costs and benefits associated with specific chemical analyses and aquatic toxicity tests.

The approach recommended was based upon the application of existing acute and chronic toxicity test methods suitable for receiving waters and sediments. Consideration was given to the types of aquatic organisms used in prospective toxicity tests as well as to the degree of standardization of available procedures, test sensitivity and reproducibility, and test cost. Reports and publications were sought which delineated methods applied previously for predicting and monitoring the potential toxic impact of agricultural-use pesticides toward sensitive aquatic life.

The approach recommended for future predictive and monitoring assessments of aquatic toxicity, attributable to agricultural pesticides in surface waters draining farmland within the region under investigation, is delineated in section 8.

3 PESTICIDE SELECTION, PROPERTIES AND USES

3.1 Pesticides Selected

The pesticides **dinoseb** and **endosulfan** were selected for investigation of potential aquatic toxicity within the study area. These pesticides were chosen due to their known high toxicity to aquatic life and high persistence in aqueous solution (see section 4), together with recent evidence for their common use in the Lower Fraser Valley of British Columbia (Moody 1989) and presence in certain regional farm ditch water and/or sediment samples throughout the year (Wan 1989).

3.2 Uses of Selected Pesticides

3.3.1 Dinoseb

Dinoseb (2-sec-butyl-4,6-dinitrophenol) has been registered for pesticide use in Canada since 1947 (Agriculture Canada 1989). It is distributed for use in oil or oil-water emulsions or as water-soluble salts. Trade names include DNBP, DNOSBP, DNSBP, DN 289, Basanite, Chemox, Gebutox, Dow General and Sinox General Weedkillers, Potato Top Killer 300, Potato Top Killer, Dytop Potato Top Killer, Topper Potato Top Killer, Yellow Stuff G Herbicide, Dinitro General Weedkiller EC Concentrate, VW&R Guardsman Weed & Top Killer Agricultural, Later's Dinoseb General Emulsifiable Herbicide, Dyanap Liquid Weed Killer, Pfizer Dinoseb 300 P.T.K. Agricultural (Spencer 1981; C. Ranger, pers. commun.). The amine formulation (e.g. Premerge - Dow Chemical) is no longer registered for use in Canada and the other formulations are presently under re-evaluation by Agriculture Canada (M. Edwards, pers. commun.). Within the U.S., a cancellation agreement

between the Environmental Protection Agency (EPA) and dinoseb manufacturers provides for elimination of all dinoseb uses after 1989 (Agriculture Canada 1989).

Dinoseb is normally described as a contact-action herbicide, although it has also been used as an insecticide and miticide (Norris et al. 1983, Windholz et al. 1983). In Canada, the major agricultural use of dinoseb (approximately 65 - 70% in 1988) is as a top-kill for potatoes. Other principal uses are for control of seedling weeds in certain crops (e.g. beans, corn, cucumbers, peas and potatoes) and for the pre-harvest desiccation of flax, soybeans and certain forage legume crops for seed production. Based on discussions with provincial agencies and growers, critically needed uses appear to be for weed control in peas, beans and cucumbers (Agriculture Canada 1989).

Within the Lower Fraser Valley, dinoseb is used to control weeds in cucumbers, peas, corn, potatoes and certain fruit crops (strawberries, grapes, bush fruits). It is also used as a pre-harvest top-kill for potatoes. A recent survey of pesticide distribution patterns for this region indicated that dinoseb was one of the herbicides in common use (Moody 1989).

3.2.2 Endosulfan

Endosulfan is registered for use in Canada under a number of trade names including the following: Thiodan, Thiodan 4 EC Insecticide, Thiodan 50-WP Insecticide, Thiodan-2 Zinbeb-5, Thimul, Thionex 50W Endosulfan Insecticide, Cyclodan, Thiofor, Malix, FMC 5462, Pfizer Endosulfan 400, Pfizer Endosulfan 50 W, VW&R Guardsman Maneb-Thiodan Dust, Wilson's Borer Kill Liquid Insecticide, and Clean Crop Endosulfan 4E Insecticide/Miticide (Spencer 1981; C. Ranger, pers. commun.). This chemical is one

of the remaining organochlorinated pesticides registered for the control of a broad spectrum of insect pests (Anon. 1975). It has been included on the U.S. EPA's restricted list, which limits its usage in that country (EPA 1980). However, significant commercial use of endosulfan for insect control on vegetables and fruits continues.

Endosulfan is among the insecticides most commonly used by Lower Fraser Valley vegetable growers (Moody 1989). Target species include aphids, rust mites and white flies. Vegetable crops to which endosulfan may be applied include beans, broccoli, brussel sprouts, cabbage, cauliflower, celery, corn, cucumbers, lettuce, peas, potatoes, squash, tomatoes and turnips (M. Edwards, pers. comm.). A recent survey of pesticide distribution and use patterns in the Lower Fraser Valley indicated that endosulfan is used regionally to protect cole crops, celery, eggplant, lettuce, peppers, potatoes, tomatoes and strawberries (Moody 1989). Depending upon the crop being protected, application rates may be 0.6 - 1.7 kg ai/ha[1] (Anon. 1975).

3.3 Properties of Selected Pesticides

3.2.1 Dinoseb

Dinoseb (chemical formula $C_{10}H_{12}N_2O_5$, molecular weight 240.2) is an orange-brown viscous liquid that is soluble in most organic solvents (Windholz et al. 1983). This pesticide is subject to volatilization from soil. Its residual life in warm, moist soils is reportedly only 3 to 5 weeks, and carryover from one season to the next is not expected (Klingman and Ashton 1975). Substantial decomposition in soil by microbial action can take place within

[1]Kilogram of active ingredient per hectare.

the first year following application (Norris et al. 1983). Norris et al. (1983) reported that dinoseb adsorbs strongly to highly acidic soil, although this pesticide is not tightly adsorbed to most agricultural soils.

Agriculture Canada has rated dinoseb high on their list of 86 pesticides identified as having potential to contaminate water. This rating was based on its very high propensity to leach from soil into water, the volume of this pesticide used within Canada, and other considerations (McRae 1989). Under neutral-pH conditions, the solubility of dinoseb in water has been measured as 52 mg/L (Melnikov 1971). The above properties indicate that dinoseb has a considerable potential to be transported to and solubilized in water.

3.2.2 Endosulfan

The chemical name for endosulfan is 6,7,8,9,10,10-hexachloro-1,5,5a,6,9,9a-hexahydro-6,9-methano-2,4,3-benzodioxathiepin-3-oxide. Its molecular formula is $C_9Cl_6H_6O_3S$ (mol. wt. 407.0).

Endosulfan is a light to dark brown crystalline solid with a terpene-like odour. It is formulated as a dust, emulsifiable concentrate, wettable powder, or in granular form (Berg 1976, Moody 1989). Technical grade endosulfan has a purity of 95% and is comprised of a mixture of two stereoisomers, referred to as alpha and beta or I and II and usually present in the ratio 70% I to 30% II.

This pesticide is stable to sunlight, but is susceptible to oxidation and hydrolyses slowly on contact with water (Spencer 1981). Under aerobic conditions, the principal degradation product of endosulfan in contact with vegetation or adsorbed to

soil is endosulfan sulfate (Anon. 1975, EPA 1980). Isomer I appears to decompose in soil quite rapidly (half-life reported of 60 days), whereas degradation of isomer II is much slower (reported half-life of 800 days). The rate of endosulfan degradation in soil is dependent upon the nature of the soil, and seasonally cool or cold temperatures reduce or prevent its degradation. Endosulfan sulfate is apparently stable in soil for several years (Anon. 1975).

Volatilization and leaching of endosulfan from soils appears to be much lower in soils containing large amounts of organic material. Isomer II may be more strongly adsorbed to soil than isomer I. Endosulfan exhibits a solubility in water (pH 7.2) of 60 to 150 ug/L (0.06 - 0.15 mg/L) (Anon. 1975).

4 LITERATURE REVIEW

4.1 Dinoseb

4.1.1 Toxicity to freshwater biota

Available information concerning the acute lethal toxicity (i.e. median lethal concentration; LC50) of dinoseb to freshwater fish and invertebrate species is shown in Table 1. Lowest test concentrations of dinoseb found to cause adverse sublethal toxic effects toward exposed fish (i.e. Lowest Observed Effect Concentration; LOEC) are presented in Table 2. A summary of lowest dinoseb strengths reported to cause acute lethal, acute sublethal, or chronic sublethal toxic effects toward freshwater biota is given in Table 3.

Dinoseb has been demonstrated to be acutely lethal (within 96 hours) to sensitive salmonid fish species (trout and salmon) at concentrations ranging from 32 to 1,400 ug/L (0.03 - 1.4 mg/L). The highest (least toxic) values reported are for dilution waters adjusted to pH 8.5; whereas the lowest (most toxic) reported values represent dilution water adjusted to pH 6.5 (Woodward 1976). Based on these LC50's, dinoseb was 30 - 40 times more toxic at pH 6.5 than at pH 8.5. These and earlier (Lipschuetz and Cooper 1961) findings demonstrate that the aquatic toxicity of dinoseb is highly pH-dependent, and that this herbicide is appreciably more toxic to freshwater life if discharged to acidic or neutral-pH waters. Other receiving-water characteristics including hardness (2X more toxic in very hard water, relative to soft water) and temperature can modify the aquatic toxicity of dinoseb (Woodward 1976); however receiving-water pH appears to be the single most important natural variable which may modify the aquatic toxicity of this pesticide.

TABLE 1. REVIEW OF ACUTE LETHAL TOXICITY OF DINOSEB TO FRESHWATER BIOTA.

Type of Organism	Test Species	Life Stage	Water Quality pH	Water Quality Hardness[3]	LC50[1] (ug/L)[2] 1 h	24 h	48 h	96 h	Reference
FISH	rainbow trout	fingerling	6.9	79	300	73	-[4]	-	Lipschuetz&Cooper 61
	rainbow trout	fingerling	8.0	79	1,500	300	-	-	Lipschuetz&Cooper 61
	cutthroat trout	fingerling	6.5	30-40	-	-	-	41	Woodward 1976
	cutthroat trout	fingerling	7.5	30-40	-	-	-	130	Woodward 1976
	cutthroat trout	fingerling	8.5	30-40	-	-	-	1,350	Woodward 1976
	lake trout	fingerling	6.5	30-40	-	-	-	32	Woodward 1976
	lake trout	fingerling	7.5	30-40	-	-	-	77	Woodward 1976
	lake trout	fingerling	8.5	30-40	-	-	-	1,400	Woodward 1976
	cutthroat trout	fingerling	7.8	40-48	-	-	-	550	Woodward 1976
	cutthroat trout	fingerling	7.8	160-180	-	-	-	340	Woodward 1976
	cutthroat trout	fingerling	7.8	280-320	-	-	-	280	Woodward 1976
	cutthroat trout	fingerling	7.4	40-44	-	-	-	71	Mayer&Ellersieck 1986
	speckled trout	fingerling	-	-	-	180	-	110	Christie&Penney 1972
	coho salmon	smolt	7.7	101	-	190[5]	-	-	Lorz et al. 1979
	coho salmon	smolt	7.0	100	-	-	60[5]	-	Lorz et al. 1979
	Atlantic salmon	fingerling	-	-	-	-	-	700	Zitko et al. 1976
	blacknose dace	-	8.0	79	-	240	-	-	Lipschuetz&Cooper 61
	redside shiner	-	7.6	18	-	160	-	-	Webb 1951
	redside shiner	-	8.2	105	-	240	-	-	Webb 1951
	fathead minnow	-	7.5	45	-	800	700	700	Call et al. 1984
	fathead minnow	-	7.0-7.4	77	-	-	-	230	Gersich & Mayes 1986
INVERTEBRATE	Daphnia magna	neonate	7.6-8.2	77	-	-	240	-	Gersich & Mayes 1986

[1]Median Lethal Concentration (i.e. the concentration estimated to be lethal to 50% of the test organisms).

[2]Micrograms per liter (parts per billion).

[3]Total hardness, mg $CaCO_3$/L.

[4]Not determined/not indicated.

[5]Amine salt formulation.

TABLE 2. REVIEW OF SUBLETHAL TOXICITY OF DINOSEB TO FRESHWATER BIOTA.

Type of Organism	Test Species (Life Stage)	Duration of Exposure (days)	Test pH	Nature of Response	LOEC[1] (ug/L)[2]	Reference
FISH	coho salmon (smolt)	2	7.0	inhibited downstream migration	40[3]	Lorz et al. 1979
	coho salmon (smolt)	7	7.0	minor histopathologies for liver, kidney, gill	40[3]	Lorz et al. 1979
	fathead minnow (egg, fry)	64	7.5	reduced fry survival + weight gain	49	Call et al. 1984
	lake trout (egg, alevin, fry)	81	7.4	impaired fry survival	10	Woodward 1976
	lake trout (egg, alevin, fry)	81	7.4	reduced growth	<0.5	Woodward 1976

[1]Lowest Observed Effect Concentration.

[2]Micrograms per liter (parts per billion).

[3]Amine salt formulation.

TABLE 3. SUMMARY OF LOWEST CONCENTRATIONS OF DINOSEB AND ENDOSULFAN REPORTED TO AFFECT THE SHORT-TERM SURVIVAL OF FRESHWATER BIOTA OR TO CAUSE ACUTE OR CHRONIC SUBLETHAL TOXIC EFFECTS.

Nature of Toxic Effect	Lowest Observed Effect Concentration	
	Dinoseb (ug/L)[1]	Endosulfan[2] (ug/L)
ACUTE LETHAL	32	0.3
ACUTE SUBLETHAL	40	0.1
CHRONIC SUBLETHAL	0.5	0.4

[1]Micrograms per liter (parts per billion).

[2]Technical grade or as measured in forumulated product.

The findings of Christie and Penny (1972) and Call et al. (1984) indicate that the duration of fish-exposure to dinoseb does not modify its toxicity appreciably, at least for exposures of 1 to 4 days' duration. Lorz et al. (1979) determined that increasing the duration of exposure of coho salmon smolts from 6 to 16 days resulted in a slight increase in the toxicity of dinoseb (amine salt), with LC50 values decreasing from 60 ug/L for 2 days' exposure to values of 50 and 30 ug/L, respectively. Similarly, Woodward (1976) found that the lethal toxicity of dinoseb to cutthroat and lake trout was only slightly increased when exposures were increased from 4 to 10 days (flowthrough tests). Shorter (e.g. 1 - 2 h) periods of exposure suggest a lesser toxicity (Lipschuetz and Cooper 1981); however the effects of transient exposures could result in subsequent mortalities if fish were transferred or otherwise returned to uncontaminated water. The effects of transient (few hours) exposure to this pesticide on subsequent survival or physiological condition and performance of fish have not been reported.

No major differences in acute tolerance of fish to dinoseb, due to species or life stage, were apparent from the data reviewed (Table 1). Differences undoubtedly exist, although few studies have examined the modifying influences of fish age and species on susceptibility to dinoseb under identical water quality conditions.

Gersich and Mayes (1986) determined that dinoseb was acutely lethal to the cladoceran invertebrate, Daphnia magna (a sensitive freshwater invertebrate used routinely in Canada and elsewhere for aquatic toxicity tests) at a concentration of 240 ug/L. No other studies were found which reported the sensitivity of this or other freshwater invertebrates to dinoseb. Similarly, no reports of algal susceptibility to this herbicide were found.

Few studies have examined the sublethal toxicity of dinoseb to freshwater fish species. Lorz et al. (1979) reported that coho salmon smolts exposed for 2 days to dinoseb (amine salt formulation) concentrations as low as 40 ug/L displayed some degree of inhibited downstream movement when released to a stream following their exposure. These investigators also noted minor histological damage to the liver, kidney and gill tissue of coho salmon smolts held in 40 ug/L dinoseb for 7 days (Table 2). Partial life-cycle studies with fathead minnows revealed reduced weight gain and impaired survival at strengths of 49 ug/L or higher (Call et al. 1984). Chronic (81-day) partial life-cycle tests with lake trout showed impaired survival at 10 ug/L, and reduced growth at all dinoseb strengths tested including 0.5 ug/L (Woodward 1976). No reports of studies examining acute or chronic sublethal effects of this herbicide toward freshwater invertebrates or algae were found in the literature searched.

The above review indicates that, under laboratory conditions, dinoseb is highly toxic to sensitive salmonid and other freshwater fish species. In soft, slightly acidic water, this herbicide can be acutely lethal to fish at concentrations as low as 30 - 40 ug/L (0.03 - 0.04 mg/L). For neutral-pH water, salmonid fish exposed to strengths as low as 70 ug/L can be killed within 24 h (Table 1). Acute sublethal effects (e.g. modified migrant behaviour) can occur due to short-term exposure to concentrations as low as 40 ug/L, and chronic sublethal effects (i.e. reduced growth) can result from prolonged exposure to dinoseb at strengths as low as 0.5 ug/L (0.0005 mg/L) (Tables 2 and 3).

4.1.2 Bioaccumulation in fish

Few studies have examined the bioaccumulation of dinoseb in fish or other aquatic species. Based on the physicochemical properties of dinoseb and other alkyldinitrophenols, Zitko et al. (1976) concluded that such compounds are not likely to be bioaccumulated or biomagnified. Lorz et al. (1979) detected low (0.1 - 1.4 ug/kg, wet weight basis) concentrations of dinoseb in the gill, spleen, gall bladder, liver and kidney of coho salmon smolts exposed to dinoseb (amine formulation) concentrations of 20 to 60 ug/L for periods of 16 or 6 days, respectively. Dinoseb concentrations in skin and muscle were below detection limits. Based upon the findings of concurrent bioaccumulation and toxicity studies with guppies (Poecilia reticulata) exposed for 24 - 96 h to dinoseb and other substituted phenols (including chlorinated phenolic compounds), Saarikoski and Viluksela (1982) concluded that dinoseb was accumulated clearly less than the chlorinated compounds, and that a steady-state concentration in whole-body tissue was reached by 24 h. These researchers also concluded that the high toxicity of the dinitrophenols tested (including dinoseb) was probably not due to their being accumulative but rather to their high intrinsic toxicity.

Call et al. (1984) studied the uptake and clearance of dinoseb in whole-body tissue of fathead minnows (Pimephales promelas) exposed to 0.6 or 7.2 ug/L for 24 days and subsequently transferred to fresh water for 14 days. A bioconcentration factor (i.e. concentration in tissue divided by concentration in water) of only 1.4 was determined for these fish, with a steady-state concentration evident within the first 24-h period of exposure. Elimination of dinoseb was rapid upon transfer to uncontaminated water, with 71 and 96% cleared after 1 and 14 days, respectively. Separate studies with rainbow trout injected

with radiolabelled dinoseb also demonstrated that this herbicide can be readily eliminated from the fish, as 90% of the herbicide was recovered in the water within 24 h (Call et al. 1984).

From the foregoing, it is evident that dinoseb can accumulate rapidly (within 24 h) in certain tissues of exposed fish (primarily the gill, spleen, gall bladder, liver and kidney). However, a steady-state tissue burden is reached within a short period of exposure (implying metabolism and excretion of this pesticide), the extent of bioaccumulation is low, and dinoseb is rapidly cleared from the fish upon its return to uncontaminated water.

4.1.3 Aquatic persistence

Some evidence is available from the chemical analytical data associated with toxicity tests, which indicates the degree of persistence of dinoseb or its metabolites in water. Zitko et al. (1976) reported a recovery of no less than 80% of nominal concentrations after fish-exposure to test concentrations for 48 h under static conditions. As part of the toxicity tests undertaken with dinoseb using cutthroat trout, Woodward (1976) aged one set of test solutions under laboratory conditions for four weeks prior to fish exposure and found no appreciable change in toxicity (96-h LC50, 71 ug/L for fresh solutions; 87 ug/L for solutions aged 4 wk). Similarly, Call et al. (1984) found that dinoseb concentrations in water declined by only 21% during a 40-day test period.

The above results indicate that dinoseb is persistent in aqueous solution under laboratory conditions, and that the toxicity of

dilutions of this herbicide remains relatively unchanged even when test solutions are held for up to 28 days. However, these findings should not be considered as representative of the persistence of dinoseb entering natural freshwater bodies, inasmuch as the rates of photochemical and microbial degradation in receiving waters could be substantially more significant than that under laboratory conditions. For instance, Crosby and Li (1969) reported that dinoseb in water was readily photolabile when irradiated by ultraviolet light. Other receiving-water conditions including perhaps the presence of suspended solids and chelating organic material (e.g. humic and fulvic acids) could also modify the aqueous persistence of dinoseb and its bioavailability within the natural environment.

4.2 Endosulfan

4.2.1 Toxicity to freshwater biota

Known information indicating the acute lethal toxicity of endosulfan to freshwater fish and invertebrate species is tabulated in Table 4. Test concentrations of endosulfan demonstrated previously to cause adverse sublethal toxic effects toward exposed freshwater biota are listed in Table 5. Summary data concerning the lowest strengths of endosulfan previously found to cause acute lethal, acute sublethal, or chronic sublethal toxic effects are given in Table 3. Readers wishing additional information regarding the aquatic toxicity of endosulfan are directed to the review articles by the National Research Council of Canada (Anon. 1975) and the U.S. Environmental Protection Agency (EPA 1980).

TABLE 4. REVIEW OF ACUTE LETHAL TOXICITY OF ENDOSULFAN TO FRESHWATER BIOTA.

Type of Organism	Test Species	Life Stage	Water Quality pH	Water Quality Hardness[3]	Test Material	LC50[1] (ug/L)[2] 1 h	24 h	48 h	96 h	Reference
FISH										
	rainbow trout	fingerling	7.4	45	Thiodan	-[4]	2.1	1.0	0.3	Schoettger 1970
	rainbow trout	fingerling	6.8-7.1	54	technical	-	-	-	0.3[5]	Lemke 1981
	rainbow trout	fingerling	7.6	46	technical	-	-	-	0.4[5]	Lemke 1981
	rainbow trout	fingerling	7.7	48	technical	-	-	-	0.7[5]	Lemke 1981
	rainbow trout	fingerling	-	30	technical	-	-	-	1.6	Nebeker et al. 1983
	rainbow trout	fingerling	-	27	technical	-	-	-	0.3[5]	Nebeker et al. 1983
	rainbow trout	fingerling	7.1	44	technical	-	2.3	-	1.1	Mayer&Ellersieck 1986
	fathead minnow	fingerling	-	36	technical	-	-	-	1.3	Nebeker et al. 1983
	fathead minnow	fingerling	-	39	technical	-	-	-	1.0[5]	Nebeker et al. 1983
	fathead minnow	juvenile	7.3-7.8	47	-	17.	1.8	1.7	1.3	Kleiner et al. 1984
	fathead minnow	fingerling	7.1	44	technical	-	2.4	-	1.5	Mayer&Ellersieck 1986
	bluegill	fingerling	7.1	44	technical	-	3.3	-	1.2	Mayer&Ellersieck 1986
	bluegill	juvenile	7.3-7.8	47	-	17.	2.2	-	-	Kleiner et al. 1984
	channel catfish	fingerling	7.1	44	technical	-	1.8	-	1.5	Mayer&Ellersieck 1986
	Heteropneustes sp.	juvenile	8.4	152	-	-	-	-	1.1[5]	Rao & Murty 1982
	Mystus cavasius	fingerling	8.4	152	-	-	-	-	1.9[5]	Rao & Murty 1982
	Mystus vittatus	fingerling	8.4	152	-	-	-	-	2.2[5]	Rao & Murty 1982
	Labeo rohita	juvenile	8.4	152	technical	-	-	-	1.1[5]	Rao et al. 1980
	Labeo rohita	juvenile	8.4	152	isomer I	-	-	-	0.3[5]	Rao et al. 1980
	Labeo rohita	juvenile	8.4	152	isomer II	-	-	-	7.1[5]	Rao et al. 1980
	major carp	juvenile	8.4	123	technical	-	-	-	1.3	Swarup et al. 1981
	major carp	juvenile	8.4	123	isomer I	-	-	-	0.6	Swarup et al. 1981
	major carp	juvenile	8.4	123	isomer II	-	-	-	8.8	Swarup et al. 1981
	Channa punctata	juvenile	8.4	152	technical	-	-	-	4.8	Devi et al. 1981
	Channa punctata	juvenile	8.4	152	isomer I	-	-	-	0.2	Devi et al. 1981
	Channa punctata	juvenile	8.4	152	isomer II	-	-	-	6.6	Devi et al. 1981
	mosquitofish	adult	7.5	32	Thiodan 35EC	-	6.	4.8	3.2	Joshi & Rege 1980
	mosquitofish	adult	7.5	32	technical	-	12.	9.4	8.	Joshi & Rege 1980
	mosquitofish	adult	7.8	12	Thiodan	-	-	-	1.3	Naqvi & Hawkins 1988

[1]Median Lethal Concentration (i.e. the concentration estimated to be lethal to 50% of the test organisms).

[2]Micrograms per liter (parts per billion).

[3]Total hardness, mg $CaCO_3$/L.

[4]Not determined/not indicated.

[5]Flowthrough test (replacement of solutions throughout the test).

TABLE 4. REVIEW OF ACUTE LETHAL TOXICITY OF ENDOSULFAN TO FRESHWATER BIOTA (cont'd.)

Type of Organism	Test Species	Life Stage	Water Quality pH	Water Quality Hardness[3]	Test Material	LC50[1] (ug/L)[2] 1 h	24 h	48 h	96 h	Reference
INVERTEBRATE										
	Daphnia magna	-[4]	7.4	45	Thiodan	-	68.	62.	56.	Schoettger 1970
	Daphnia magna	neonate	6.8-7.1	35	-	-	-	166.	-	Macek et al. 1976
	Daphnia magna	neonate	6.8-7.1	54	technical	-	-	378.	-	Lemke 1981
	Daphnia magna	neonate	7.6	46	technical	-	-	158.	-	Lemke 1981
	Daphnia magna	neonate	7.7	48	technical	-	-	282.	-	Lemke 1981
	Daphnia magna	neonate	-	33	technical	-	-	271.	-	Nebeker et al. 1983
	Daphnia magna	neonate	-	35	technical	-	-	343.	-	Nebeker et al. 1983
	Gammarus lacustris (freshwater amphipod)	mature	7.1	44	technical	-	9.2	-	5.8	Mayer&Ellersieck 1986
	Ischura sp. (damselfly)	nalad	7.4	45	Thiodan	-	275.	175.	107.	Schoettger 1970
	Enallagma sp. (aquatic insect)	nymph	6.9-7.2	120	technical	-	29.	21.	18.	Gopal et al. 1981
	Lamellidens corrianus (bivalve mollusc)	adult	7.2-8.8	120-165	Thiodan 35EC	-	-	-	17-44	Mane & Muley 1984
	L. marginalis	adult	7.2-8.8	120-165	Thiodan 35EC	-	-	-	2-22	Mane & Muley 1984

[1]Median Lethal Concentration (i.e. the concentration estimated to be lethal to 50% of the test organisms).

[2]Micrograms per liter (parts per billion).

[3]Total hardness, mg $CaCO_3$/L.

[4]Not determined/not indicated.

TABLE 5. REVIEW OF SUBLETHAL TOXICITY OF ENDOSULFAN TO FRESHWATER BIOTA.

Type of Organism	Test Species	Life Stage	Test Material	Duration of Exposure (days)	Nature of Response	LOEC[1] (ug/L)[2]	Reference
FISH							
	Labeo rohita	juvenile	technical	0.1	increased oxygen consumption	0.1	Rao et al. 1980
	Macrognathus aculeatum	-[3]	technical	0.1	decreased oxygen consumption	1.0	Rao et al. 1981
	Macrognathus aculeatum	-	technical	0.1	decreased nitrogen excretion	1.0	Rao et al. 1981
	Heteropneustes fossilis	adult	technical	0.1-0.3	increased blood lactate	1.5	Singh & Srivastava 81
	Heteropneustes fossilis	adult	technical	0.1-4	increased blood glucose	1.5	Singh & Srivastava 81
	Heteropneustes fossilis	adult	technical	0.1-4	decreased muscle glycogen	1.5	Singh & Srivastava 81
	Clarias batrachus	-	technical	0.2-2	increased blood glucose	10.0	Gopal et al. 1980
	Channa striatus	-	Thiodan	2.0	intestinal histopathology	0.8	Jauhar&Kulshrestha 83
	Channa striatus	-	Thiodan	2.0	moderate gill histopathology	0.8	Jauhar&Kulshrestha 83
	Channa striatus	adult	Thiodan	2.0	altered histology of testes	0.8	Arora&Kulshrestha 84
	Labeo rohita	juvenile	technical	4.0	increased vacuolation of liver	0.9	Rao et al. 1980
	Channa punctata	juvenile	isomer I	4.0	decreased muscle protein + lipid	0.05	Murty & Devi 1982
	Channa punctata	juvenie	isomer I	4.0	increased kidney protein	0.05	Murty & Devi 1982
	Channa striatus	-	Thiodan	8-30	marked intestinal histopathology	0.8	Jauhar&Kulshrestha 83
	Channa striatus	-	Thiodan	8-30	marked gill histopathology	0.8	Jauhar&Kulshrestha 83
	Channa striatus	adult	Thiodan	8-30	altered histology of testes	0.8	Arora&Kulshrestha 84
	Channa striatus	adult	Thiodan	8-30	reduced gonadosomatic index	0.8	Arora&Kulshrestha 84
	Clarias batrachus	-	technical	10	increased hematocrit + hemoglobin	2.0	Gopal et al. 1982
	Channa gachua	-	-	15-30	inhibited liver ATPase activity	3.7	Shama 1988
	tropical cichlid	adult	Thiodan	20	histopathology of pituitary	1.0	Shukla & Pandey 1986
	tropical cichlid	adult	Thiodan	20	histopathology of ovary	1.0	Shukla & Pandey 1986
	tropical cichlid	adult	Thiodan	28	delayed breeding behaviour	0.6	Matthiessen&Logan 84
	tropical cichlid	adult	Thiodan	28	histology of ovary and testes	>0.5	Matthiessen&Logan 84
	fathead minnow	egg to adult	Thiodan	280	reduced hatch + adult survival	0.4	Macek et al. 1976
INVERTEBRATE							
	Daphnia magna	3 generations	Thiodan	64	reduced chronic survival	7.0	Macek et al. 1976

[1]Lowest Observed Effect Concentration.

[2]Micrograms per liter (parts per billion).

[3]Not determined/not indicated.

TABLE 5. REVIEW OF SUBLETHAL TOXICITY OF ENDOSULFAN TO FRESHWATER BIOTA (cont'd.)

Type of Organism	Test Species	Life Stage	Test Material	Duration of Exposure (days)	Nature of Response	LOEC[1] (ug/L)[2]	Reference
ALGAE							
	Chlamydomonas reinhardtii	growth phase	technical	0.1-0.3	reduced cell population	10,000	Gandhi et al. 1987
		growth phase	technical	0.1	inhibited photosynthesis	>10,000	Gandhi et al. 1987
	Anabaena sp.	growth phase	technical	0.1	inhibited photosynthesis	20,000	Tandon et al. 1988
	Aulosira fertilissima	growth phase	technical	0.1	inhibited photosynthesis	1,000	Tandon et al. 1988
	Chlamydomonas reinhardtii	growth phase	technical	3	inhibited growth	4,000	Gandhi et al. 1987
	Anabaena sp.	growth phase	technical	5-30	inhibited growth	1,000	Tandon et al. 1988
	Aulosira fertilissima	growth phase	technical	10-30	inhibited growth	1,000	Tandon et al. 1988

[1]Lowest Observed Effect Concentration.

[2]Micrograms per liter (parts per billion).

The acute lethal toxicity of technical-grade endosulfan has been well studied using a variety of freshwater fish species including rainbow trout. For this sensitive species, 96-h LC50 values as low as 0.3 ug/L (0.0003 mg/L) have been reported for exposed fingerlings by at least three independent investigators (Table 4). Median lethal values as high as 1.6 ug/L have also been determined using fingerling rainbow trout. Comparative data using static (no replacement of solutions) and flowthrough test conditions indicate that much of this difference in toxicity is due to the loss of endosulfan from solution (and therefore its lesser toxicity) under static conditions (EPA 1980, Nebeker et al. 1983). The least toxic (highest) LC50 value for technical-grade endosulfan or one of its formulations, noted for tests using various free-swimming stages (fingerling, juvenile and adult) of a variety of freshwater fish species, was 8 ug/L. Unlike these findings, rainbow trout eggs have been shown to survive acute or prolonged exposure to endosulfan strengths as high as 50,000 ug/L (Schoettger 1970).

The relative toxicity of technical-grade endosulfan and its two isomers has been investigated in recent years. Tests with three separate species of fish found that isomer I (alpha) was 16 - 33 times more toxic than isomer II (beta), and that the toxicity of technical endosulfan (normally a mixture of 70% I and 30% II) was intermediate (Rao et al. 1980, Devi et al. 1981, Swarup et al. 1981) (Table 4).

Acute lethal tests with freshwater invertebrates indicate a considerable species-dependent variation in sensitivity to endosulfan. For the test results reviewed, LC50 values ranged from 2 ug/L (freshwater bivalve mollusc) to 378 ug/L (Daphnia magna). Numerous tests with D. magna have indicated that this filter-feeding cladoceran (normally as sensitive to aquatic contaminants as salmonid fish) is two to three orders of magnitude less sensitive to endosulfan than salmonid or other

fish species. The present data base shows that freshwater invertebrates are generally more acutely tolerant of endosulfan than fish (Table 4). However, as some of the aquatic invertebrate species known to be contaminant-sensitive (e.g. mayfly nymphs) have yet to be studied using this insecticide, no firm conclusion to this effect should be made without further comparative tests.

The duration of fish exposure to endosulfan undoubtedly will influence their ability to survive. As with dinoseb, definitive information is presently unavailable regarding the effect of brief (e.g. hours, few days) pesticide exposure on their prolonged survival (i.e. if death is not immediate). One study (Schoettger 1970) reports that fingerling rainbow trout able to withstand a 5-day exposure to endosulfan died within a week following their transfer to uncontaminated water. Unlike this finding, Kleiner et al. (1984) found that surviving fathead minnows and bluegills, challenged with brief (1 - 20 h) exposure to otherwise lethal (if exposures were continued) concentrations of endosulfan, recovered quickly when transferred to pesticide-free water, and that nearly all fish deaths occurred during the exposure period. Although time-dependent LC50 values show a tendency to decline somewhat within a typical 96-h test period (Table 4), these values generally reach a minimum in less than 120 h (Anon. 1975).

Some information is available regarding the modifying influence of differing water quality conditions on the acute lethal toxicity of endosulfan to freshwater organisms. Unlike dinoseb, differences in dilution-water pH within the range typical of natural waters do not appear to cause a marked difference in endosulfan toxicity (Table 4; Lemke 1981). Nor does water hardness (Table 4; Anon. 1975). However, since no studies were found which examined the influence of pH on the toxicity of endosulfan under otherwise identical conditions, a modifying

influence of dilution-water (and receiving-water) pH on endosulfan toxicity should not be dismissed. The influence of water temperature on endosulfan toxicity has received some attention. Comparative tests with rainbow trout, conducted at controlled temperatures ranging from 1.5 to 10 C, found that endosulfan was three times as toxic at the higher temperature. Similarly, endosulfan was twice as toxic to Daphnia magna when tested at 19 C compared to 10 C (Schoettger 1970). Other studies (Singh and Narain 1982) also suggest that this pesticide is somewhat more toxic at warmer temperatures.

A number of researchers have examined fish for toxic responses (biochemical, physiological, histopathological, behavioural) caused by their exposure to sublethal concentrations of endosulfan. The preponderance of sublethal-effect data relate to tropical fish species, and no reports are available which indicate sublethal concentrations of this insecticide harmful to salmonid fish species.

Acute ($\leq$4 days) exposures to endosulfan concentrations as low as 0.05 ug/L have been reported to cause biological effects (Murty and Devi 1982; Table 5). Responses including acute stress reactions and altered rates of oxygen consumption and nitrogen excretion have been noted for juvenile or adult fish exposed to endosulfan strengths ranging from 0.1 to 1.5 ug/L for periods as brief as 2 - 3 h (Rao et al. 1980, 1981; Singh and Srivastava 1981). Endosulfan concentrations of less than 1 ug/L have been shown to cause histopathologies in fish gills, livers, intestines and testes within 2 - 4 days, with more pronounced effects due to extended (up to 30 days) exposures (Jauhar and Kulshrestha 1983, Arora and Kulshrestha 1984; Table 5).

Prolonged (20-day) exposure of tropical cichlids to 1 ug/L caused marked changes in the histology of their pituitary glands and

ovaries (Shukla and Pandey 1986). Delayed breeding behaviour was also noted for this fish species when exposed to strengths as low as 0.6 ug/L (Matthiessen and Logan 1984). Macek et al. (1976) chronically exposed fathead minnows to endosulfan in complete life-cycle studies. Concentrations of 0.4 ug/L and higher caused the reduced survival of adults and impaired hatching success of their progeny.

Reports of sublethal effects of endosulfan toward freshwater invertebrate species are restricted to chronic-exposure studies using Daphnia magna. Macek et al. (1976) found that, for three generations of daphnids tested, their chronic survival was reduced significantly by endosulfan concentrations of 7 ug/L and higher. Using somewhat different test procedures, other investigators have reported chronic LOEC values for this species ranging from 20 to 154 ug/L (Nebeker 1982).

Freshwater algae do not appear to be particularly sensitive to endosulfan or its metabolites (Anon. 1975, EPA 1980). Five-day studies with Chlorella sp. or Scenedesmus sp. exposed to endosulfan and its metabolites at concentrations below 2,000 ug/L showed no effects on biomass production, photosynthesis or rates of cell division (Anon. 1975). A separate study with Chlorella vulgaris reported that 10,000 ug/L did not affect the growth of this green algae (EPA 1980). More recent investigations by Gandhi et al. (1987) and Tandon et al. (1988) have shown a similar resistance of freshwater algae to endosulfan, with LOEC's ranging from 1,000 to 20,000 ug/L (Table 5).

In summary, endosulfan is extremely toxic to freshwater fish, with acute lethal effects evident for rainbow trout at strengths as low as 0.3 ug/L (technical grade) and at somewhat higher concentrations (1 - 2 ug/L) for numerous other fish species (Table 4). Although studies indicating threshold sublethal strengths harmful to salmonid fish have not been found in the

published literature, reports have been identified which show acute and chronic sublethal toxic responses of other fish species to endosulfan at strengths as low as 0.1 and 0.4 ug/L, respectively (Tables 3 and 5). Available information for freshwater invertebrate and algal species indicates a somewhat greater (invertebrates) or appreciably greater (algae) tolerance to this insecticide than that demonstrated for fish.

4.2.2 Bioaccumulation in fish

No studies were found which examined the bioaccumulation of endosulfan in tissues of salmonid fish. Using radiolabelled Thiodan, Schoettger (1970) found that 20 ug/L endosulfan accumulated rapidly (within 2 - 9 h of exposure) in tissues of western white suckers (Catastomus commersoni), with highest concentrations evident in the liver and lowest accumulations in the skin and muscle. Levels in all tissues plateaued in less than 12 h, at which time endosulfan residues were approximately 11 mg/kg and 1 mg/kg (dry weight basis) for liver and muscle, respectively. Studies with goldfish (Carrasius auratus) exposed to 7 ug/L endosulfan for up to 20 days indicated highest concentrations (5 - 13 mg/kg; wet weight basis) in the liver, brain and peritoneal fat, with lower (1.7-2.5 mg/kg) levels detected in muscle and skin (Schoettger 1970). Residues in these fish at 20 days were no greater than those for fish exposed for 11 days or less. Additional studies with goldfish demonstrated a whole-body residue of 0.4 mg/kg (wet weight) after 5 days' exposure to 1 ug/L endosulfan, and a half-life for tissue clearance of approximately 3 days following fish transfer to uncontaminated water (Anon. 1975). After 14 days, tissue concentrations were less than 1% of levels accumulated during the

5-day exposure period. Other reports of tissue clearance rates for endosulfan were not found in the literature reviewed.

Studies with western white suckers indicated that toxic endosulfan sulfate was an intermediate metabolite in fish exposed to endosulfan, with the appreciably less-toxic endosulfan diol (alcohol) excreted in liver bile (Schoettger 1970). Results from additional studies with a number of tropical fish species suggested that the liver and kidney were principal storage organs for endosulfan and its metabolites, and that these organs played a key role in the elimination of this pesticide (Rao et al. 1980, 1981; Devi et al. 1981; Swarup et al. 1981; Rao and Murty 1982; Novak and Ahmad 1989). Metabolites detected in fish tissues and liver bile included endosulfan sulfate, diol, ether and lactone.

The above information, while deficient for salmonid fish species, indicates that, relative to certain other organochlorine compounds, the degree of bioaccumulation of endosulfan in fish tissues is low. Uptake and accumulation in edible tissue (muscle, skin) is particularly low. Metabolites of this pesticide are readily formed and rapidly excreted from fish, and metabolic clearance is rapid if fish are returned to clean water.

4.2.3 Aquatic persistence

A few studies have examined the stability and persistence of dilute aqueous solutions of endosulfan. Schoettger (1970) found that solutions containing 20 ug/L Thiodan, if stored at pH 6.4 or pH 7.4 and 19 C for periods of up to 240 h, were just as toxic to fish as the unaged solutions. Unlike this finding, solutions stored at pH values $\geq$8.4 were not toxic or only slightly toxic after 72-h aging. Laboratory tests with endosulfan (10 ug/L) solutions held in glass containers at room temperature and

artificial or natural lighting conditions showed the disappearance of isomers I and II by 4 weeks, and the formation of endosulfan diol within 1 week (Anon. 1975). Greve and Wit (1971) demonstrated that endosulfan can be degraded by aquatic microorganisms when the water has a high dissolved oxygen content and a pH $\geq$7. For instance, at pH 7 and 20 C, the half-life of endosulfan in air-saturated water was approximately 1 week.

From these observations, it appears that dilute solutions of endosulfan in water can be degraded reasonably rapidly (few days to few weeks) if water temperatures are warm (e.g. 19 - 20 C) and air-saturated, pH values are neutral or alkaline, and microorganisms are present. However, except for the limited work by Schoettger (1970) which compared the acute lethal toxicity of fresh solutions of endosulfan with those aged at differing pH values, the above studies indicating loss of endosulfan (isomers I and II) from solution do not provide information regarding the formation of toxic metabolites (e.g. endosulfan sulfate) and their persistence in aqueous solution.

5 DESCRIPTION OF STUDY AREA

5.1 General

The Nicomekl River was chosen as the receiving waterbody to be investigated. This river drains a large portion of Lower Fraser Valley farmland within the Fraser River Estuary Management Program region of concern (FREMP 1986). Approximately 34 km in length, the headwaters of the Nicomekl River originate in uplands 4 km east of Langley, B.C.. The river flows initially NW for 3 km, turns abruptly SW for an additional 3 km, bisects the City of Langley east to west, then travels WSW until it reaches Mud Bay (the northeasterly extension of Boundary Bay, which faces southeast onto the southern Strait of Georgia) (Figure 1).

The Nicomekl River shares an arable valley with the Serpentine River (approximately 2 km to the north) which extends 12 km eastward from Mud Bay to Cloverdale, B.C.. Moving upriver to a point 2 km southwest of Cloverdale, the Serpentine River bends sharply northward to drain the basin lying north of Cloverdale and south of the Fraser River (in the general vicinity of Barnston Island), whereas the Nicomekl River continues eastward (lying 2 km south of Cloverdale) until it bisects Langley.

The Nicomekl River has a drainage area of 175 km^2. Within this drainage, the two main tributaries of the Nicomekl R. are Anderson Creek (confluence at the western boundary of Langley) and Murray Creek (entering at eastern boundary of Langley) (Figure 1). Historical average daily river flows, as measured at a gauging site 0.5 km downstream of Anderson Creek, have ranged from 0.13 m^3/s to 35.4 m^3/s. Seven-day average low flow values measured at this station were: 10-year, 0.13 m^2/s; two-year, 0.24

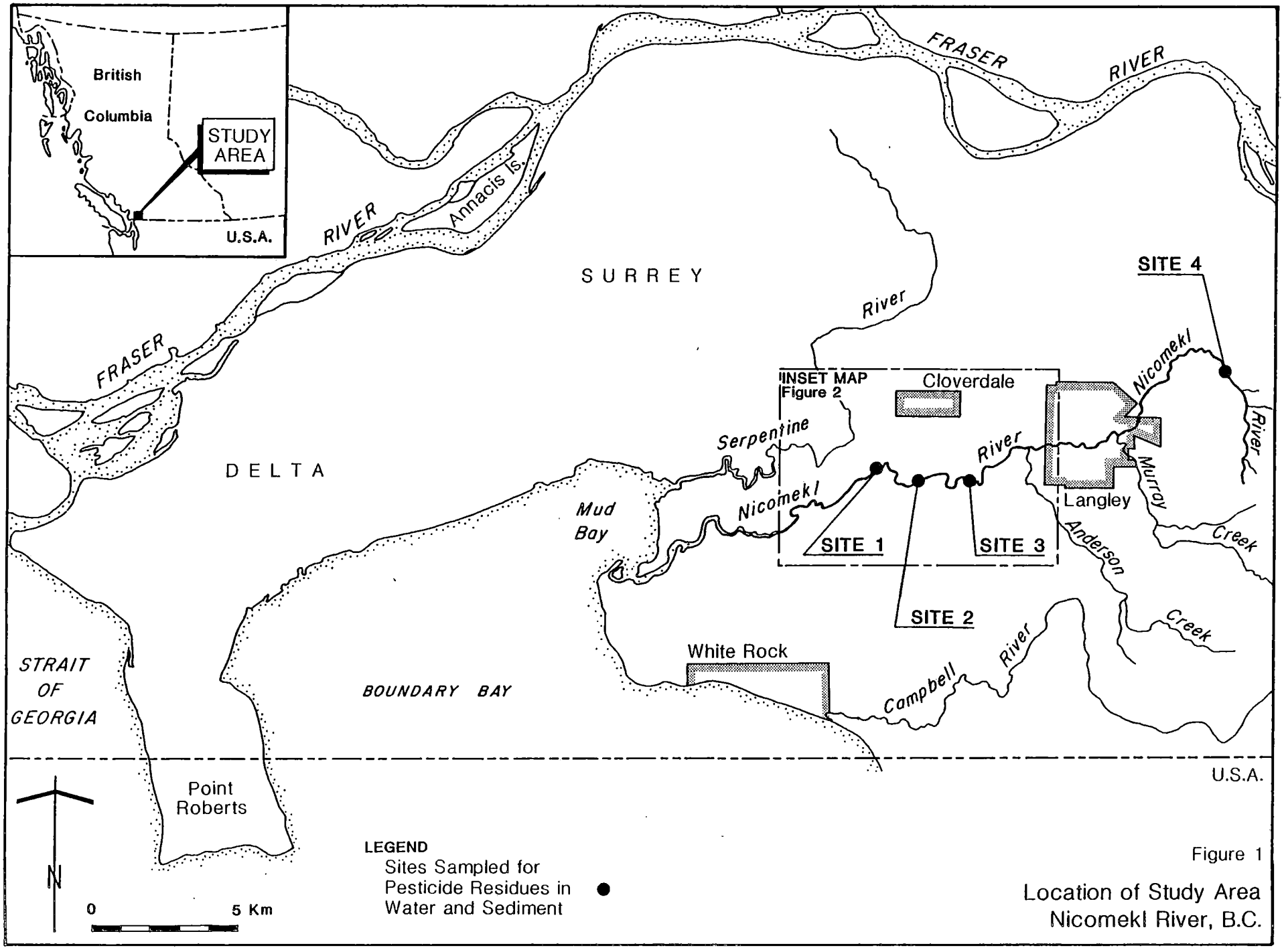

Figure 1

Location of Study Area
Nicomekl River, B.C.

m^3/s. For Anderson Creek, average daily flows near the confluence ranged from 0.11 m^3/s to 17.2 m^3. In Murray Creek, seasonal flows near the river juncture ranged from daily average values of 0.01 m^3/s to 14.45 m^3/s (Holms and Swain 1987). Except for the uplands around Anderson Creek which are pervious to groundwater flow, the drainage basin of the Nicomekl River is underlain by stoney marine clays, and is thought to drain primarily by surface water runoff (Holms and Swain 1987; H. Liebscher, personal commun.).

5.2 Wastewater Discharges and Water Quality

In addition to the agricultural ditchwater flows entering the Nicomekl River at numerous locations (section 5.4), a number of wastewaters discharging to this river have been identified (Holms and Swain 1987, Swain and Holms 1988). Stormwater runoff from Langley enters the Nicomekl River at various sites, posing a source for entry of aquatic contaminants (e.g. lead, zinc, oil etc.). Stormwater runoff from a bulk petroleum storage plant also enters the river at a point 4 km downstream from its confluence with Anderson Creek (maximum permitted flow, 430 m^3/day). A poultry processing plant located near the headwaters of Anderson Creek is permitted to discharge a maximum flow of 114 m^3/day to the creek and to spray-irrigate wastewater onto adjacent land. Leachate from a discontinued landfill site situated 1 km south of the river between Anderson and Murray Creeks discharges into a tributary of the Nicomekl River. A number of livestock feedlots situated on the river or its tributary creeks contribute nutrient (phosphorus and nitrogen) loadings to the river. Domestic sewage discharging to septic tanks and tile fields within the region also introduces some contaminants.

The B.C. Ministry of Environment and Parks recently reviewed the water quality data determined for the Nicomekl River and its major tributary creeks during the period 1972 to 1986 (Holms and Swain 1987, Swain and Holms 1988). This survey included a number of sampling stations along the river from its headwaters to its mouth. For the metals monitored, lead, copper, zinc, chromium and iron exceeded working Provincial water quality criteria for the protection of aquatic life on a number of occasions. With the exception of lead and perhaps chromium, the high metal concentrations encountered were frequently elevated throughout the watershed and their elevation attributed to natural causes (Holms and Swain 1987). Concentrations of ammonia in samples of river and creek water were consistently below the Provincial water quality criteria for maximum and average concentrations thought to be safe for freshwater life. On occasion, nitrite concentrations in the Nicomekl River exceeded B.C. criteria for maximum nitrite levels in receiving waters, safe for aquatic life. Near the mouth of the river, phosphorus concentrations were sufficiently high to cause concerns about heavy algal production. During the period May through November, dissolved oxygen levels in the river (particularly at the downstream sites monitored) were frequently depressed to values known to be harmful to aquatic life (i.e. 1 - 5 mg/L). Even for sites further upstream (e.g. at 168th Street), values as low as 5 mg/L were recorded, although minimum dissolved oxygen concentrations at these sites were consistently $\geq$70% saturation. The reviewers concluded that the largest impact on water quality within the Nicomekl River was likely from diffuse agricultural operations and perhaps urban stormwater runoff (Holms and Swain 1987, Swain and Holms 1988).

5.3 Water Uses

Provincial licences have been issued for an annual withdrawal of 2,272 dam^3 of water from the Nicomekl drainage area. Irrigation of farmland accounted for 98.5% of this licenced withdrawal (Holms and Swain 1987). Other registered uses include withdrawal for livestock watering, land improvement (construction of ponds), aquaculture and greenhouse operations, and drinking water supplies.

The Nicomekl River provides an important spawning and rearing habitat for a number of salmonid fish species including steelhead and cutthroat trout, coho and chum salmon. Significant populations of coho salmon spawn in the upper reaches of this river ($\geq$20 km upstream from Mud Bay) and in Anderson Creek and Murray Creek (Holms and Swain 1987). The estuarine waters of Boundary Bay (which receive the discharges from the Nicomekl and Surpentine Rivers) sustain crab and herring fisheries, are used as a migratory corridor by salmonid fish, and in the past have sustained an oyster fishery (Swain and Holms 1988).

5.4 Drainage and Irrigation of Adjacent Farmland

Within the vicinity south of Cloverdale (i.e. between 160th and 184th Streets), much of the arable lowland adjacent to the Nicomekl River is used by commercial vegetable and berry growers. A network of ditches drains this farmland (Figure 2), and water within the ditches is used by the local farmers for irrigation during the growing season as well as for drainage of runoff water. Much of the lowland has experienced problems with flooding in the winter, a high water table in the spring and fall, and a shortage of irrigation water during the summer. As a

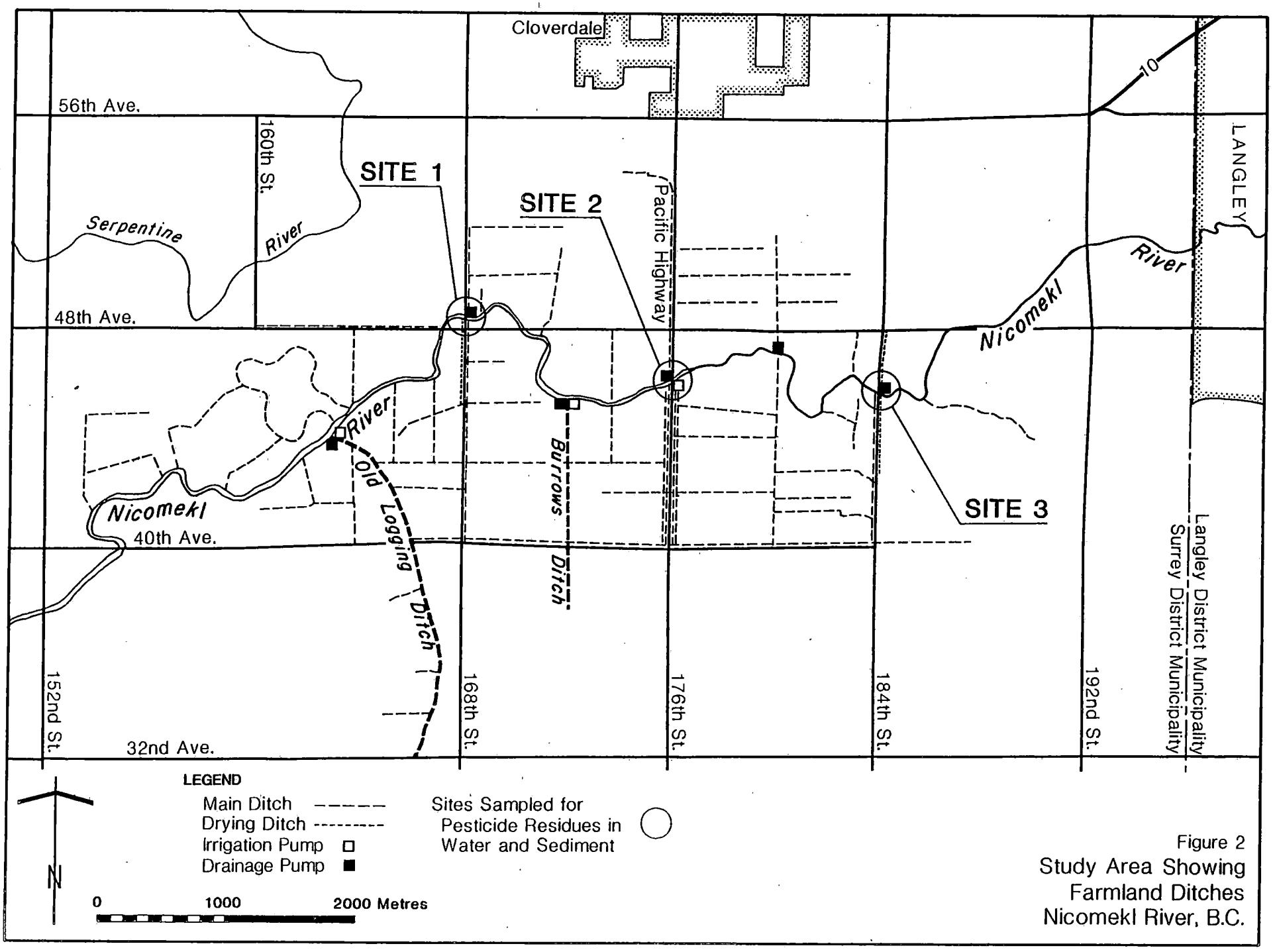

Figure 2
Study Area Showing
Farmland Ditches
Nicomekl River, B.C.

partial solution to these problems, Provincial and municipal engineers have installed a system of flood boxes and flap gates, drainage and irrigation pumps, at certain locations where the ditches reach the river (F. Smith and K. Wilson, personal commum.). In addition to the flood boxes and flap gates (which prevent back-low of riverwater into the ditches) at each point of entry of ditchwater into the river, drainage pumps are operational at the outlets of the Burrows Ditch, Old Logging Ditch, and certain ditches parallel to a number of the streets leading to the river (Figure 2). Irrigation pumps are located at the discharge points for the Burrows and Old Logging Ditches.

During periods of heavy runoff, drainage pumps are automatically activated by leveller devices set for a predetermined head of water in the ditch. The flap valves regulate the discharge of ditch water and prevent the backflow of riverwater into the ditches in instances where culvert pipes are submersed in the river. At sites where irrigation pumps are located (i.e. outlets of Burrows and Old Logging Ditches), these pumps are set for automatic operation during summer months (once the inflow from the land is no longer sufficient to meet the irrigation demand). The pumps are shut off during late summer or early fall when irrigation is no longer required. At this time or when seasonally-high fall rainfalls have commenced, stopgates to these irrigation/drainage ditches are removed allowing their drainage into the Nicomekl River.

6 SAMPLING AND ANALYSIS OF PESTICIDE RESIDUES

6.1 Sample Collection and Storage

On March 16, 1989, samples of water and sediment were taken from the Nicomekl River and farmland drainage ditches for analysis of

residues of dinoseb and endosulfan. Daily data for air temperature and precipitation recorded by Environment Canada at a nearby weather station (Surrey Municipal Hall) on this and preceding dates (starting March 1, 1988) are provided in Appendix IV. The general location of the four sampling sites is shown in Figures 1 and 2. Specific locations at which samples of both water and sediment were collected were as follows:

Site No.	Site Description
1	north side of river, 50 m above 168th Street
1	north side of river, 50 m below 168th Street
1	NE ditch at 168th Street, 10 m from outflow
1	SE ditch at 168th Street, 10 m from outflow
2	south side of river, 50 m above 176th Street
2	south side of river, 50 m below 176th Street
2	SE ditch at 176th Street, 10 m from outflow
2	2nd SE ditch at 176th St.[2], 10 m from outflow
2	NW ditch at 176th Street, 10 m from outflow
3	north side of river, 50 m below 184th Street
3	NE ditch at 184th Street, 10 m from outflow
4	river 10 m below 64th Ave (near 228th Street)

Water samples were taken by submersing a sample bottle to a depth of approximately 15 cm at each of the above locations. Each water sample was collected in a 4-L amber glass bottle which had been previously rinsed with acetone and heat-treated at 300 C. Concentrated sulphuric acid (2 mL; 50% concentration) was added to each sample bottle before field sampling to minimize adsorption of pesticides onto the glass surface. The lid for each bottle was lined with aluminum foil.

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[2]At 176th Street, two SE ditches (i.e. on the south side of the river and the east side of the street) run parallel to the road. The 1st and 2nd SE ditches were 6 m and 35 m (respect.) from the edge of the street.

Sediment samples were collected using a circular (15-cm diameter) steel trowel fitted to a wooden handle. Three sub-samples of the surficial 5 cm of sediment were taken at each sampling location. These sub-samples were used to fill a new 500-mL wide-neck clear glass bottle. Each bottle was sealed using caps lined with aluminum foil.

The water and sediment samples were transported to the laboratory. Upon arrival (within 24 h of collection), the sediment samples were transferred to a freezer (-20 C) for storage. Water samples were held at 5 C until solvent-extracted in preparation for analysis of pesticide residues. All samples were extracted within 24 h following their delivery to the laboratory.

6.2 Analytical Procedures

6.2.1 Water

Analyses for dinoseb and endosulfan (isomers I & II) were carried out in a single procedure which was based on established methods used by the B.C. Ministry of Environment and Parks' Environmental Laboratory (Anon. 1988). The target pesticides were extracted into an organic solvent (methylene chloride), purified by Florisil column chromatography, and analysed by electron capture gas chromatography (GC).

Tests for quality assurance/quality control (QA/QC) were performed by spiking blanks and an aliquot of one of the water samples with standards containing known quantitities of each pesticide and determining recovery efficiencies. Additional

QA/QC tests were conducted to determine pesticide recoveries from acidified versus unacidified water samples, and to measure the recovery of these pesticides from water if spiked and extracted immediately or after 24 h (samples stored under conditions identical to those for storing the test waters). Details concerning analytical procedures used for method validation and for the analyses of test waters are provided in Appendix III.

6.2.2 Sediment

The frozen sediment samples were allowed to warm slowly to ambient (20 C) temperature. Each sample (composite of three sub-samples) was mixed thoroughly before aliquots were taken for analysis. General procedures for sediment extraction and analysis for dinoseb and endosulfan residues were according to those practiced by the B.C. Ministry of Environment and Parks' Environmental Laboratory (Anon. 1988). Pesticide residues were extracted using both acetone and methylene chloride as organic solvents. Extracts were subsequently purified by Florisil column chromatography and analysed by electron capture gas chromatography. Details regarding the analytical procedure (including method validation) are given in Appendix III.

6.3 Analytical Results

The GC tracings for the water and sediment samples analysed are provided in Appendix III. Also provided in this appendix are the concentrations of dinoseb and endosulfans I and II determined to be present in these samples, together with the findings of the analyses for QA/QC and a description of possible sources of error associated with the analyses. Values determined for sediment samples are expressed in Appendix III on a wet-weight basis; the % moisture content of each sediment sample is also provided.

The concentrations of dinoseb and endosulfan found in the water and sediment samples collected from farmland ditches and the Nicomekl River on March 16, 1989 are summarized in Table 6. Endosulfan concentrations represent the sum of isomers I and II. Levels of dinoseb and endosulfan detected in each sediment sample represent dry-weight values.

For each of the ditchwater and riverwater samples analysed, concentrations of dinoseb were below trace levels (i.e. not detectable). With the exception of the samples of riverbed sediment collected at 168th Street in which no dinoseb was detected, this herbicide was found to be present in each sediment sample at concentrations ranging from trace quantitities (<10 ug/kg) to 49 ug/kg. For five of the six samples of sediment taken from the Nicomekl River, dinoseb concentrations were undetectable or present in trace amounts only (Table 6). One sample of riverbed sediment, taken 50 m above 176th Street, showed a dinoseb concentration of 37 ug/kg. The highest concentrations of dinoseb detected in sediment samples (36 ug/kg for 2nd SE ditch, 49 ug/kg for NW ditch) were found in sediments collected from ditches adjacent to 176th Street. The single ditch sediment sample taken at 184th Street contained 29 ug/kg dinoseb, whereas ditch sediments adjacent to 168th Street showed trace levels of this herbicide.

Except for a single sample of ditch water (taken from the 2nd SE ditch alongside 176th Street) which indicated a trace quantity of endosulfan, this insecticide was not detectable in any of the riverwater or ditchwater samples analysed (Table 6). Additionally, endosulfan was undetectable in any of the sediment samples collected from the Nicomekl River. For the ditch sediments analysed, this pesticide was not detected or present only in trace quantities. An exception to the above was the sample of sediment taken from the 2nd SE ditch at 176th Street, which showed a very high (428 ug/kg) concentration of endosulfan.

TABLE 6. CONCENTRATIONS OF DINOSEB[1] AND ENDOSULFAN[2] MEASURED IN MARCH 16, 1989 SURFACE WATER AND SEDIMENT SAMPLES.

Site No.	Source of Sample	Description of Sampling Location	Dinoseb		Endosulfan	
			water (ug/L[3])	sediment (ug/kg[4])	water (ug/L[3])	sediment (ug/kg[4])
1	NE ditch	168th Street, 10 m from outflow	-[5]	TR[6]	-	-
1	SE ditch	168th Street, 10 m from outflow	-	TR	-	TR
1	river	50 m above 168th Street	-	-	-	-
1	river	50 m below 168th Street	-	-	-	-
2	NW ditch	176th Street, 10 m from outflow	-	49	-	TR
2	SE ditch	176th Street, 10 m from outflow	-	TR	-	TR
2	2nd SE ditch	176th Street, 10 m from outflow	-	36	TR	428
2	river	50 m above 176th Street	-	37	-	-
2	river	50 m below 176th Street	-	TR	-	-
3	NE ditch	184th Street, 10 m from outflow	-	29	-	-
3	river	50 m below 184th Street	-	TR	-	-
4	river	10 m below 64th Avenue	-	TR	-	-

[1]Detection limits, 0.1 ug/L in water, 10 ug/kg in sediment.
[2]Endosulfan I + endosulfan II. Detection limits, 0.1 ug/L in water, 10 ug/kg in sediment.
[3]Micrograms per liter (parts per billion, ppb).
[4]Micrograms per kilogram (parts per billion, ppb), dry weight basis.
[5]Not detected.
[6]Trace amount, i.e. may be present but at a concentration below the detection limit.

Separate analyses for recovery of pesticides from spiked water samples showed mean recovery efficiencies of 84 to 103%. Mean recovery efficiencies for spiked sediment ranged from 88% (for endosulfan I) to 132% (for dinoseb) (Appendix III). Tests for recovery of dinoseb or endosulfan (isomers I and II) from deionized water spiked with these pesticides and extracted immediately or 24 h thereafter indicated no differences in recovery (i.e. loss of pesticide) due to this period of storage prior to sample extraction.

7 APPRAISAL OF PESTICIDE RESIDUE DATA

The values determined in the current tests for percentage recovery of dinoseb and endosulfan I and II from spiked water and sediment samples (mean values ranging from 83 to 132% recovery) indicate that some improvement in analytical techniques might be warranted. The analysts identified a number of possible sources of error that may have contributed to the loss in precision and accuracy observed, and recommended the development of modified procedures to improve the reproducibility and accuracy of the analytical results (Appendix III). Environment Canada investigators reporting the levels of selected pesticides in samples of water and sediment from B.C. farmland ditches found mean ($\pm$ SE) recovery efficiencies as follows: dinoseb, 70 $\pm$ 10% for water, 39 $\pm$ 7% for sediment; endosulfan, 84 $\pm$ 7% for water, 104 $\pm$ 6% for sediment (Wan 1989). Overall, these recovery efficiencies are no better than those evidenced in the present study. Analytical equipment (Hewlett Packard 5880 gas liquid chromatograph equipped with electron capture detector) and procedures used for the Environment Canada analyses differed somewhat from those employed in this study (Wan 1989).

The trace quantities of dinoseb and endosulfan detected in certain water and/or sediment samples collected from the Nicomekl River or adjacent farmland ditches (Table 6) should be considered with caution. Since values indicated as "trace" quantities were below the acceptable limit of detection for these pesticides, and since no confirmatory analyses were conducted, they may represent small peaks on the GC-ECD chromatograms caused by other organic material with similar properties. The water and sediment samples analysed showed an appreciable array of peaks caused by organic compounds besides the pesticides under investigation (see chromatograms in Appendix III). More detailed chemical analyses would be required to identify these compounds and to determine if they were from natural or anthropogenic sources.

The current findings of undetectable concentrations of dinoseb in ditch and river water samples, and undetectable or trace (one sample only) levels of endosulfan in the same samples (Table 6) are not surprising given the date that the samples were collected (i.e. mid-March sampling preceding seasonal applications of crop pesticides). Wan (1989) found detectable (4 - 5 ug/L) levels of dinoseb in water sampled from three of five Lower Mainland (B.C.) farmland ditches during February, although ditch water sampled from the same locations during May (before the spraying season) showed no detectable concentrations of dinoseb. Higher (up to 19 ug/L) dinoseb concentrations were evident in certain ditch-water samples collected during October and December (following fall spraying). Wan (1989) was also unable to detect endosulfan in water sampled in February, May, July, October or December from agricultural ditches draining cropland except for a sample taken shortly (0.5 h) after spraying. The absence of endosulfan in surface waters (except shortly after its application) is consistent with the previous conclusion by Canadian scientists that detectable amounts of this insecticide generally are not found in surface water runoff, two weeks (or longer) after a

single application (Anon. 1975). However, this may not be universally true. For instance, a survey of pesticides in eleven agricultural watersheds in Ontario during the mid-1970's indicated that low concentrations of endosulfan (overall mean, 0.003 ug/L; maximum mean, 0.016 ug/L); could be found in 20% of the water runoff samples collected throughout the year (Frank et al. 1982).

In this study, detectable (i.e. above trace amounts; 29 - 49 ug/kg) concentrations of dinoseb were present in three of the five sediment samples collected at site #2 (176th Street) and in the sediment taken from a ditch at 184th Street. A high (428 ug/kg) concentration of endosulfan was present in one sample of ditch sediment from site #2 whereas this pesticide was below the limit of detection (<10 ug/kg) in all other sediment samples collected March 16, 1989. Wan (1989) found a mean level of 81 ug/kg dinoseb in ditch sediments collected during October 1987, December 1987 and February 1988 from the south side of the Nicomekl River at 168th Street, although dinoseb levels in ditch sediments sampled at this site during May and July were below the limit of detection. At a separate site, Wan (1989) found significant (>100 ug/kg) amounts of dinoseb in ditch sediment during October and December, although subsequent post-season samples taken in February of the following year showed levels to be lowered to the limit of detection.

Unlike the present findings, high concentrations of endosulfan (mean, 652 $\pm$ 126 ug/kg; range, 334 - 926) were found in all (May, July, October, December 1987; February 1988) sediment samples taken from 168th Street ditches on the south side of the Nicomekl River by Wan (1989). These (Wan 1989) data provided evidence that endosulfan was relatively persistent in ditch sediments and that this pesticide could survive the period from post-spray

through to next year's application. However, at a separate site, Wan (1989) found that a seasonally-high (465 ug/kg; October value) concentration of endosulfan in agricultural ditch sediment declined to very low levels (<10 ug/kg) by February of the following year. Accordingly, the data indicate that endosulfan can be very persistent in aquatic sediments receiving runoff from cropland where this pesticide is applied. However, as for dinoseb, the degree of pesticide persistence is dependent upon site characteristics, use patterns and climatic conditions. Thus the persistence of pesticide in soil and sediment can vary appreciably from year to year even for the same site. The ability of endosulfan (and its metabolites) to persist in sediment has been reported previously (Anon. 1975).

A number of water quality criteria and objectives have been established for endosulfan. Researchers at Oregon State University proposed 0.003 ug/L as the maximum concentration of dissolved endosulfan in rivers, and 0.03 ug/L as the maximum concentration in feeder streams (Newton 1977). The International Joint Commission (IJC) gives 0.003 ug/L as a maximum objective for lake and stream water entering the Great Lakes (Frank et al. 1982). To protect freshwater aquatic life, the U.S. Environmental Protection Agency (EPA 1980) has derived water quality criteria for endosulfan of 0.056 ug/L as a 24-h average, and 0.22 ug/L as a concentration that should not be exceeded at any time. Inasmuch as endosulfan concentrations as low as 0.3 ug/L have been shown to be acutely lethal to fingerling salmonid fish under laboratory conditions (Table 4), and that acute sublethal toxic effects toward fish can occur at 0.1 ug/L (Tables 3 and 5), the EPA criteria are too lenient. On the other hand, the criteria put forward by Newton (1977) and IJC (Frank et al. 1982) offer an appreciable margin of safety for the protection of the more sensitive species and life stages of freshwater life. This statement should be tempered by our present lack of data which demonstrate Lowest Observed Effect Concentrations of

endosulfan causing acute or chronic sublethal toxic effects toward salmonid fish species.

It is noteworthy that the limit of detection of endosulfan in water was 0.1 ug/L in the present study and 1 ug/L in the study by Wan (1989). Neither limit allows an assessment as to whether the levels of endosulfan in ditch water ("feeder streams") or the Nicomekl River meet or exceed the water quality criteria of Newton (1977) or IJC for this pesticide. However, a limit of detection of 0.1 ug/L does allow a determination of whether concentrations of endosulfan in water samples are higher than the lowest values shown in laboratory studies to cause sublethal or lethal toxic effects toward salmonid fish or other sensitive aquatic life (Table 3).

The toxicity of dinoseb to salmonid fish species has been shown to be highly dependent on the pH of the dilution (receiving) water, with appreciably greater toxicity if mixed with slightly acidic or neutral (pH 6.5 - 7.0) waters, relative to alkaline (e.g. pH 8.5) waters (section 4.1.1). Seasonal water quality data determined for the Nicomekl River at 168th Street indicate a mean pH of 7.3, with values ranging from 6.4 to 8.4 (Holms and Swain 1987). Thus conditions in this receiving water could exist where the toxicity of dinoseb to salmonid fish could be as great as that found in tests with non-alkaline waters (Tables 1 - 3). Although detectable levels of dinoseb were not found in any of the ditch or riverwater samples analysed in this study, Wan (1989) did find detectable levels of this pesticide in certain cropland ditches which remained elevated (3 - 19 ug/L) throughout the post-spray months of October through February. Consideration of these data together with the summary toxicity data for this pesticide (Tables 1 - 3) indicates that such levels of dinoseb would be unlikely to cause acute lethal or sublethal toxic effects within receiving streams or rivers. However, concerns for chronic toxic effects may be justified if the dilution of

dinoseb within the receiving water were minimal and sensitive life stages of salmonid fish were exposed to this pesticide for a prolonged period.

No existing water quality criteria for dinoseb were identified during the present review of available literature. Using the analytical procedures practiced in this study and for water samples collected by Wan (1989), the limit of detection for dinoseb in water is 0.1 ug/L. This detection limit is adequate to identify concentrations of this pesticide in water which exceed the lowest levels presently known to adversely affect salmonid fish or other sensitive freshwater life (Tables 1 - 3).

Existing water quality data for the Nicomekl River show that wide fluctuations in dissolved oxygen (DO) content can occur seasonally in this waterbody. For instance, seasonal DO values measured at 168th Street have ranged from 5.4 to 16.5 mg/L (ie. 186% to 70% saturation). Very low DO levels (1 - 2 mg/L) have been noted frequently in the lower reaches of this river during the fall when flows are low (river is dammed near its mouth). Riverwater temperature at 168th Street can reach as high as 23 C during warm summer periods when flow is minimal (Holms and Swain 1987, Swain and Holms 1988).

Conditions of high temperature (e.g. $\geq$20 C) and low dissolved oxygen (e.g. 5 - 7.5 mg/L) are well known to be stressful to salmonid fish, and can reduce their adaptive capabilities and chance for survival. The contribution of stormwater runoff, landfill leachate and other diverse aquatic contaminants to this river (section 5.2) places an additional stress loading on resident or migrant fish and other aquatic life. The tolerance of these organisms to agricultural-use pesticides entering the river in surface-water runoff may be reduced significantly due to their simultaneous (or previous) exposure to other aquatic contaminants or natural stressors.

The combined interaction of multiple stressors could result in deleterious effects of endosulfan and dinoseb (and other agricutural pesticides) at levels below those indicated from laboratory studies (where water quality conditions are otherwise optimal). On the other hand, the presence of other materials in this receiving water (e.g. humic/fulvic acids and other dissolved organics; fine suspensions of clay and silt) could reduce the biological availability of pesticides from that indicated by the laboratory studies (where such chelating/adsorbing substances are absent or minimally present). The toxicity data base for the pesticides under investigation should therefore be used only as an initial guideline of possible adverse impacts. Site-related biological and chemical studies are necessary if predictions of the presence or absence of significant toxic impact are to be made with confidence.

The environmental relevance of the concentrations of dinoseb and endosulfan detected in this study for cropland ditches discharging to the Nicomekl River, or in other B.C. agricultural ditches (Wan 1989), is not known. These sediments act as a "sink" for pesticides leached from vegetation and soil and carried into the drainage ditches together with stormwater runoff. However, information is presently lacking regarding the biological availability of sediment-bound pesticides or their ability to recycle into overlying ditch water and receiving waters in a state that is potentially toxic to exposed aquatic life.

8 RECOMMENDED APPROACH FOR SITE-SPECIFIC TOXICITY ASSESSMENTS

8.1 General Approach

Data requirements for environmental toxicity studies associated with the registration of crop-protection pesticides are listed in Agriculture Canada's memorandum T-1-237 (Agriculture Canada 1983). This memorandum specifies requirements for routine acute lethal toxicity studies, usually conducted using two fish species and an aquatic invertebrate (usually Daphnia magna). Special studies with aquatic organisms may also be required, including chronic-exposure laboratory tests and simulated or actual field trials. However, at present there are no detailed guidelines for undertaking environmental toxicity studies that are comparable to the existing guidelines for determining the environmental chemistry and fate of pesticides (Agriculture Canada 1987).

Reasons for the present absence of detailed guidelines for undertaking environmental toxicity studies associated with pesticide registration and use include the site-specific nature of studies that predict and monitor the toxic impact of pesticides on particular receiving-water bodies. Additionally, the lack of standardized aquatic toxicity tests (lethal and sublethal, acute and chronic) appropriate for the protection and monitoring of the Canadian aquatic environment (Sergy 1987) restricts the testing of receiving waters and samples thereof in a consistent and meaningful manner. This deficiency has been addressed by Environment Canada, and a 4-year program has been launched to develop and establish a suite of lethal and sublethal aquatic toxicity test methods suitable for national and regional needs (Scroggins 1989, Wells 1989). The test procedures being developed include those suitable for the collection, storage and testing of receiving waters and sediments.

The use of biological tests for water pollution assessment and control is now an accepted and popular approach both within Canada and elsewhere (OECD 1984, 1987). An expanding body of scientific evidence supports the concept that a battery of single-species toxicity tests (lethal and sublethal; acute and chronic), if thoughtfully selected, can adequately predict and monitor the impact of pesticides and other aquatic contaminants on sensitive aquatic life in nature (Hansen and Garton 1982, Adams et al. 1983, Chapman 1983, Mount et al. 1984, Wells 1988). This approach is now widely referred to as "ecotoxicology" (Blaise 1984) and, although frequently involving laboratory evaluations of field-collected samples, is intended to provide results that are relevant to real-world situations.

The design of an approach for testing runoff waters and receiving waters in order to make site-specific assessments of their degree of risk to indigenous aquatic life is complex, and no standardized "blueprint" is available. Selected ecotoxicological tests can and should be used as biological tools ("biomarkers") for assessing runoff waters or receiving waters and for predicting their likelihood of causing adverse biological impacts on exposed aquatic life. Chemical analyses of certain test waters must be conducted in conjunction with these tests if culprit pesticides and concentration-effect relationships are to be defined. Consideration should also be given to the sensitive aquatic species frequenting receiving waters, susceptible life stages at risk, and their likely pattern of exposure (acute or chronic). Ultimately, biological field surveys are required to confirm the integrity of aquatic communities (EPA 1989).

Recent review articles are available which describe the impact of pesticides on sensitive freshwater fish and invertebrate species (Murty 1986, Muirhead-Thomson 1987). Included in these publications are descriptions of various approaches and test

procedures used previously by researchers concerned with predicting and monitoring the toxic impact of pesticides reaching the freshwater environment. An ecotoxicological approach for evaluating the toxic risk to regional (B.C.) fisheries resources posed by forest-use herbicides has also been published (McLeay 1988) and is pertinent to present needs. These have been considered while deriving a test approach suitable for the present site-specific investigation.

Recommendations are made here for future toxicity assessments associated with the study area selected. The approach offered is conceived in consideration of the specific characteristics of the study area, our present understanding of seasonal pesticide use in the region, the nature of flows within the agricultural ditches draining pesticide-treated cropland, seasonal climatic conditions specific to the site of interest, accessibility of sampling sites, and costs associated with site-specific assessments requiring complex biological and chemical testing and analyses. A sequential (tiered) approach toward testing and evaluation of drainage waters and the receiving environment is presented to allow for decisions to be made regarding additional investigative studies to be undertaken, and to provide for flexibility in the overall approach.

Sampling to date has identified the presence and concentration of two "priority" agricultural-use pesticides (dinoseb and endosulfan) within certain cropland drainage/irrigation ditches and at a number of locations within the receiving-water body (Nicomekl River). The analytical values derived from these samples serve to indicate the past use of these pesticides within the study area and their continued presence and partitioning (sediment and water) in the runoff drainage and receiving waters, several months after application.

The next phase of activity to be undertaken within the study area as part of this project is intended to address the following questions:

1. Are surface water runoffs draining into the Nicomekl River from commercial cropland ditches toxic to aquatic life?

2. How does the toxicity of runoff water entering the river at specific locations vary seasonally?

3. What degree of dilution of cropland runoff water is required to prevent toxic effects from being observed?

4. Are the receiving waters within the study area periodically toxic to sensitive aquatic organisms? If so, is this toxicity due to the cropland runoff water?

4. Can pesticides (in particular, dinoseb and endosulfan) be identified as major contributors to toxicity observed within the farmland drainage ditches and receiving waters?

8.2 Sampling Stations

Sampling stations appropriate for biological testing of cropland runoff water and receiving waters should be chosen from within the study area (Figure 1). The following are proposed as candidate sampling stations:

- headwaters of Nicomekl River (site 4, Figure 1; reference station).
- river at 192nd St. (just downstream of Anderson Creek and the influence of stormwater runoff from Langley) (Fig. 2).

- ditch discharges at 184th St. (site 3, Figure 2).
- ditch discharges at "180th St." (between sites 2 and 3).
- ditch discharges at 176th St. (site 2, Figure 2).
- river below 176th St.
- discharge from Burrows Ditch.
- ditch discharges at 168th St.
- discharge from Old Logging Ditch.
- river below Old Logging Ditch.

This represents 19 - 20 candidate sampling stations, although accessibility to all is not assured. Site visits should be made to seek access to those sites on private land (Burrows Ditch, Old Logging Ditch, ditches at "180 Street") and to decide upon their suitability. Visits should also be made to a number of commercial vegetable farms adjacent to the proposed sampling stations to determine their pesticide uses and periods of application. Experience has indicated that the information obtained by these visits may be fragmented (Moody 1989); however this effort is warranted.

Budgetary and other considerations may restrict the number of stations sampled from the initial 19 - 20 stations proposed. A final selection of sampling stations will be necessary during the first intensive field collection (early fall, once seasonal rainfalls have commenced and flows to the river are established). The interest and cooperation of the Surrey Diking District should be sought as it is responsible for the regulation of flows draining and irrigating the farmland within the study area, and familiar with accessibility to the proposed sampling stations. Close contact with their control foreman (Mr. Gordon Bishop; phone no. 576-6438) is advised to obtain advance warning of planned conversion of the system from irrigation to drainage mode, information regarding historical patterns of flow, and any records of flow disruption or atypical flow events during the study.

8.3 Biological Tests

8.3.1 Tests for Acute Lethal and Sublethal Toxicity

Water samples should be taken from each of the designated sampling stations for determinations of their acute lethal and sublethal toxicity to salmonid fish. The proposed frequency of sampling and testing should include one occasion prior to the seasonal runoff period (August; riverwater samples only); two separate occasions shortly after the seasonal runoff of water from croplands to the river commences (e.g. mid-September, October) as well as two later post-spray occasions prior to cessation of these flows (e.g. December, March/April).

Each water sample should be transported to the laboratory for acute (96-h) survival tests using young (swimup fry or fingerling) rainbow trout. Testing should be commenced within 24 hours of sample collection. These tests should be conducted according to established practices (Environment Canada 1989). Samples should be aerated prior to the test if necessary, as well as during the test. Each sample should be tested undiluted (10 fish per solution). At least one control solution (laboratory water supply to which fish are acclimated) should be included with each series of tests.

In all instances where toxicity is evident (>10% mortality of test fish within 96 h and/or changes in fish behaviour or appearance relative to those in the control solution), those sites showing toxicity are to be immediately revisited and a sufficient sample volume collected to enable a 96-h LC50 (median lethal concentration) test to be performed. River water collected at 192nd Street should be used as the dilution water for these tests unless toxicity at this site is indicated. In this event, the laboratory water supply to which fish have been

acclimated should be used as the dilution and control water (Environment Canada 1989).

Fish in each test solution should be observed closely at approximately 1, 4, 24, 48, 72 and 96 hours after commencement of the test. Numbers of dead fish in each solution should be recorded and the fish removed. At these times, fish should also be carefully observed for obvious sublethal toxic responses to the test solution (e.g. erratic swimming behaviour, surfacing, respiratory distress, loss of equilibrium, discolouration). Any differences in behaviour or appearance from that of the control fish should be noted.

Test results should report % fish mortalities and any sublethal effects noted at each observation period. Where toxic responses were evident and LC50 tests were performed, the No-Observed-effect Concentration (i.e. the highest concentration of test water in which no difference in fish behaviour or appearance, relative to the control, is apparent at any time during the test) should be reported together with the 96-h LC50 (Environment Canada 1989).

Rapid (96-h) toxicity tests using freshwater algae should also be undertaken with aliquots of a number of the samples used in acute lethal/sublethal toxicity tests with rainbow trout. These tests are necessary to assess the relative sensitivity of algae to the test waters. Such testing is particularly relevant inasmuch as no data are presently available regarding the toxicity of dinoseb (a herbicide) to freshwater algae. Standardized procedures for measuring the inhibition or stimulation of growth of the freshwater algae Selenastrum capricornutum have been published (Sergy 1987) and are recommended here. Additional in-situ tests using this or other freshwater algal species may also be appropriate for inclusion in the study.

8.3.2 Tests for Chronic Toxicity

Based upon the seasonally-intermittent river discharge of pesticides in cropland runoff water, and on existing evidence for pesticide persistence in ditch water and/or sediment, the possibility exists that pesticides could be discharged to the Nicomekl River over an extended period once seasonal flows commence. This being the case, aquatic life frequenting the receiving water could receive chronic exposure to low (sublethal) levels of pesticides. Accordingly, tests for chronic toxic effects associated with cropland drainage are recommended as part of this project.

At the present time, the two most popular and accepted chronic aquatic toxicity tests are a 7-day life cycle test using the freshwater microcrustacean Ceriodaphnia sp., and a 7-day larval growth test using fathead minnows (EPA 1985, Sergy 1987). These tests are not inexpensive, and require the daily collection and delivery of fresh water from each test site for use in the 7-day bioassay. Thus their use must be restricted to a few well-chosen samples. It is proposed that samples of riverwater be tested for chronic toxicity using at least one of these test species and procedures and preferably both, funding permitting. The selection of sampling stations for these tests should be decided following a review of the findings of the initial series of acute lethal/sublethal toxicity tests with rainbow trout. Candidate river sampling stations are those indicated in section 8.2, although these may be modified (e.g. deletion of one or more stations, addition of new stations immediately above discharges shown to be acutely toxic and of appreciable flow). Discharges from certain cropland drainage ditches may also be examined, although priority should be given to testing receiving waters above and below these flows.

Sampling for chronic toxicity tests should be undertaken on at least two occasions. Appropriate test periods would be during the fall when discharges from drainage ditches have commenced in earnest, and again later in the post-spray season. For each of the sampling stations chosen for chronic toxicity evaluations, water samples should be taken daily for seven days. These samples are to be used for daily replacements of the testwater to which the organisms are exposed. To reduce costs, riverwater samples could be tested without dilution. Results indicating chronic toxic responses to test waters should be examined for significance using Dunnett's or other appropriate statistical procedures (EPA 1985).

8.3.3 Field Toxicity Tests

Aquatic toxicity tests performed within or adjacent to the receiving environment (e.g. controlled-exposure tests using caged fish and/or aquatic invertebrates; on-site tests using apparatus provided a continuous flow of river water) can measure toxic responses to receiving-water conditions unmodified by laboratory influence or change during transport and storage. For instance, captive organisms held at specific stations in the Nicomekl River above and below sites receiving runoff from cropland drainage ditches would be exposed to ambient conditions of temperature, pH and dissolved oxygen together with fluctuating levels of pesticides and other aquatic contaminants. These conditions provide a "window" of environmental influence during the test period and thus a realistic measure of biological effect. Drawbacks to in-situ or on-site (flow-through) toxicity tests are that they are labour-intensive, high-risk and, in many cases, cost-prohibitive, and that the apparatus used may stress the organisms unduly and produce spurious results. In spite of these

disadvantages, field toxicity tests often prove worthwhile and provide greater confidence in the conclusions drawn from laboratory tests with field-collected samples.

Field toxicity tests are warranted as part of this project. However, due to perceived budgetary constraints, such studies would best be delayed until the first year of biological testing is completed and the results evaluated. Using this approach, sites perceived as particular regions of concern (based upon findings of the laboratory toxicity tests and associated chemical analyses) could be focused on for subsequent (1990/91) field evaluation. Field tests might also be coordinated to coincide with field surveys of distribution and abundance of indigenous species including salmonid fish and aquatic invertebrates.

8.4 Chemical Analyses

The chemical analysis of test waters is prerequisite to understanding the probable causes of the toxic responses observed. For each of the environmental samples (ditch water, river water) evaluated in laboratory toxicity tests, water quality characteristics including pH, hardness, specific conductivity, suspended solids/turbidity and metal concentrations (preferably dissolved) should be measured, resources permitting. Appropriate metals for analysis include copper, iron, lead, zinc and chromium (Holms and Swain 1987). Analyses might also include un-ionized ammonia and nitrite concentrations, particulate and dissolved organic nitrogen, and total and dissolved phosphorus. Temperature, pH and dissolved oxygen content in each test water should be monitored during the toxicity tests and the results reported. Measurement of specific conductance of test solutions is also advised.

Test waters demonstrated to be toxic should be analysed for concentrations of dinoseb and endosulfan. Analyses for endosulfan should include the toxic metabolite endosulfan sulfate together with isomers I and II (Wan 1989). The limit of detection for dinoseb and endosulfan should be 0.1 ug/L or lower. Water samples should be filtered before extraction and analysis. Sub-samples taken for pesticide analyses should be extracted within 24 h of their collection. Other pesticides (e.g. glyphosate) might be included in certain analyses if warranted and cost-effective. Alternatively, chemical screening tests to detect the presence and concentration of extractable chlorinated organic compounds could be considered.

Summary toxicity data should be reviewed and assessed with respect to the chemical analytical data. For water samples for which sufficient LC50 and NOEC values were derived from multiple-concentration toxicity tests, correlation coefficients should be determined for these toxicity data versus respective pesticide and other chemical measurements.

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APPENDIX I. List of Contacts

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APPENDIX II. List of Databases Searched.

A. Databases Searched

CA SEARCH (1976-1989)
BIOSIS PREVIEWS (1969-1989)
AQUACULTURE (1970-1984)
WATERNET (1971-1989)
POLLUTION ABSTR. (1970-1989)
ANALYT. ABSTR. ONLINE (1980-89)
LIFE SCI. COLL. (1978-1989)
CHEMICAL EXPOSURE (1974-1987)
AQUATIC SCI. ABST. (1979-1989)
WATER RESOUR. ABST. (1968-1989)
ENVIROLINE (1970-1989)
ENVIRON. BIBLIOG. (1974-1988)
EMBASE (1974-1989)

B. Key Words Searched

Toxicity to Freshwater Biota

dinoseb or endosulfan
aquatic toxicity
freshwater biota or life or organisms
fish or invertebrates or algae
lethal or sublethal
acute or chronic

Bioaccumulation and Clearance

dinoseb or endosulfan
freshwater life or freshwater organisms
fish or invertebrates or algae
bioaccumulation or bioconcentration
or biomagnification or uptake
clearance or depuration

Aquatic Fate and Persistence

dinoseb or endosulfan
environmental or aquatic or water
or freshwater or sediment
fate or persistence or degradation or conversion or transformation or adsorption or dissipation or volatilization or hydrolysis or loss

APPENDIX III. Analysis of Dinoseb and Endosulfan in Samples of Sediment and Water Collected From the Nicomekl River and Farmland Ditches on March 16, 1989.

(see the following 34-page Analytical Report by B.C. Research Corporation dated April 1989).

ANALYSIS OF DINOSEB AND ENDOSULFAN IN SAMPLES OF SEDIMENT AND WATER COLLECTED FROM THE NICOMEKL RIVER AND FARMLAND DITCHES ON MARCH 16, 1989

Prepared for:

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April, 1989
BCR File No: 4-03-489

SUMMARY

The concentrations of dinoseb and endosulfans I & II were determined in twelve sediment and surface water samples collected from the Nicomekl River, B.C. or adjoining farmland ditches. None of the water samples were found to contain dinoseb and endosulfans I & II at levels above the detection limit of 0.1 parts per billion (ppb). However, four sediment samples contained pesticides at levels ranging between 14 and 139 ppb (wet weight basis):

Site #2 - Upstream River, 176th St.:	19 ppb dinoseb
Site #2 - 2nd SE Ditch, 176th St.:	14 ppb dinoseb
	28 ppb endosulfan I
	139 ppb endosulfan II
Site #2 - NW Ditch, 176th St.:	25 ppb dinoseb
Site #3 - NE Ditch, 184th St.:	14 ppb dinoseb

Concentrations of dinoseb and endosulfans I and II were below detection limits in the remaining sediment samples.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF FIGURES (Continued)

1.0 BACKGROUND

The analytical results reported in this study were performed in support of a study by Coastline Environmental Services Ltd. to assess the toxic effects of in-use pesticides on aquatic biota in the Nicomekl River, B.C.. British Columbia Research Corporation was sub-contracted to provide chemical analyses of dinoseb and endosulfans I & II in sediment and water sampled for this study.

2.0 OBJECTIVE

1. To determine the concentrations of dinoseb and endosulfans I and II in sediment and surface water sampled from farmland ditches and the river at four discrete sites.

2. To establish confidence limits (% recovery efficiencies) for the methods used for analysing dinoseb and endosulfans I and II in sediment and water.

3.0 EXPERIMENTAL

3.1 ANALYSIS OF WATER SAMPLES

3.1.1 Analytical Procedures

Water samples were extracted within 24 hours of delivery to B.C. Research. Each sample bottle was shaken vigorously to homogenize the water sample before aliquots were taken for analysis.

An 800 mL aliquot of unfiltered, acidified water was extracted three times with methylene chloride (80, 50, and 50 mL, respectively). The methylene chloride extract was filtered through acidified anhydrous sodium sulphate, 3 mL of iso-octane was added, then concentrated to approximately 3 mL with a rotary evaporator under reduced pressure. The concentrate was diluted with petroleum ether (20 mL), re-evaporated to 5 mL to remove residual methylene chloride, then diluted to 20 mL with petroleum ether. Diazomethane, generated by reaction of N-nitrosomethylurea with sodium hydroxide solution, was bubbled through the extract with a gentle stream of nitrogen until a yellow colour persisted.

The solution was rotary evaporated to 5 mL and applied to the top of a glass column (9 mm I.D.) containing 2% deactivated Florisil (5 g) topped with 2 cm of acidified anhydrous sodium sulphate. Petroleum ether (50 mL) was first run through the column to remove nonpolar co-extractives. Dinoseb and endosulfans I & II were eluted in a second fraction with 50 mL of petroleum ether containing 10% ethyl acetate. This fraction was concentrated to approximately 2 mL with a rotary evaporator then made up to 4.0 mL with iso-octane. It was analysed by gas chromatography using electron capture detection.

3.1.2 Gas Chromatographic Analysis

A Hewlett-Packard 5890 gas chromatograph equipped with an electron capture detector, HP-5 (30m x 0.53 mm I.D. x 0.88 m film thickness) megabore column, split/splitless and cool on-column injection ports, and automatic sampler was used to analyse dinoseb and endosulfans I and II in water samples.

Purified extracts were injected by splitless or cool on-column mode using the following conditions to analyse for dinoseb and endosulfans I & II:

Carrier Gas:	Helium	
Oven Temperature:	Initial	80 C, hold 1 min
	Rate	20 C/min to 200 C
		10 C/min to 275 C, hold 2.5 min
Column Head Pressure:	95 kPa	

Selected extracts were examined by cool on-column injection of extracts onto an HP-5 (30m x 0.25 mm I.D. x 0.25 um film thickness) or DB-17 (30m x 0.32 mm I.D. x 0.25 um film thickness) capillary column using the same GC separation conditions.

3.1.3 Method Validation

Recovery of dinoseb and endosulfans I & II from acidified water samples was checked by comparing pesticide recoveries from acidified and unacidified samples. Aliquots of deionized water (800 mL) acidified with 0.5 mL of 50% sulphuric acid were spiked with 0.40 mL of a standard solution containing 1.0 ppm dinoseb and endosulfans I & II. The spiked solutions were extracted immediately after spiking and after storage at 5 C for 24 hours. Unacidified water samples were also spiked with the pesticides, extracted and analysed by the same procedures as were the acidified samples. Water samples were analysed immediately and after 24 hours storage.

The validity of analytical procedures used to analyse water samples was evaluated in triplicate by spiking an aliquot of a test water (sample from site #3 - NE ditch) with 0.4 mL of a standard mixture containing 1.0 ppm of each of the target pesticides. Recovery efficiency of dinoseb and endosulfans I and II was determined by comparison with a control mixture containing 0.1 ppm of each of the target pesticides.

All reagents and solvents used for the analyses were obtained from the same batch or reagent bottles to minimise variations in background impurities. Background interferences from solvents and reagents were monitored with blank samples which were treated identically to analytical samples.

3.2 ANALYSIS OF SEDIMENT SAMPLES

3.2.1 Analytical Procedures

Twenty grams of each sediment sample were sonicated twice for 15 minutes with 75 mL acetone/0.5% conc. sulphuric acid. The sonicated sediment was then rinsed twice with 25 mL acidified acetone.

The acetone solution was transferred to a 1 L separatory funnel and extracted with a mixture of 150 mL methylene chloride and 50 mL water. Water was used to partition the acetone into the lower methylene chloride layer. The upper aqueous layer was extracted twice with 50 mL of methylene chloride. The organic extract was filtered through acidified anhydrous sodium sulphate, 3 mL of iso-octane added, then concentrated to about 10 mL under reduced pressure with a rotary evaporator. The sample was split into two equal portions, one of which was reserved for repeat analyses. An organic deposit was usually present in the solution at this point. The precipitate in one portion of the extract was redissolved with up to 5 mL acetone. The solution was methylated three times with diazomethane and excess diazomethane was removed under a gentle stream of nitrogen.

To re-dissolve any precipitated material, iso-octane (3 mL) was added and the solution re-evaporated to 2-3 mL, diluted with 20 mL petroleum ether/10% ethyl acetate, then re-evaporated to 2-3 mL. The solution, usually containing particulate deposits, was shaken with 2 mL ethyl acetate to dissolve polar materials, then 18 mL petroleum ether was added.

The solution and deposits were applied to the top of a glass column (9 mm I.D.) containing 2% deactivated Florisil (5 g) topped with glass wool and 2 cm of acidified anhydrous sodium sulphate. Dinoseb and endosulfans I & II were eluted from the column with 50 mL petroleum ether/10% ethyl acetate. The eluate was concentrated to approximately 2 mL with a rotary evaporator and the volume made up to 4.0 mL with iso-octane. The solution was analysed by electron capture gas chromatography (GC-ECD).

3.2.2 Gas Chromatographic Analysis

A Hewlett-Packard 5890 gas chromatograph, equipped with an electron capture detector, cool on-column injection port, automatic sampler, and DB-5 (30 m x 0.25 mm I.D. x 0.25 um film thickness) capillary column, was used to measure dinoseb and endosulfans I & II in sediment extracts.

Instrument conditions were:

Carrier Gas:	Helium	
Oven Temperature:	Initial	80 C, hold 1 min
	Rate	20 C/min to 200 C
		10 C/min to 275 C, hold 2.5 min
Column Head Pressure:	20 kPa	

3.2.3 Method Validation

The analytical method was tested by spiking three 20 gram aliquots of sediment sampled from site #4 (reference site at the headwaters of the Nicomekl River) with 0.4 mL of a standard mixture containing 1.0 ppm each of dinoseb and endosulfans I & II, and measuring recovery of the pesticides. Analyses were carried out using the same procedures as those used with aliquots of the unspiked sediment. Background impurities from reagents and solvents were monitored with blank samples which were treated identically to sediment samples.

The linearity of GC response to dinoseb methyl ether and endosulfans I & II was checked with 0.02, 0.05, and 0.10 ppm standards.

3.3 MOISTURE MEASUREMENT

Twenty gram aliquots of each sediment sample were weighed into aluminum dishes and dried for 24 hours in an oven at 110 C. Percentage moisture and dry solids contents of each sample were determined. Pesticide residue levels determined were adjusted to dry-weight values by dividing the residue levels measured on a wet-weight basis by the fraction of dry solids (%/100) in the sediment sample.

4.0 RESULTS AND DISCUSSION

4.1 WATER SAMPLES

The analytical results listed in Table 1A show that dinoseb and endosulfans I and II were not present in any of the river or ditch water samples at levels above the detection limit of 0.1 parts per billion.

Representative gas chromatograms of extracts from sites #1, #2 and #4 obtained using cool on-column injection of the analyte onto an HP-5 capillary column are shown in Figures 1A-3A. The retention times for dinoseb methyl ether, endosulfan I and endosulfan II were 11.15, 13.88 and 14.97 minutes, respectively (Figure 4A). The absence of peaks in Figures 1A and 2A with these retention times demonstrates that dinoseb and endosulfans I and II were not present in the samples analysed. Other water samples produced very similar chromatograms to Figures 1A-3A.

A small peak with the same retention time as endosulfan II is evident in Figure 3A (sample from site #2 - 2nd SE ditch). This peak suggests the presence of a trace (<0.1 ppb) concentration of endosulfan II in this sample.

The peak with a retention time of 11.121 minutes shown in Figure 3A is close to dinoseb when chromatographed on an HP-5 megabore column. However, it was shown not to be dinoseb when the sample was re-examined by gas chromatography on a more polar DB-17 capillary column. Using this column, the unknown organic causing this peak had a retention time of 10.28 minutes compared with 10.12 minutes for dinoseb methyl ether. It was concluded that the peak did not represent dinoseb.

4.2 SEDIMENT SAMPLES

Twelve sediments obtained from the same locations as river and ditch water samples were analysed to determine the concentrations of dinoseb and endosulfans I and II. The analytical results for dinoseb and endosulfans I and II in sediment samples, calculated on a dry-weight basis, are listed in Table 1A. Dry solids contents of the sediments are given in Table 2A.

The target pesticides were found in four sediments at levels above the detection limit of 10 ppb. The sampling locations and the pesticide concentrations (dry-weight basis) found are summarized below:

Site #2 - NW Ditch, 176th St.:	49 ppb dinoseb
Site #2 - 2nd SE Ditch, 176th St.:	36 ppb dinoseb
	72 ppb endosulfan I
	356 ppb endosulfan II
Site #2 - Upriver, 176th St.:	37 ppb dinoseb
Site #3 - NE Ditch, 184th St.:	29 ppb dinoseb

The highest pesticide levels were found at site #2 (2nd SE ditch, 176th St.). Interestingly, this location produced the only water sample with a trace of endosulfan II. The sediment sample from this location also contained the greatest amount of organic deposits in the extract of any of the samples analysed.

Gas chromatograms of extracts from the samples of sediment collected at twelve sampling locations are shown in Figures 5A-16A. The retention times for dinoseb methyl ether, endosulfan I and endosulfan II for this series of samples were 11.22, 13.96, and 15.06 minutes, respectively (Figure 17A). All peak intensities are drawn to the same scale except for Figure 12A (sample taken at site #2 from the SE ditch), which is four times higher.

With the exception of the upstream riverwater sample collected at site #1, all other sediment samples appeared to contain trace levels of pesticides at levels below 10 ppb. Sediments collected from site #2 were more contaminated with the target pesticides than samples from all other sites.

4.3 QUALITY CONTROL

No difference was found in recoveries of acidified solutions when extracted immediately or after 24 hours. Similarly, no difference in recovery of either pesticide was found for acidified versus non-acidified water samples spiked with pesticides and extracted after storage for 24 hours.

Mean recovery efficiencies exceeded 80% for the three pesticides. However, the standard errors of the mean values (n = 3) were relatively large (Table 3A).

Reagents and solvents used in the analyses were confirmed to be free of interfering compounds with similar GC retention times to dinoseb and endosulfans I & II (Figure 18A).

4.4 SOURCES OF ERROR

1. **Co-extractives** - Possible trapping of target pesticides in deposits which formed when sediment extracts were concentrated and the solvent replaced by a less polar medium may result in incomplete recovery of pesticides. Deposits of humic material in the extracts of many sediment samples complicated the methylation and Florisil column chromatography. It was difficult to obtain a clear solution in these steps of the procedure although strenuous efforts were made to redissolve the deposits.

2. **Methylation of Dinoseb** - The presence of suspended solids, other acidic materials, and orange or yellow colours in sediment extracts, may result in incomplete methylation of dinoseb or masking of the methylation endpoint. Fatty acids in relatively high concentrations were detected in a sediment extract (site #4 - reference site at headwaters of river) analysed by gas chromatography-mass spectrometry, including tetradecanoic acid, hexadecanoic acid, octadecanoic acid, octadecenoic acid, and tetracosanoic acid. They were probably present in all of the sediment extracts to varying degrees. Fatty acids will

compete with dinoseb for diazomethane in the methylation step. All sediment extracts were triple- methylated to ensure full methylation; however, extended methylation also increased the possibility that some of the pesticide residues might be blown out of the sample container with the nitrogen carrier gas. In spite of these precautions, methylation of dinoseb was somewhat variable and was probably responsible for the large standard deviations seen in spike and recovery tests.

3. **Losses During Evaporation of Extracts** - After the methylation step in the extraction procedure, losses of dinoseb methyl ether can occur during rotary evaporation of the extracts. Addition of iso-octane to theextracts, low heating bath temperatures (35 C) and close monitoring of the evaporation step minimised possible losses of dinoseb.

4. **Gas Liquid Chromatography** - Co-extractives which were not removed by Florisil column chromatography may cause interferences in GC-ECD detection. This was particularly a problem with sediment and water samples from site #2 - 2nd SE ditch, which had an interfering compound with a similar GC retention time to dinoseb. Gas chromatography using a DB-17 column eliminated the possibility of incorrect identification of this interferent.

5. **Sample Preservation** - Adsorption of pesticides onto glass surfaces of sample container may lower recovery efficiency. Acidification of the sample during field collection minimized this possibility. Tests of acidified and unacidified deionized water spiked with dinoseb showed that dinoseb was fully recovered when the sample was extracted within 24 hours. Sediments were frozen prior to analysis and did not require addition of an acid preservative.

6. **Water Sample Preparation** - Inclusion of suspended solids in the water samples analysed may skew the analytical results. Unfiltered water samples may show higher pesticide levels than filtered samples due to pesticides adsorbed onto suspended solids.

7. **Sediment Sample Preparation** - The inhomogeneity of sediment samples may lead to errors in obtaining representative sub-samples for analysis. Many of the sediments contained organic debris and other coarse material which made it difficult to obtain a truly homogeneous sample.

4.5 DISCUSSION OF ANALYTICAL METHOD

The method used for analysis of dinoseb and endosulfans I and II was based mainly on methods used by the British Columbia Ministry of Environment and Parks (MOEP) Environmental Laboratory to analyse the target compounds in water and soil (Anon. 1988). These Anon. 1988) procedures analyse dinoseb and endosulfan separately.

Discussions with A.S.Y. Chau (1989) (National Water Resources Institute of Environment Canada, Burlington, Ontario) indicated that dinoseb and endosulfan could be determined in a single procedure. Accordingly, established procedures for separate analyses of the target pesticides (Wegman 1983, Anon. 1988) were modified to allow simultaneous extraction and detection of dinoseb and endosulfans I & II in water and sediment samples.

Where adjustments to established methods were made, the integrity of the procedures was checked before samples were analysed. Combined analysis of dinoseb and endosulfan required that water samples be acidified to minimise dinoseb adsorption onto the surface of glass storage bottles and that the extract be methylated for detection of dinoseb by GC-ECD. Preliminary tests confirmed that endosulfan did not degrade in the acidified storage bottles and during methylation. Florisil column chromatography conditions for clean-up and elution of dinoseb and endosulfans I and II were also established prior to sample work-up.

5.0 RECOMMENDATIONS

If additional analyses for dinoseb and endosulfan are required for future samples from similar sampling sites, it is recommended that alternative procedures be developed to improve the precision of the analytical method. Such work might include the following:

- Reduce the amount of interferent compounds, especially in sediments, by more selective extraction materials and clean-up procedures;
- Investigate alternative derivatization methods for analysis of dinoseb. Alternatively, evaluate other chromatographic methods, such as liquid chromatography, which will not require a derivatization step for determination of dinoseb.

6.0 REFERENCES

Anon. 1988. Organic methods manual update. Project Report No. 4-03-966. Prepared by British Columbia Research Corp. for B.C. Ministry of Environment and Parks. Vancouver, B.C.

Wegman, R.C.C. and Wammes, J.I. 1983. Determination of nitrophenols in water and sediment samples by high-performance liquid chromatography. Mededelingen van de Faculteit Landbouwetenschappen, Rijksuniversiteit Gent. 48(4): 961-969.

Chau, A.S.Y. 1989. Personal communication to J. McKinley (B.C. Research Corp.).

TABLE 1A. CONCENTRATIONS OF DINOSEB AND ENDOSULFANS I & II IN SAMPLES OF SEDIMENT AND WATER COLLECTED FROM THE NICOMEKL RIVER AND FARMLAND DITCHES ON MARCH 16, 1989.

Detection limits:
Sediment = 10 ppb
Water = 0.1 ppb

	SEDIMENT (ppb)[1]			WATER (ppb)		
	Dinoseb	Endo[2] I	Endo II	Dinoseb	Endo I	Endo II
#1 - NE DITCH	tr[3]	-	-	-	-	-
#1 - SE DITCH	tr	tr	tr	-	-	-
#1 - UPRIVER	tr	-	-	-	-	-
#1 - DOWNRIVER	-	-	-	-	-	-
#2 - NW DITCH	49	tr	-	-	-	-
#2 - SE DITCH	tr	tr	tr	-	-	-
#2 - 2ND SE DITCH	36	72	356	-	-	tr[2]
#2 - UPRIVER	37	-	-	-	-	-
#2 - DOWNRIVER	tr	-	-	-	-	-
#3 - NE DITCH	29	-	-	-	-	-
#3 - DOWNRIVER	tr	-	tr	-	-	-
#4 - REFERENCE (RIVER AT 64TH AVE)	tr	-	-	-	-	-

[1]Sediment results are calculated on a dry-weight basis.
[2]Endosulfan.
[3]The "tr" designation indicates that the corresponding pesticides were observed below the detection limit of 10 ppb for sediment and 0.1 ppb for water.

TABLE 2A. DRY SOLIDS CONTENT OF SEDIMENTS.

SITE/LOCATION	DRY SOLIDS CONTENT (%)
#1 - NE DITCH	60.0
#1 - SE DITCH	45.0
#1 - UPRIVER	55.5
#1 - DOWNRIVER	54.0
#2 - NW DITCH	51.5
#2 - SE DITCH	52.5
#2 - 2ND SE DITCH	39.0
#2 - UPRIVER	51.0
#2 - DOWNRIVER	56.5
#3 - NE DITCH	48.0
#3 - DOWNRIVER	61.5
#4 - REFERENCE (RIVER AT 64TH AVE.)	65.0

TABLE 3A. SPIKE & RECOVERY OF PESTICIDES IN SEDIMENT AND WATER.

	RECOVERY (%)			
	SPIKE A	SPIKE B	SPIKE C	MEAN (SE)[1]
WATER (Spike Level = 0.5 ppb)				
Dinoseb	61	122	126	103 (21)
Endosulfan I	104	94	51	83 (16)
Endosulfan II	116	100	91	102 (8)
SEDIMENT (Spike Level = 20 ppb)				
Dinoseb	164	125	106	132 (17)
Endosulfan I	92	89	82	88 (2)
Endosulfan II	98	93	90	94 (2)

[1]Standard error.

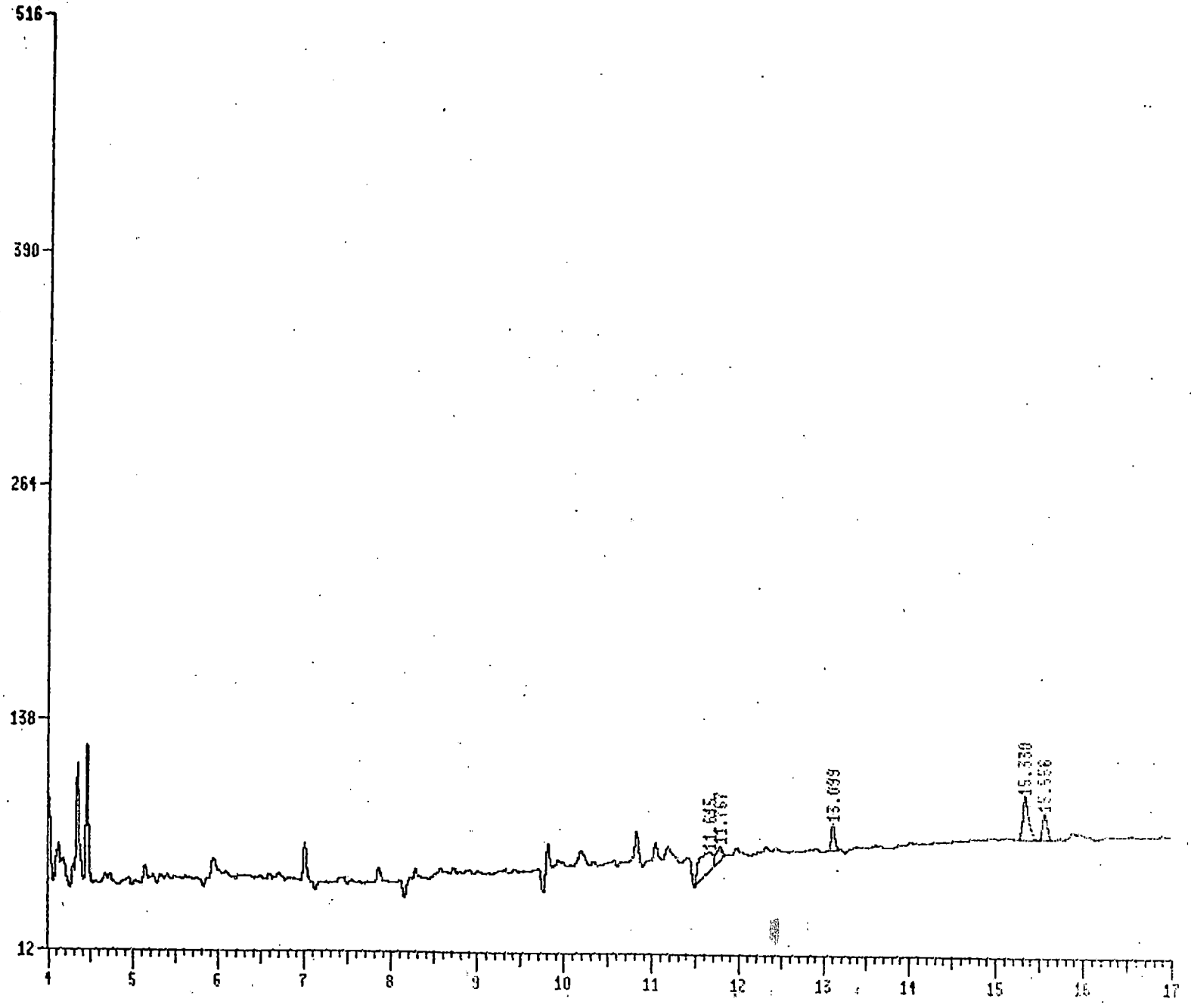

FIGURE 1A. GC-ECD CHROMATOGRAM OF WATER SAMPLE FROM SITE #1 - SE DITCH.

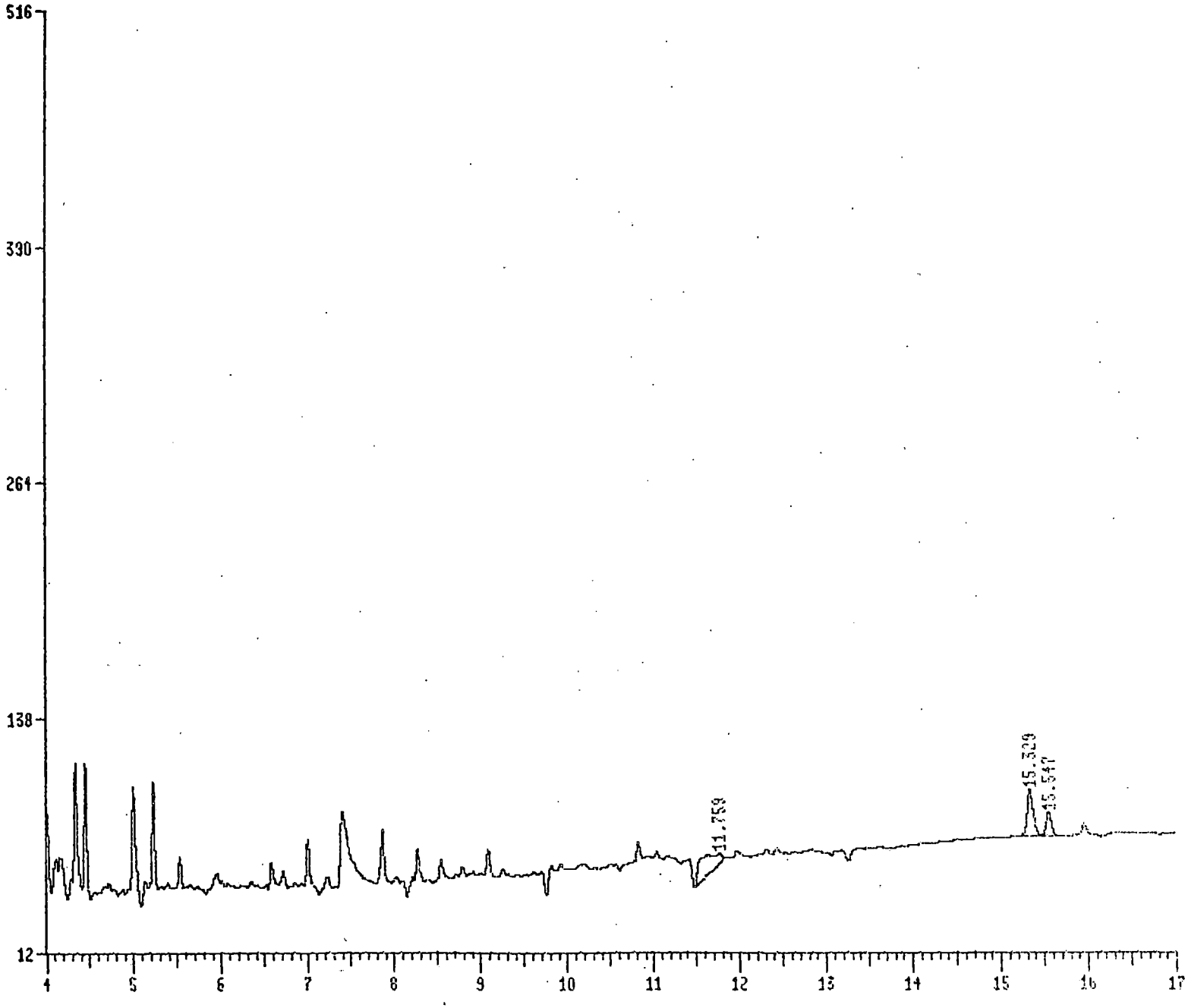

FIGURE 2A. GC-ECD CHROMATOGRAM OF WATER SAMPLE FROM SITE #4 - RIVER AT 64TH AVE.

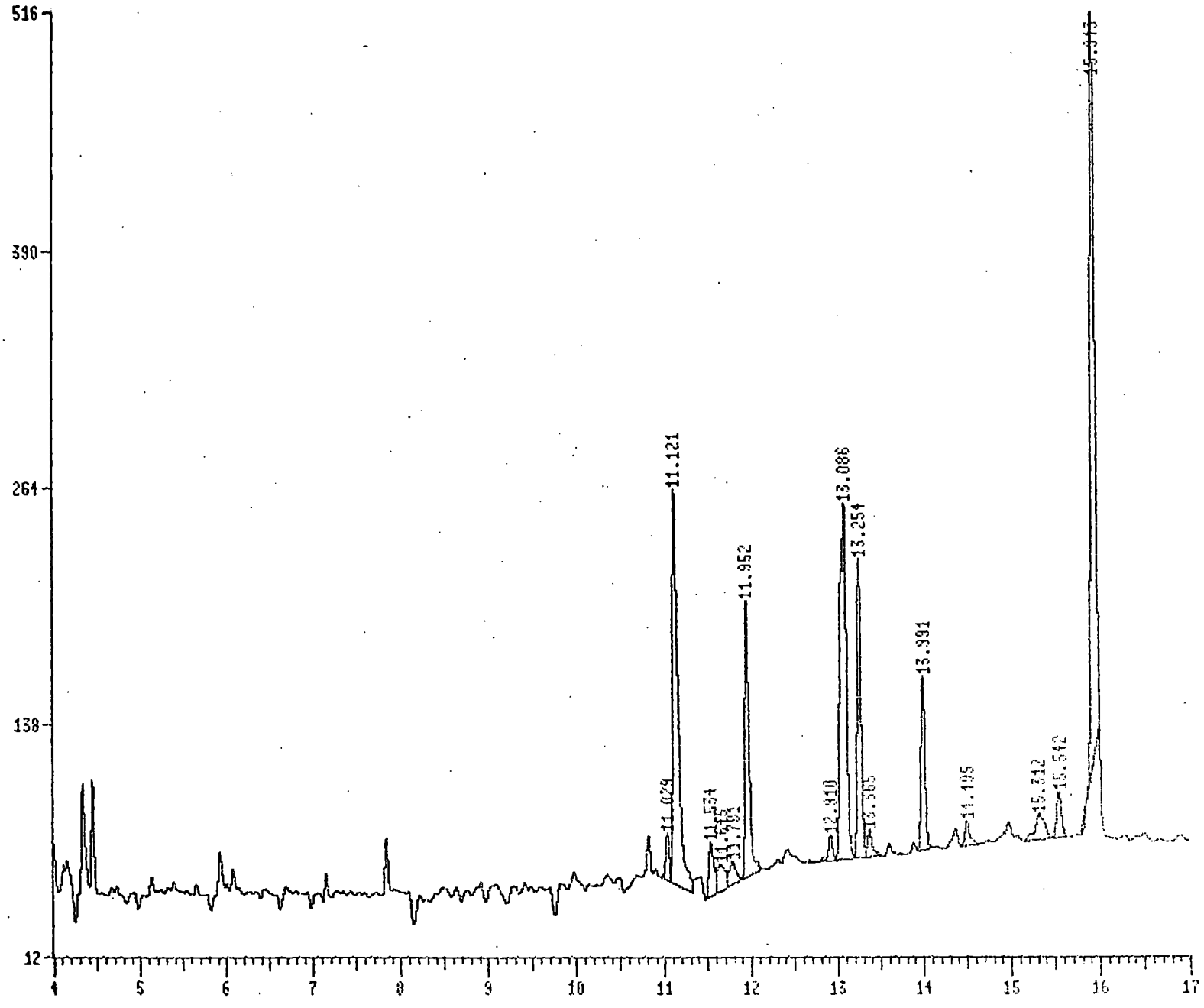

FIGURE 3A. GC-ECD CHROMATOGRAM OF WATER SAMPLE FROM SITE #2 - 2ND SE DITCH.

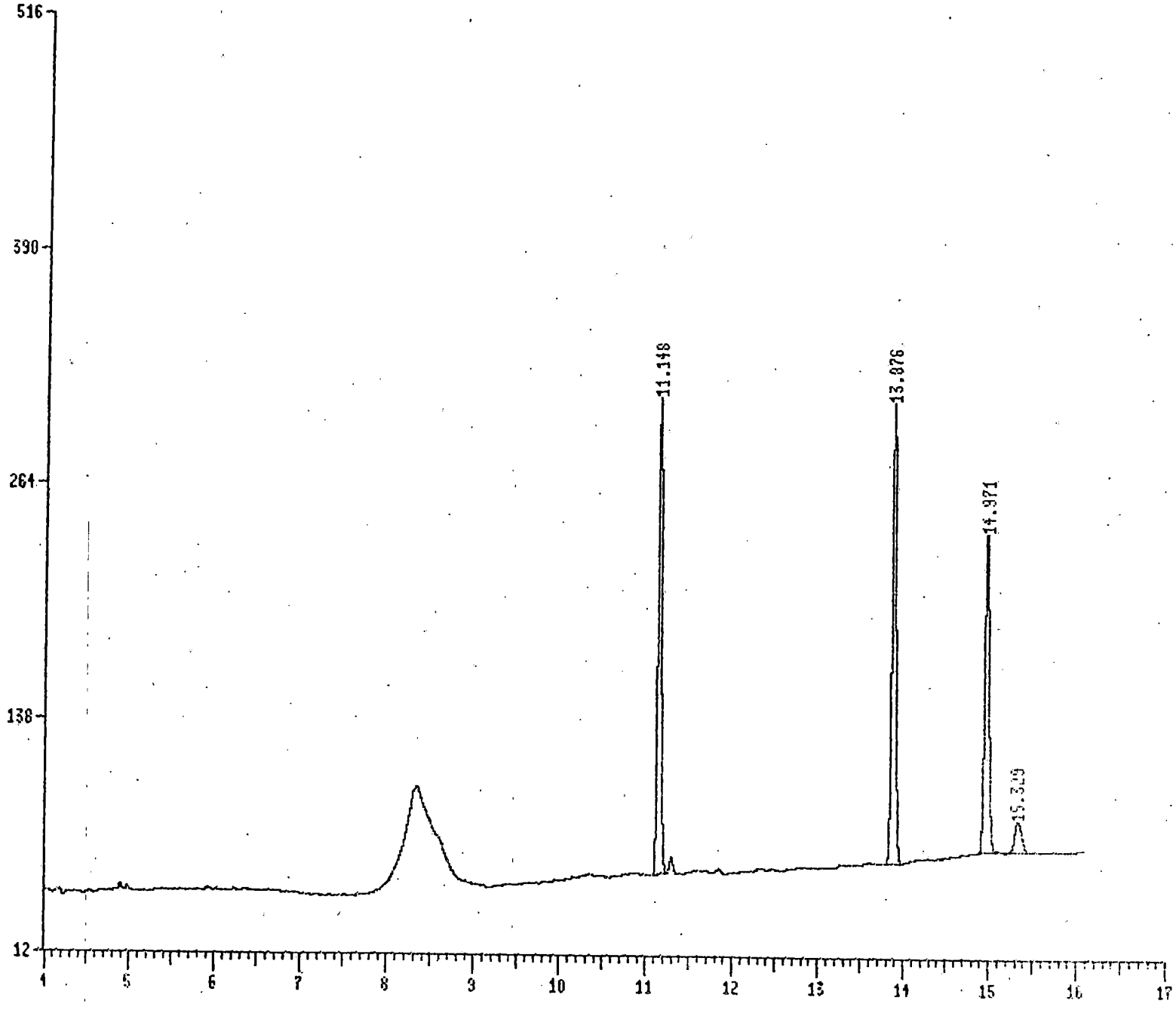

FIGURE 4A. GC-ECD CHROMATOGRAM OF 0.05 PPM STANDARD MIXTURE OF DINOSEB METHYL ETHER, ENDOSULFAN I AND ENDOSULFAN II

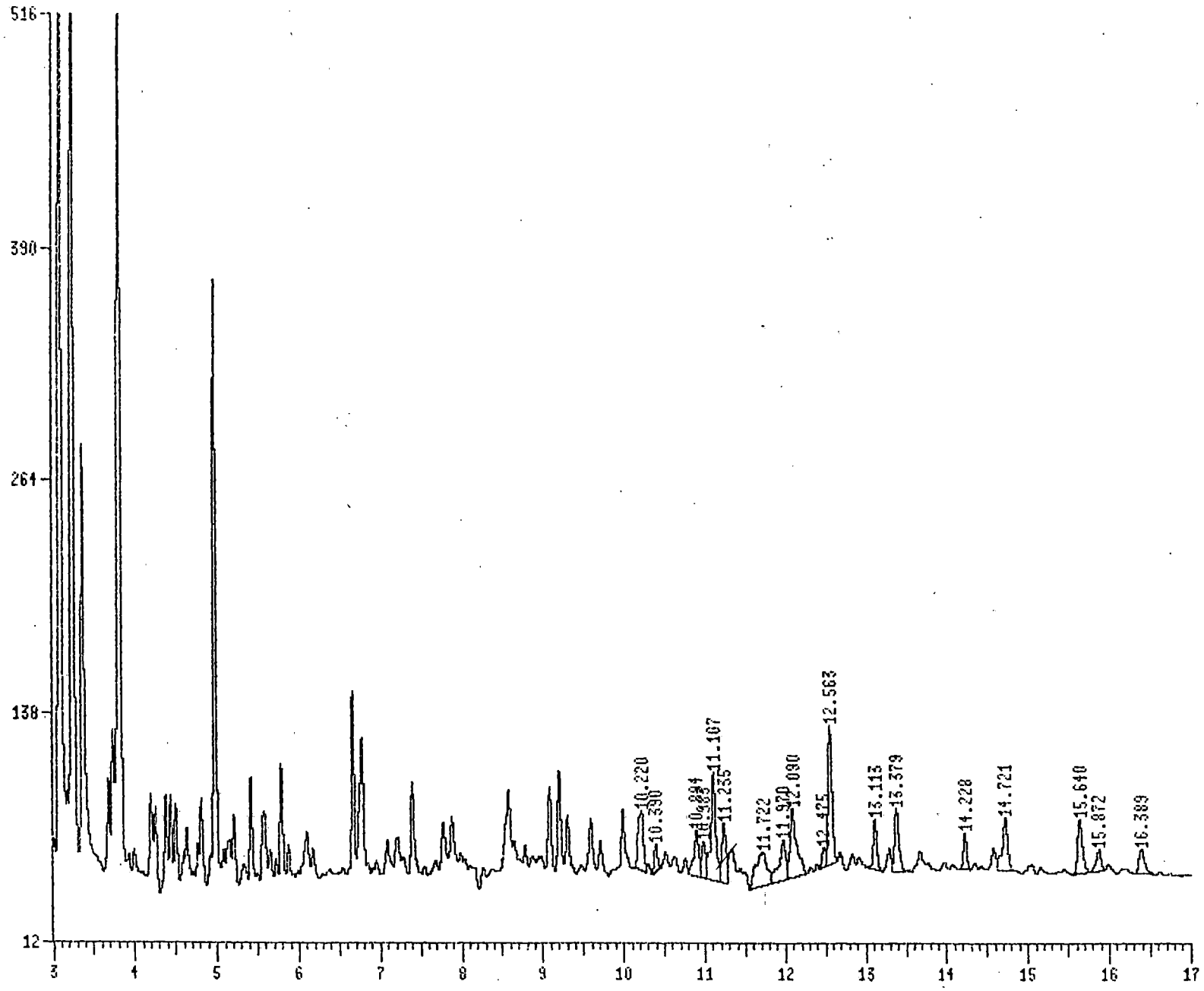

FIGURE 5A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #1 - UPRIVER.

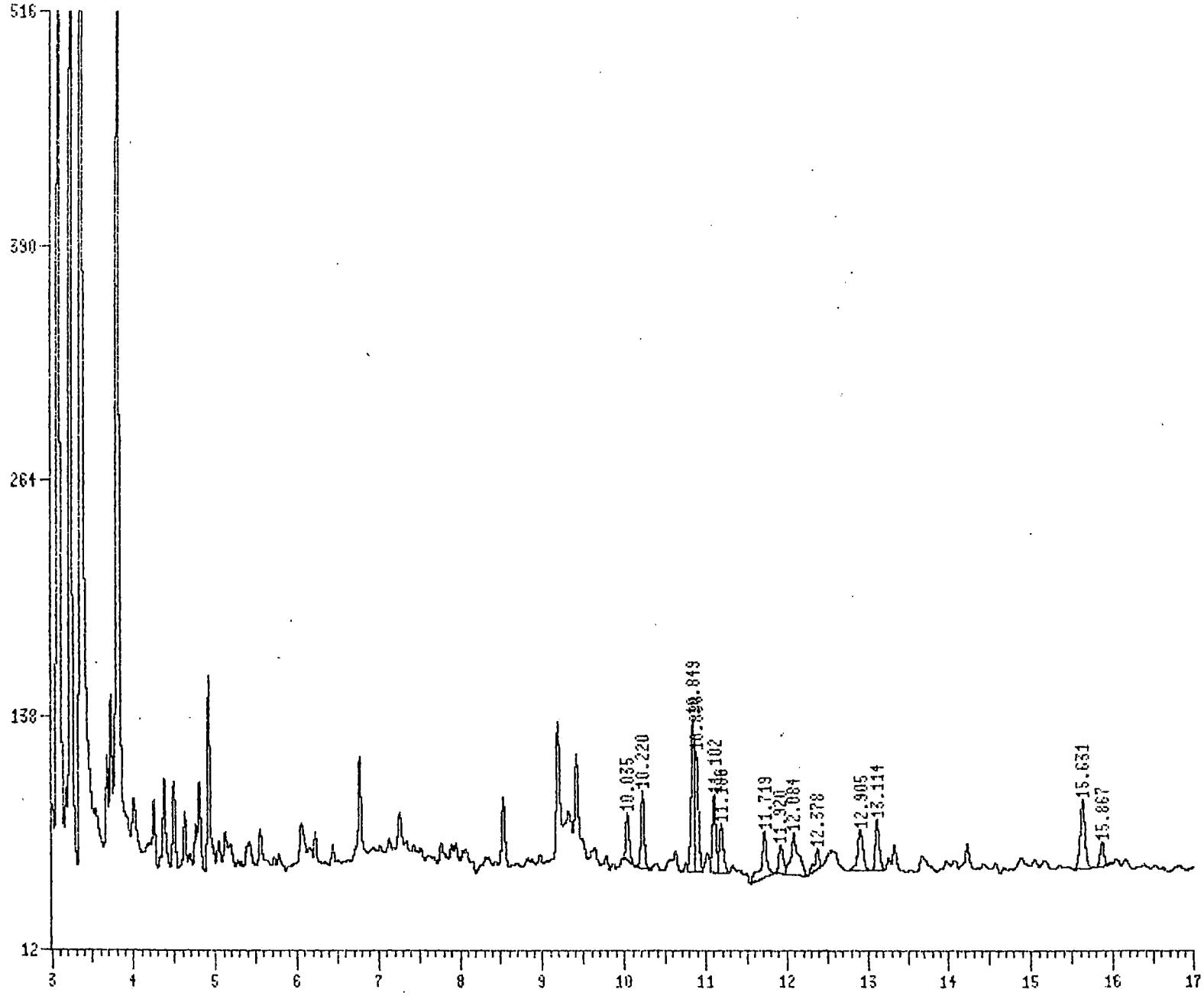

FIGURE 6A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #1 - DOWNRIVER.

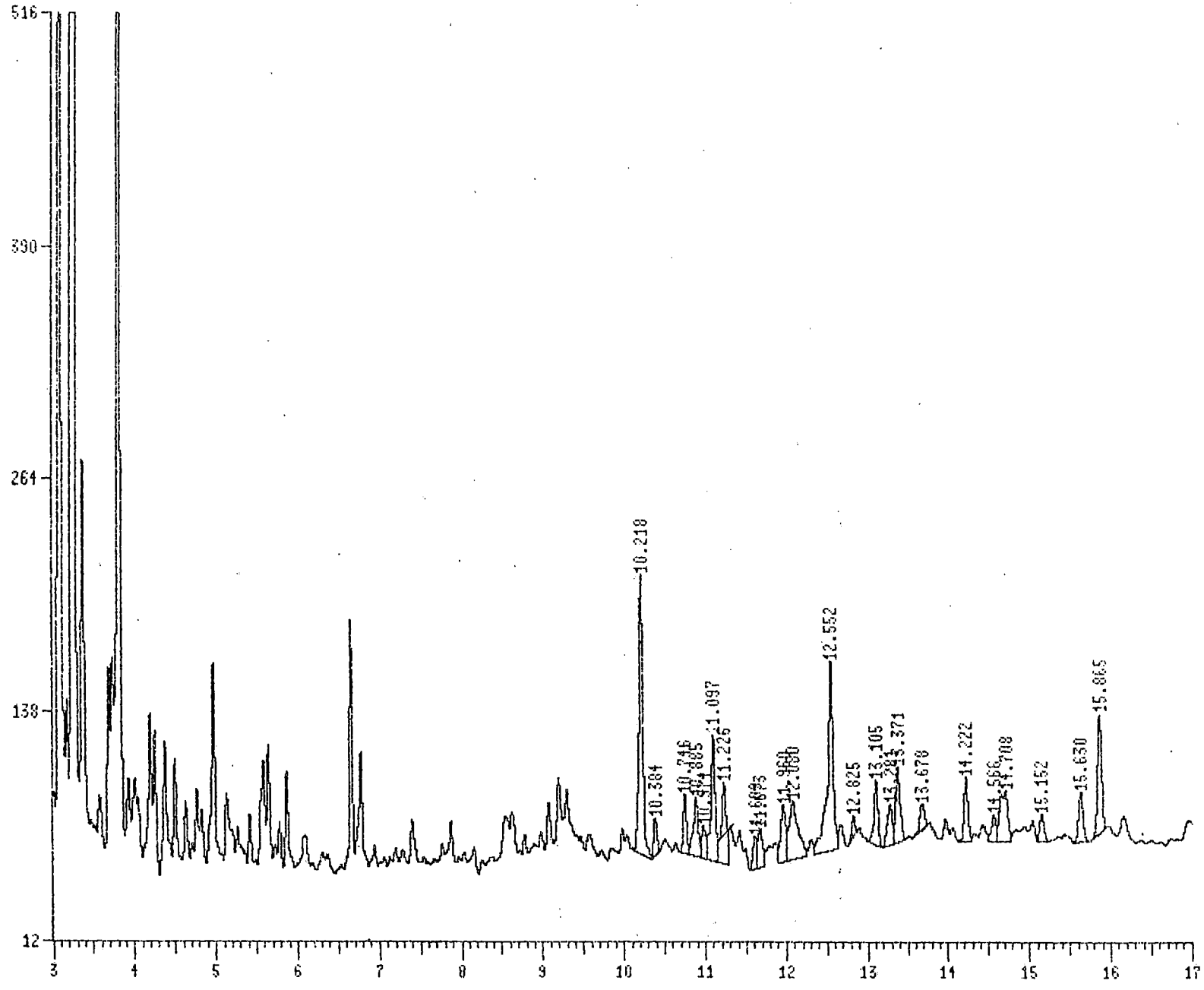

FIGURE 7A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #1 - NE DITCH.

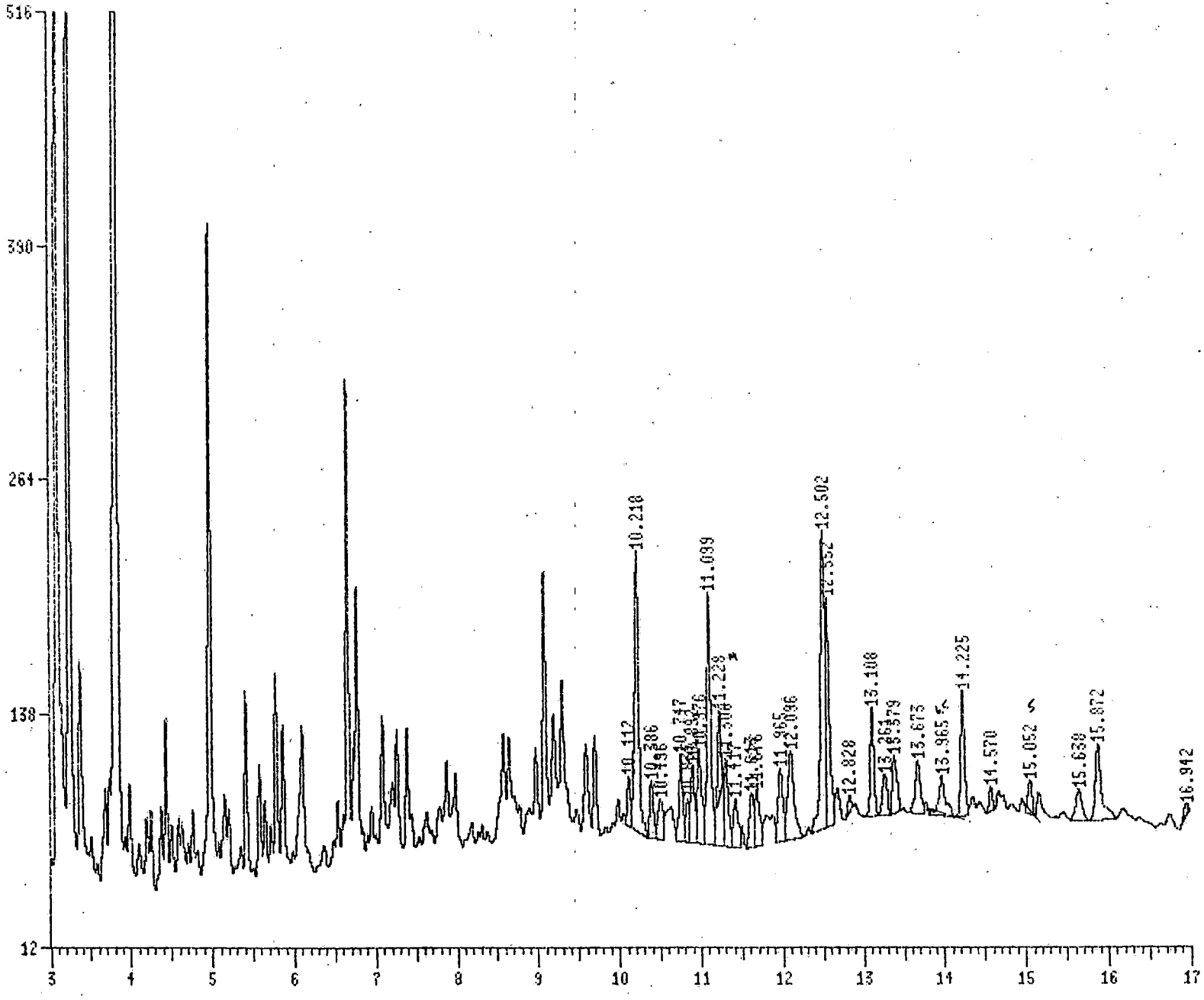

FIGURE 8A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #1 - SE DITCH.

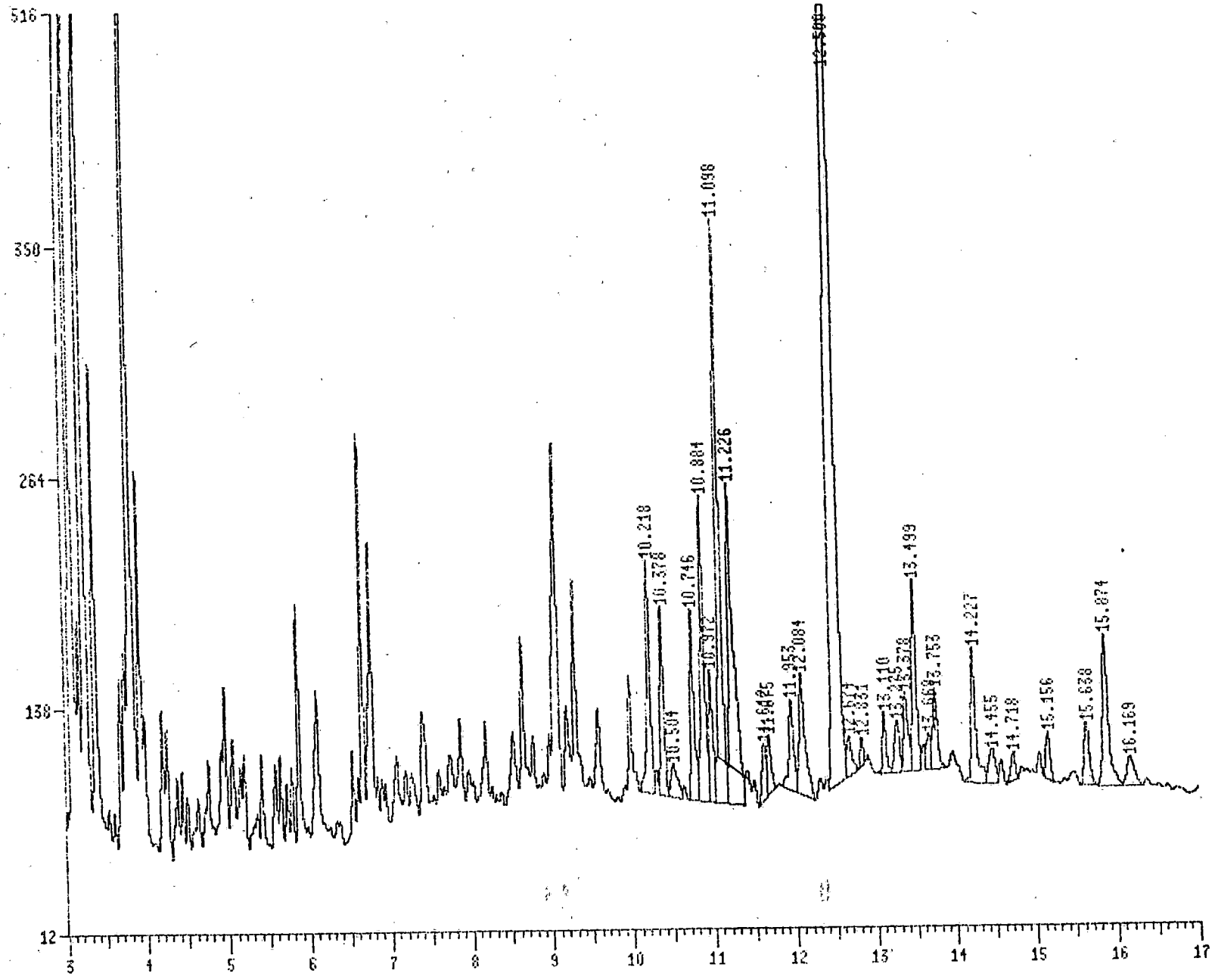

FIGURE 9A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #2 - UPRIVER.

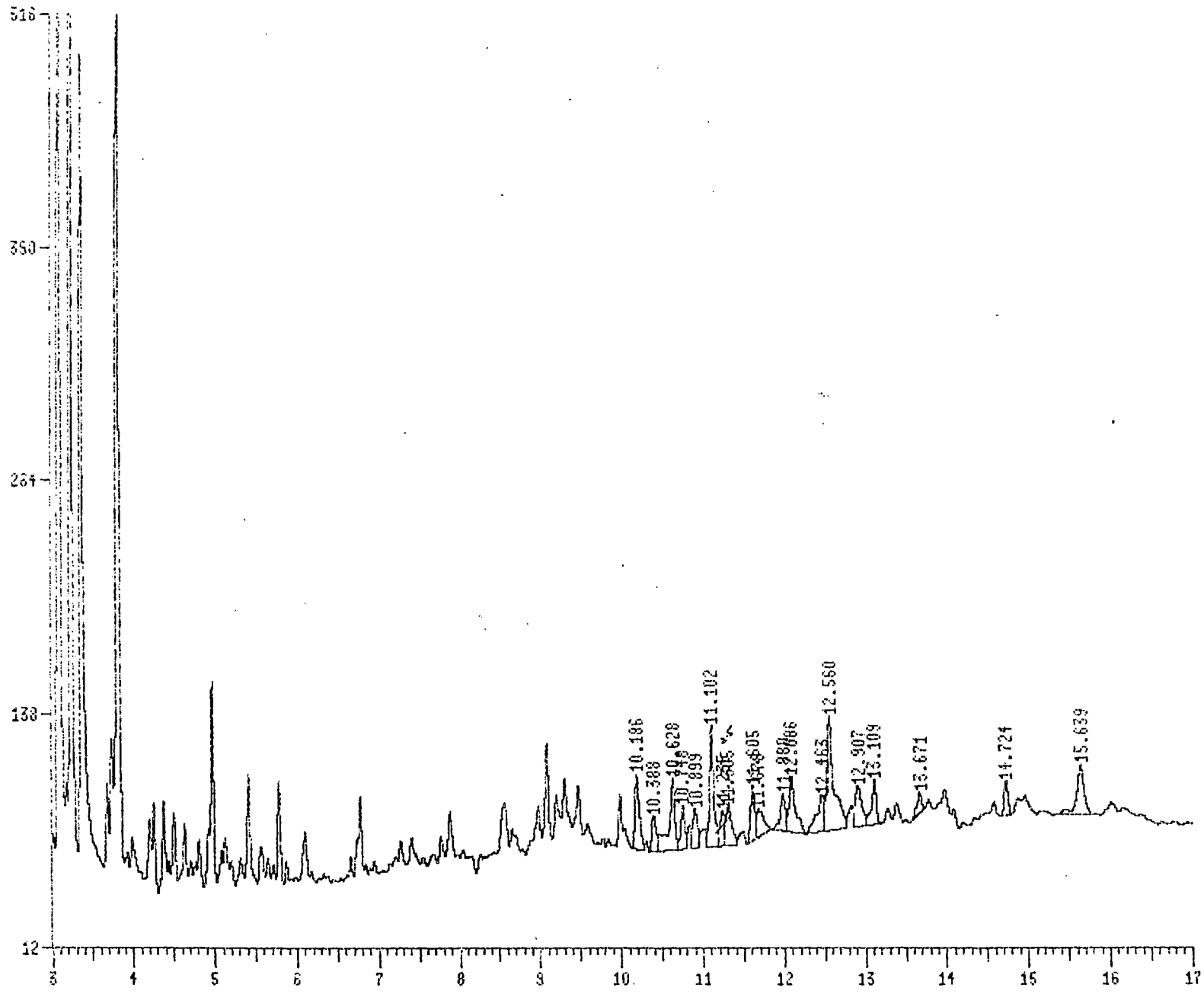

FIGURE 10A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #2 - DOWNRIVER.

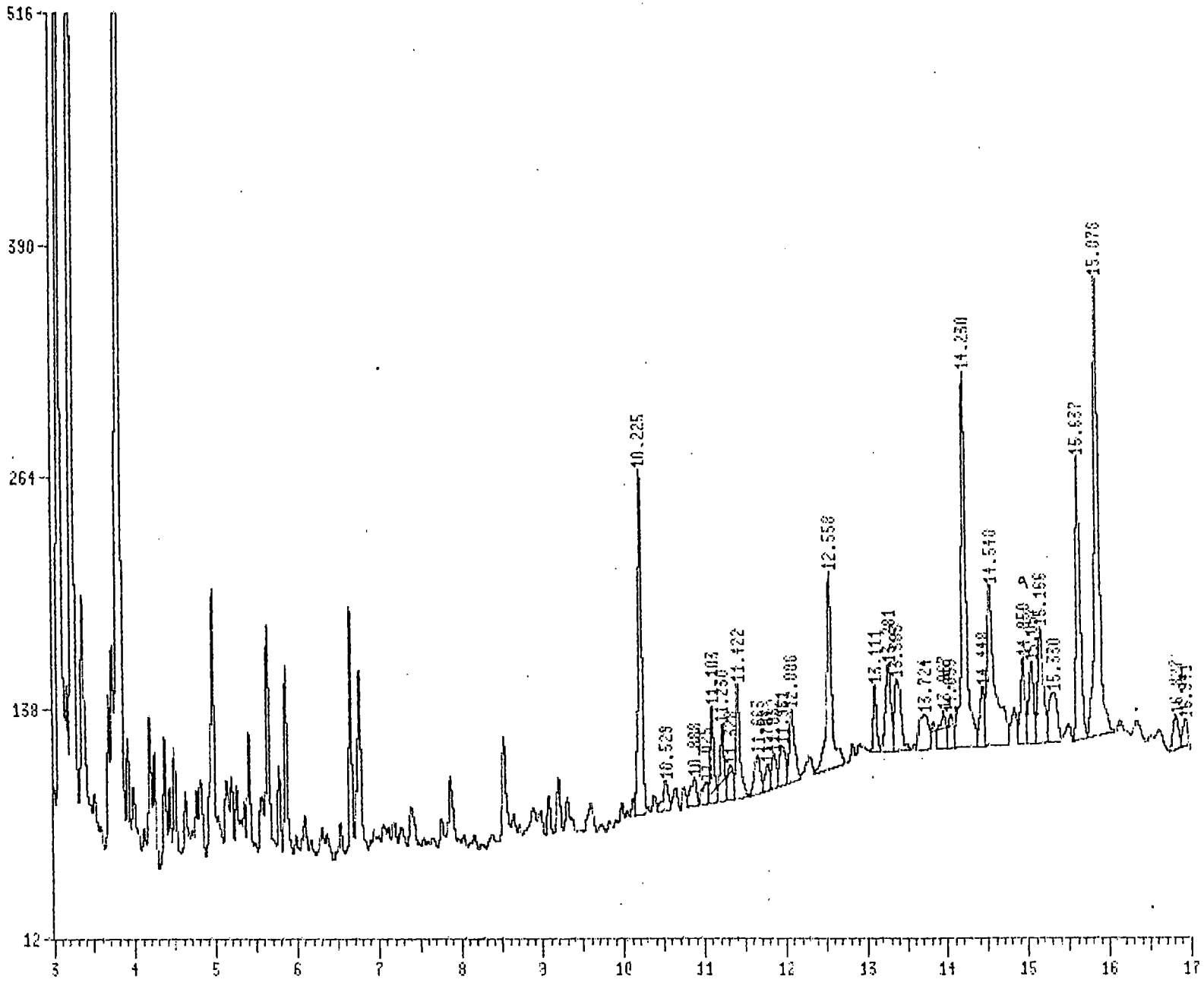

FIGURE 11A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #2 - SE DITCH.

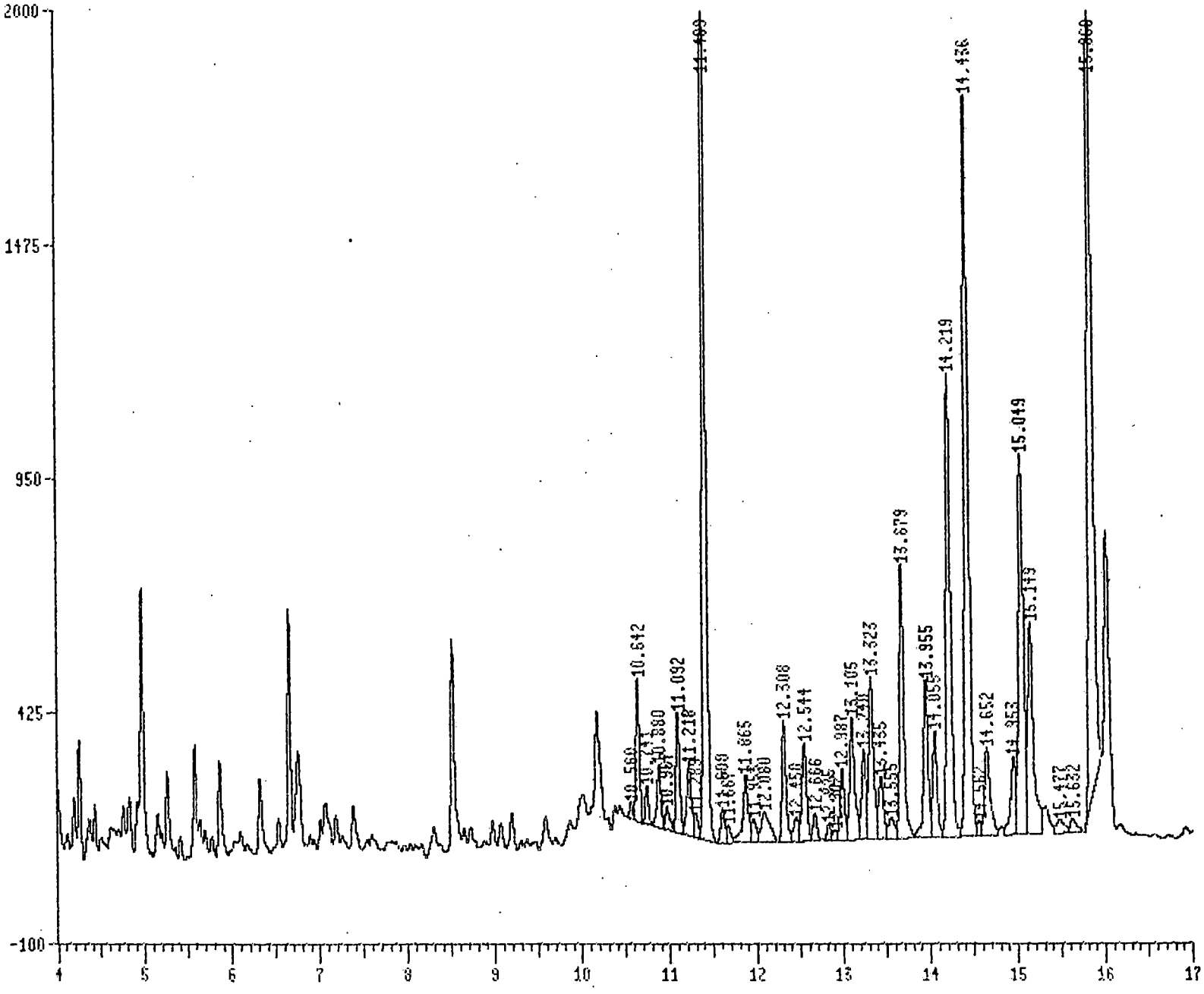

FIGURE 12A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #2 - 2ND SE DITCH.

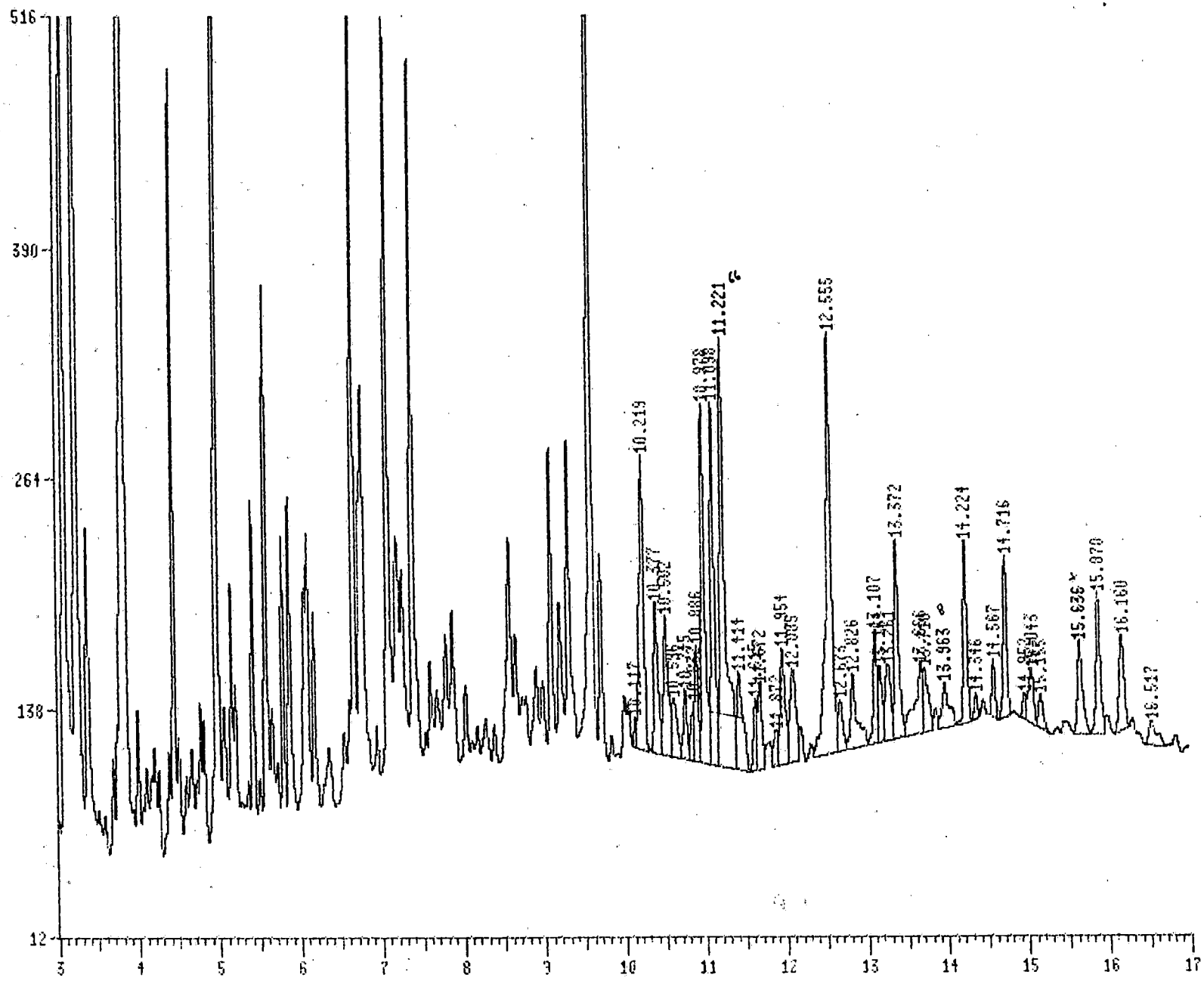

FIGURE 13A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #2 - NW DITCH.

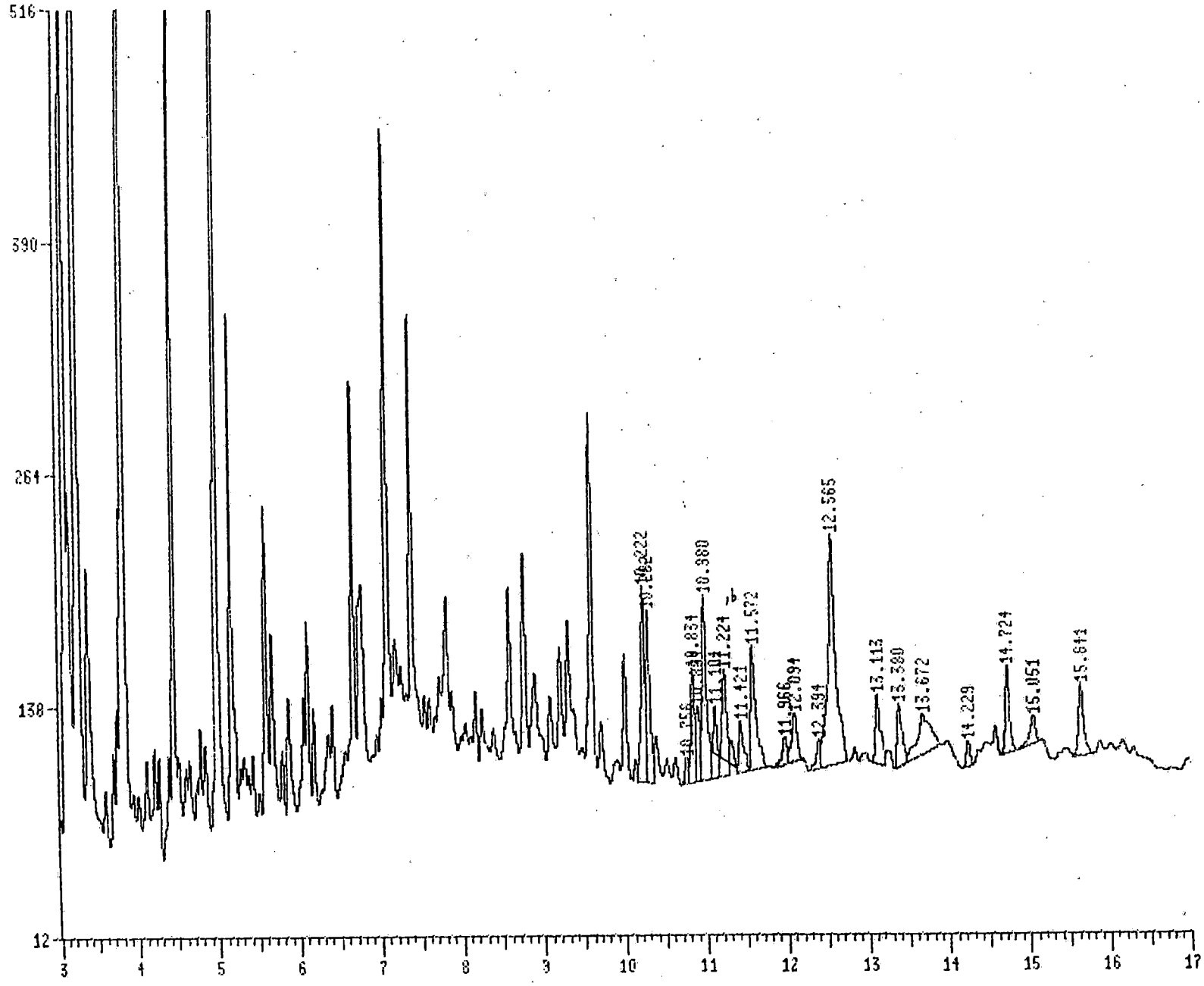

FIGURE 14A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #3 - DOWNRIVER.

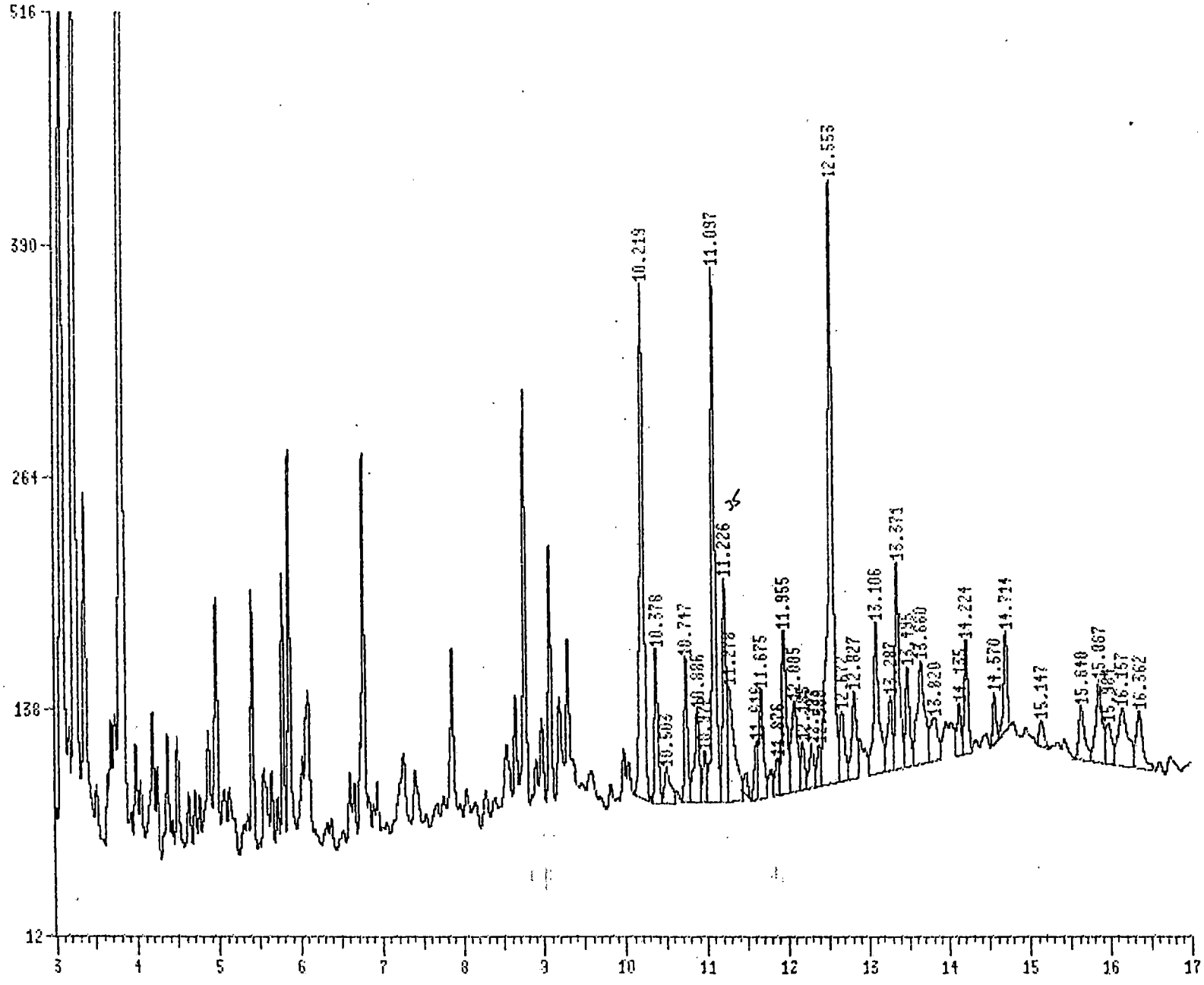

FIGURE 15A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #3 - NE DITCH.

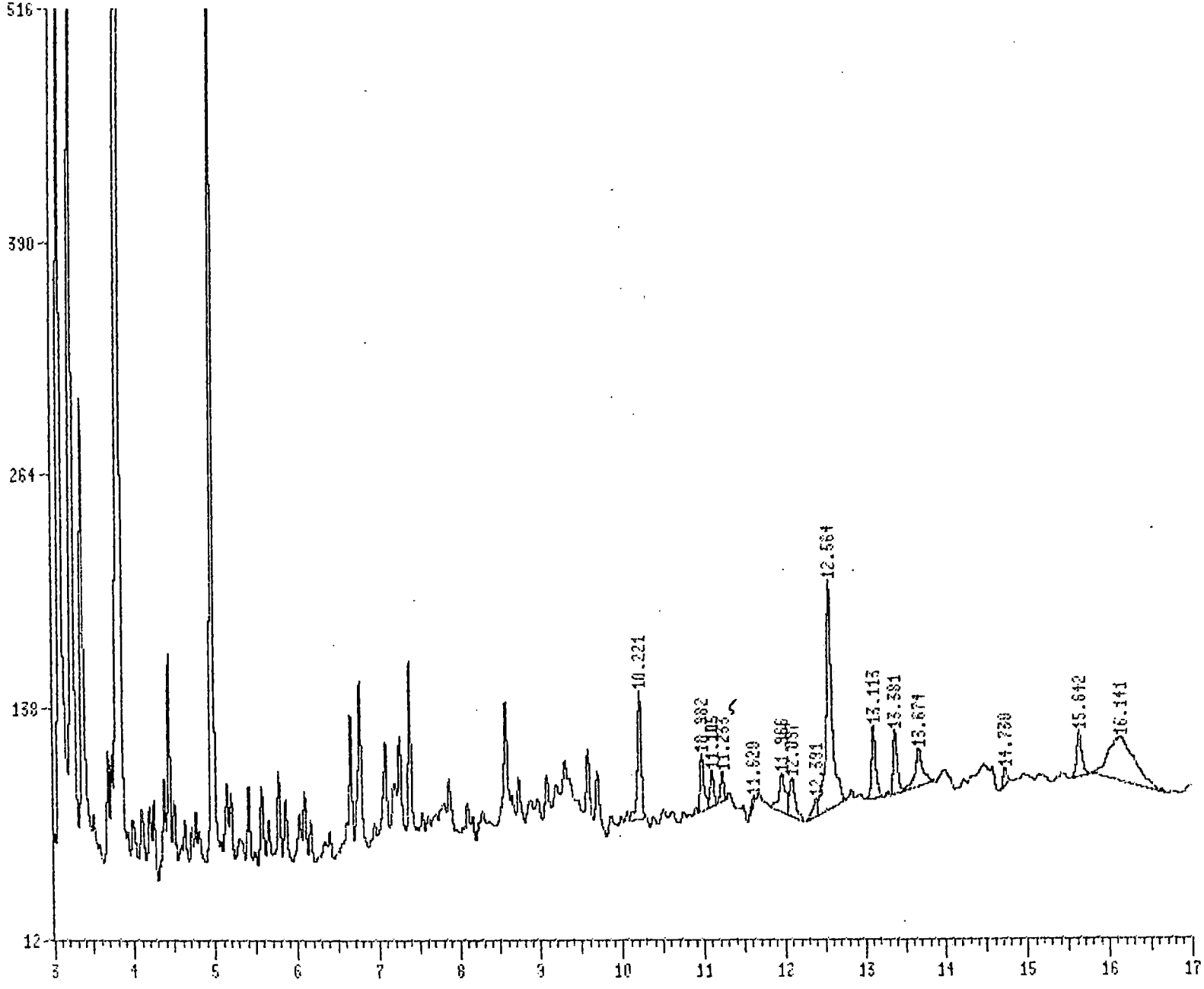

FIGURE 16A. GC-ECD CHROMATOGRAM OF SEDIMENT SAMPLE FROM SITE #4 - RIVER AT 64TH AVE.

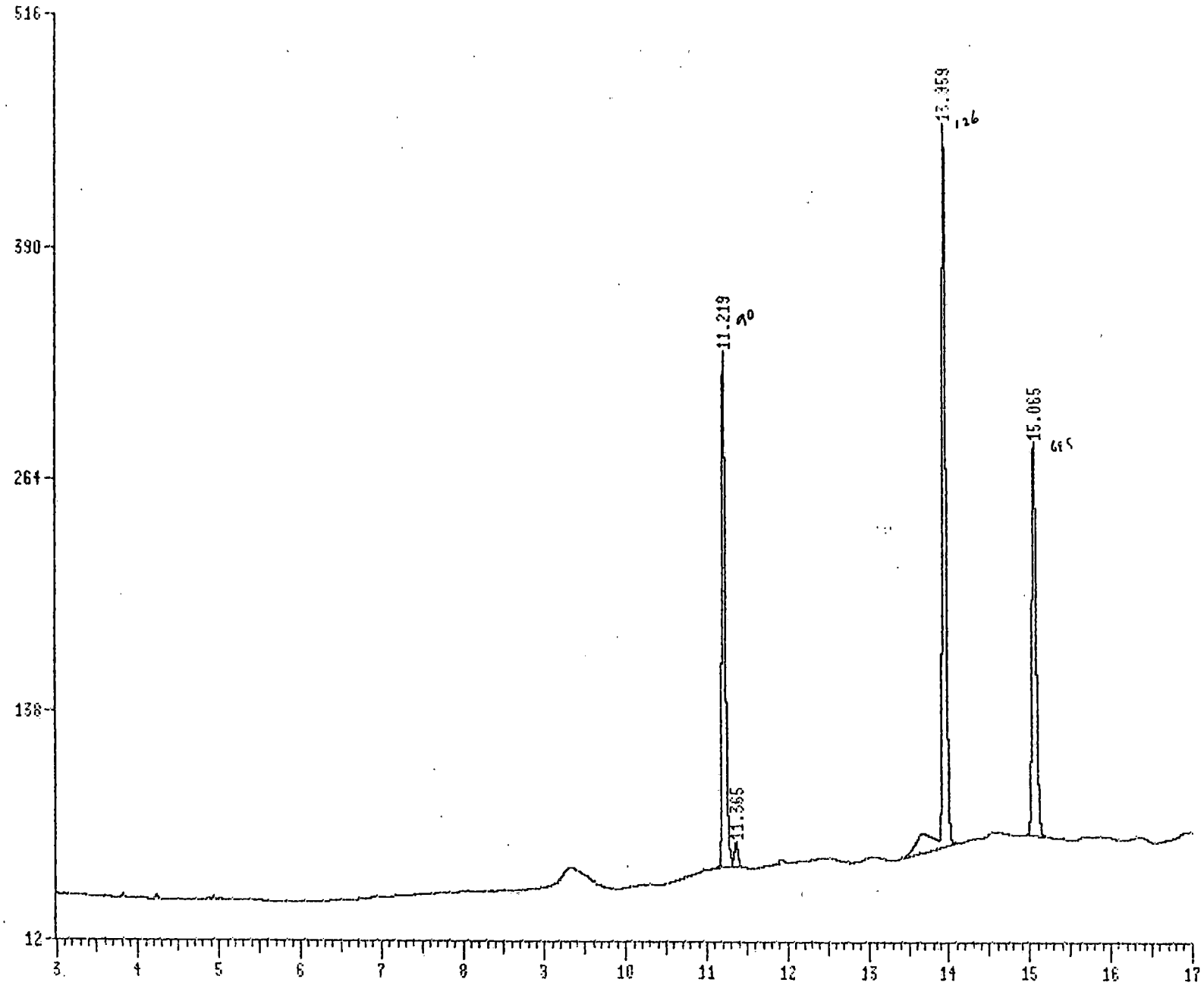

FIGURE 17A. GC-ECD CHROMATOGRAM OF 0.10 PPM STANDARD MIXTURE OF DINOSEB METHYL ETHER, ENDOSULFANS I & II

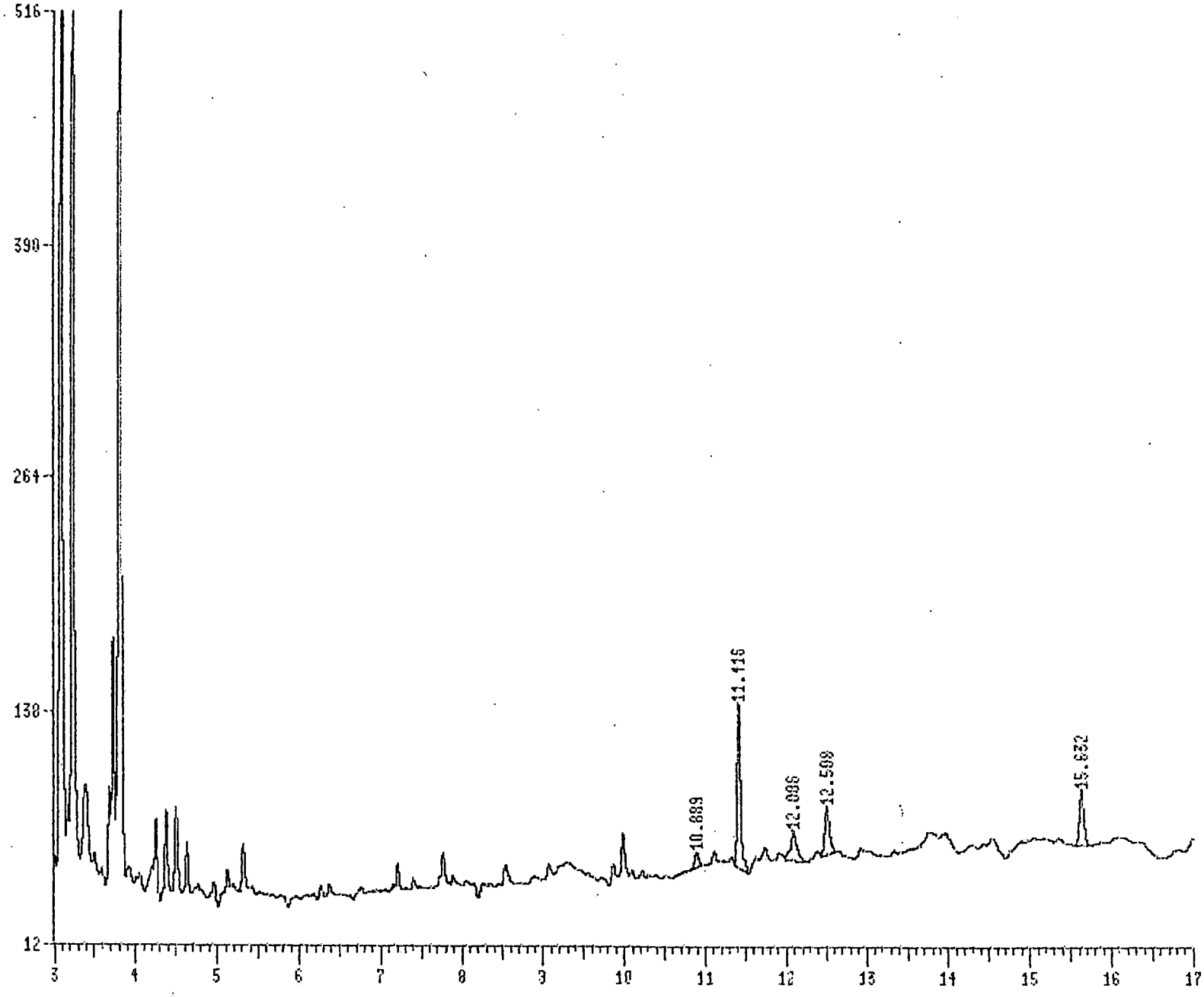

FIGURE 18A. GC-ECD CHROMATOGRAM OF QUALITY CONTROL BLANK FOR SOLVENT AND REAGENT BACKGROUND

APPENDIX IV. Monthly Climate Summary For the Period March 1, 1988 to March 16, 1989 Incl., As Recorded at the Nearest Weather Station (5.5 km WNW of Sample Site No. 1).

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR MARCH 1988

B.C. REGIONAL ID : 7140J
AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H	Z R	H A	SNOW GND
1	10.5	5.5	8.0	3.4		3.4				0
2	8.5	6.5	7.5	8.8		8.8				0
3	10.0	4.0	7.0	2.0		2.0				0
4	8.5	2.5	5.5	1.8		1.8				0
5	8.5	3.5	6.0	7.7		7.7				0
6	10.5	4.0	7.3							0
7	9.0	3.0	6.0	11.0		11.0				0
8	7.5	4.0	5.8	15.8		15.8				0
9	9.0	4.0	6.5	.4		.4				0
10	10.0	1.0	5.5	.2		.2				0
11	10.0	-1.0	4.5							0
12	10.0	1.5	5.8							0
13	10.5	1.5	6.0							0
14	12.0	1.5	6.8							0
15	16.5	1.0	8.8							0
16	15.0	1.5	8.3							0
17	13.5	2.5	8.0							0
18	16.5	3.0	9.8							0
19	14.0	6.0	10.0	4.0		4.0				0
20	10.0	7.0	8.5	13.6		13.6				0
21	11.5	3.0	7.3	6.2		6.2				0
22	9.5	4.0	6.8	23.8		23.8				0
23	9.5	5.5	7.5	1.6		1.6				0
24	6.0	4.0	5.0	5.6		5.6				0
25	8.0	3.5	5.8	33.1		33.1				0
26	8.0	3.5	5.8	4.9		4.9				0
27	10.0	1.0	5.5							0
28	7.5	-.5	3.5	16.6		16.6				0
29	9.5	1.0	5.3	.2		.2				0
30	10.5	1.5	6.0							0
31	9.0	4.0	6.5	1.4		1.4				0
TOTAL	319.0	93.0		162.1	0.0	162.1	0	0	0	
MEAN	10.3	3.0	6.7							

MONTHLY MAXIMUM TEMPERATURE WAS 16.5 ON DAY 15 18
MONTHLY MINIMUM TEMPERATURE WAS -1.0 ON DAY 11
HIGHEST RAINFALL WAS 33.1 ON DAY 25
NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 20

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 351.4

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR APRIL 1988

B.C. REGIONAL ID : 7140J
AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	TEMPERATURE MIN	TEMPERATURE MEAN	TOTAL RAIN	TOTAL SNOW	TOTAL PCP	T H	Z R	H A	SNOW GND
1	10.5	4.0	7.3	15.8		15.8				0
2	10.0	6.5	8.3	20.8		20.8				0
3	9.0	5.5	7.3	2.0		2.0				0
4	9.5	6.0	7.8	8.2		8.2				0
5	10.5	3.0	6.8	43.2		43.2				0
6	10.5	2.0	6.3	3.0		3.0				0
7	11.0	4.0	7.5							0
8	10.5	2.0	6.3							0
9	16.0	1.5	8.8							0
10	17.5	1.5	9.5							0
11	18.0	5.5	11.8							0
12	20.0	5.5	12.8							0
13	16.5	5.5	11.0	T		T				0
14	14.5	10.0	12.3	.6		.6				0
15	17.0	7.5	12.3	.4		.4				0
16	14.0	11.0	12.5	.5		.5				0
17	10.5	8.5	9.5	.4		.4				0
18	16.0	8.5	12.3							0
19	19.0	7.0	13.0							0
20	20.0	4.5	12.3							0
21	17.5	6.5	12.0							0
22	14.5	7.0	10.8	2.2		2.2				0
23	14.0	7.5	10.8	1.0		1.0				0
24	13.5	7.0	10.3	T		T				0
25	13.5	3.5	8.5							0
26	16.5	3.5	10.0							0
27	18.0	6.5	12.3	.8		.8				0
28	15.5	8.0	11.8	17.0		17.0				0
29	11.5	6.0	8.8	2.8		2.8				0
30	13.0	6.0	9.5	7.4		7.4				0
TOTAL	428.0	171.0		126.1	0.0	126.1	0	0	0	
MEAN	14.3	5.7	10.0							

MONTHLY MAXIMUM TEMPERATURE WAS 20.0 ON DAY 12 20
MONTHLY MINIMUM TEMPERATURE WAS 1.5 ON DAY 9 10
HIGHEST RAINFALL WAS 43.2 ON DAY 5
NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 16

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 239.5

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR MAY 1988

B.C. REGIONAL ID : 7140J
AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H	Z R	H A	SNOW GND
1	13.0	3.5	8.3	10.0		10.0				0
2	8.0	3.5	5.8	13.8		13.8				0
3	13.0	4.5	8.8	1.0		1.0				0
4	15.0	6.0	10.5							0
5	14.0	4.5	9.3	.8		.8				0
6	14.0	7.5	10.8	T		T				0
7	18.0	6.5	12.3							0
8	19.0	6.5	12.8							0
9	17.5	6.5	12.0							0
10	24.0	11.5	17.8							0
11	25.5	12.5	19.0							0
12	22.5	13.5	18.0	25.0		25.0				0
13	14.5	9.0	11.8	10.1		10.1				0
14	16.5	8.5	12.5	1.1		1.1				0
15	18.5	8.5	13.5	16.4		16.4				0
16	12.0	9.0	10.5	8.8		8.8				0
17	13.5	7.5	10.5	4.0		4.0				0
18	11.0	7.5	9.3	1.0		1.0				0
19	16.5	8.0	12.3	T		T				0
20	19.0	6.0	12.5							0
21	25.5	8.5	17.0							0
22	15.5	13.5	14.5	6.9		6.9				0
23	M	6.5	M	1.2		1.2				0
24	16.5	6.0	11.3	.6		.6				0
25	17.5	9.0	13.3							0
26	17.0	9.0	13.0	4.2		4.2				0
27	16.0	11.0	13.5	11.7		11.7				0
28	15.5	10.0	12.8	.2		.2				0
29	15.5	7.0	11.3	4.1		4.1				0
30	16.0	6.0	11.0	1.2		1.2				0
31	16.0	9.0	12.5	19.0		19.0				0
TOTAL	496.0	246.0		141.1	0.0	141.1	0	0	0	
MEAN	16.5	7.9	12.2							

MONTHLY MAXIMUM TEMPERATURE WAS 25.5 ON DAY 11 21
MONTHLY MINIMUM TEMPERATURE WAS 3.5 ON DAY 1 2
HIGHEST RAINFALL WAS 25.0 ON DAY 12
NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 20

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 178.3

HEATING DEGREE-DAYS ESTIMATED - MEAN MONTHLY TEMP USED FOR 1 MISSING TEMP

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR JUNE 1988

B.C. REGIONAL ID : 7140J

AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	THRA	SNOW GND
1	12.0	7.0	9.5	8.0		8.0	X	0
2	13.0	6.0	9.5	5.4		5.4		0
3	17.0	10.0	13.5	1.4		1.4		0
4	16.5	8.0	12.3	.4		.4		0
5	17.0	6.0	11.5	4.6		4.6		0
6	16.0	N	M	1.6		1.6		0
7	17.5	9.5	13.5	T		T		0
8	17.5	9.5	13.5					0
9	21.0	8.0	14.5	23.0		23.0		0
10	16.5	10.5	13.5	4.0		4.0		0
11	18.0	8.5	13.3					0
12	19.5	8.5	14.0					0
13	24.0	9.5	16.8					0
14	28.0	13.0	20.5					0
15	24.5	14.0	19.3					0
16	20.5	11.0	15.8	.9		.9		0
17	14.5	11.5	13.0	.9		.9		0
18	21.5	12.5	17.0					0
19	21.0	11.0	16.0					0
20	21.5	11.0	16.3					0
21	23.0	13.0	18.0	.2		.2		0
22	21.0	15.0	18.0					0
23	18.0	10.5	14.3					0
24	23.5	8.5	16.0					0
25	24.0	12.0	18.0					0
26	23.5	12.5	18.0					0
27	18.0	12.0	15.0					0
28	17.0	9.0	13.0					0
29	19.0	11.0	15.0					0
30	20.0	10.0	15.0	.6		.6		0
TOTAL	584.0	298.5		51.0	0.0	51.0	1 0 0	
MEAN	19.5	10.3	14.9					

MONTHLY MAXIMUM TEMPERATURE WAS 28.0 ON DAY 14

MONTHLY MINIMUM TEMPERATURE WAS 6.0 ON DAY 2 5

HIGHEST RAINFALL WAS 23.0 ON DAY 9

NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 12

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 95.3

TEMPERATING DEGREE-DAYS ESTIMATED - MEAN MONTHLY TEMP USED FOR 1 MISSING

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR JULY 1988

B.C. REGIONAL ID : 7140J

AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T Z H R A	SNOW GND
1	18.5	13.0	15.8	6.8E		6.8E		0
2	21.0	13.5	17.3					0
3	21.0	11.0	16.0					0
4	17.0	8.0	12.5	1.0		1.0		0
5	15.0	11.0	13.0	9.0		9.0		0
6	18.5	10.5	14.5	1.0		1.0		0
7	22.0	9.0	15.5					0
8	24.5	11.5	18.0					0
9	24.0	12.5	18.3					0
10	24.0	11.5	17.8	3.0		3.0		0
11	16.0	12.0	14.0	2.0		2.0		0
12	15.0	12.0	13.5	10.0		10.0		0
13	19.5	12.0	15.8	.5		.5		0
14	19.5	9.0	14.3					0
15	21.5	11.0	16.3					0
16	22.5	10.0	16.3					0
17	22.5	10.0	16.3					0
18	29.0	10.5	19.8					0
19	32.5	15.0	23.8					0
20	29.5	17.0	23.3					0
21	23.0	13.0	18.0					0
22	22.0	11.5	16.8					0
23	23.5	11.0	17.3					0
24	26.0	11.5	18.8					0
25	31.0	11.5	21.3					0
26	29.5	16.5	23.0					0
27	23.0	15.5	19.3					0
28	21.5	12.5	17.0					0
29	24.5	14.0	19.3					0
30	24.0	12.5	18.3					0
31	23.5	12.5	18.0	3.0		3.0		0
TOTAL	704.5	372.0		36.3	0.0	36.3	0 0 0	
MEAN	22.7	12.0	17.4					

MONTHLY MAXIMUM TEMPERATURE WAS 32.5 ON DAY 19

MONTHLY MINIMUM TEMPERATURE WAS 8.0 ON DAY 4

HIGHEST RAINFALL WAS 10.0 ON DAY 12

NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 9

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 44.0

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR AUGUST 1988

B.C. REGIONAL ID : 7140J

AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H	Z R	H A	SNOW GND
1	24.0	12.0	18.0							0
2	28.5	11.5	20.0							0
3	27.5	13.5	20.5							0
4	26.5	13.0	19.8							0
5	18.0	13.0	15.5	3.0		3.0				0
6	17.5	11.5	14.5							0
7	21.5	11.0	16.3							0
8	21.5	12.0	16.8							0
9	23.0	15.0	19.0	.2		.2				0
10	22.5	12.5	17.5							0
11	23.0	12.0	17.5							0
12	20.5	12.0	16.3							0
13	21.0	13.0	17.0							0
14	20.0	13.0	16.5							0
15	15.0	11.0	13.0	24.8		24.8				0
16	17.5	13.0	15.3	5.8		5.8				0
17	20.0	12.5	16.3							0
18	21.0	11.5	16.3	.8		.8				0
19	19.5	13.5	16.5	T		T				0
20	20.0	11.0	15.5							0
21	21.0	9.5	15.3							0
22	27.0	9.5	18.3							0
23	29.5	13.5	21.5							0
24	26.0	14.0	20.0							0
25	21.5	13.0	17.3							0
26	23.5	11.5	17.5							0
27	26.0	13.5	19.8							0
28	28.5	13.0	20.8	1.2		1.2				0
29	22.0	13.0	17.5							0
30	22.5	11.0	16.8							0
31	24.0	9.5	16.8							0
TOTAL	699.5	378.5		35.8	0.0	35.8	0	0	0	
MEAN	22.6	12.2	17.4							

MONTHLY MAXIMUM TEMPERATURE WAS 29.5 ON DAY 23

MONTHLY MINIMUM TEMPERATURE WAS 9.5 ON DAY 21 22 31

HIGHEST RAINFALL WAS 24.8 ON DAY 15

NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 6

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 36.0

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR SEPTEMBER 1988

B.C. REGIONAL ID : 7140J
AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H	Z R	H A	SNOW GND
1	28.5	12.5	20.5							0
2	33.0	14.5	23.8							0
3	32.0	14.5	23.3							0
4	26.0	15.0	20.5							0
5	22.0	11.5	16.8							0
6	18.0	12.0	15.0	.6		.6				0
7	20.5	13.0	16.8							0
8	18.5	9.5	14.0							0
9	19.5	9.0	14.3							0
10	19.5	6.0	12.8							0
11	22.5	6.0	14.3							0
12	26.5	6.0	16.3							0
13	26.0	11.5	18.8							0
14	26.5	12.0	19.3							0
15	13.5	11.0	12.3	1.0		1.0				0
16	17.5	10.0	13.8	2.0		2.0				0
17	17.5	5.5	11.5							0
18	16.5	6.0	11.3	27.4		27.4				0
19	16.0	6.0	11.0	.8		.8				0
20	18.0	8.0	13.0							0
21	16.5	8.5	12.5							0
22	17.0	6.5	11.8	19.4		19.4				0
23	12.5	9.0	10.8	7.8		7.8				0
24	14.0	6.0	10.0	20.7		20.7				0
25	14.5	5.5	10.0	1.0		1.0				0
26	13.0	6.0	9.5	13.0		13.0				0
27	17.0	9.5	13.3	1.5		1.5				0
28	16.0	10.0	13.0							0
29	23.0	8.0	15.5							0
30	19.0	9.5	14.3							0
TOTAL	600.5	278.0		95.2	0.0	95.2	0	0	0	
MEAN	20.0	9.3	14.7							

MONTHLY MAXIMUM TEMPERATURE WAS 33.0 ON DAY 2
MONTHLY MINIMUM TEMPERATURE WAS 5.5 ON DAY 17 25
HIGHEST RAINFALL WAS 27.4 ON DAY 18
NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 11

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 118.1

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR OCTOBER 1988

B.C. REGIONAL ID : 7140J
AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H	Z R	H A	SNOW GND
1	24.0	9.0	16.5							0
2	24.5	9.0	16.8							0
3	14.5	9.5	12.0	1.0		1.0				0
4	15.0	11.0	13.0	3.0		3.0				0
5	16.5	13.0	14.8							0
6	15.0	10.0	12.5	.5		.5				0
7	17.5	11.5	14.5							0
8	16.5	8.0	12.3							0
9	17.5	6.0	11.8							0
10	21.5	6.0	13.8							0
11	14.0	6.0	10.0	2.0		2.0				0
12	18.0	12.0	15.0	6.0		6.0				0
13	14.0	12.0	13.0	22.5		22.5				0
14	13.0	11.0	12.0	22.2		22.2				0
15	14.0	10.5	12.3	25.0		25.0				0
16	14.0	10.5	12.3							0
17	13.0	4.5	8.8							0
18	9.5	5.5	7.5	11.0		11.0				0
19	12.0	8.5	10.3	1.0		1.0				0
20	13.0	10.5	11.8							0
21	13.5	8.0	10.8	2.0		2.0				0
22	13.5	7.0	10.3							0
23	13.5	6.5	10.0	5.6		5.6				0
24	14.0	7.0	10.5							0
25	11.0	6.0	8.5							0
26	12.0	7.0	9.5							0
27	11.5	1.5	6.5							0
28	12.0	1.0	6.5							0
29	11.5	3.0	7.3	9.8		9.8				0
30	12.0	8.0	10.0	26.0		26.0				0
31	14.0	10.0E	12.0	12.0		12.0				0
TOTAL	455.5	249.0		149.6	0.0	149.6	0	0	0	
MEAN	14.7	8.0	11.4							

MONTHLY MAXIMUM TEMPERATURE WAS 24.5 ON DAY 2
MONTHLY MINIMUM TEMPERATURE WAS 1.0 ON DAY 28
HIGHEST RAINFALL WAS 26.0 ON DAY 30
NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 15

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 205.1

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR NOVEMBER 1988

B.C. REGIONAL ID : 7140J

AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	THRA	SNOW GND
1	13.0	11.0	12.0	12.0		12.0		0
2	13.0	8.0	10.5	21.0		21.0		0
3	11.0	8.0	9.5	20.0		20.0		0
4	9.0	8.0	8.5	23.8		23.8		0
5	13.0	8.0	10.5	41.7		41.7		0
6	13.5	7.0	10.3	1.0		1.0		0
7	10.0	3.5	6.8	12.0		12.0		0
8	6.5	5.0	5.8	8.0		8.0		0
9	10.0	4.5	7.3	17.0		17.0		0
10	9.0	6.0	7.5	7.0		7.0		0
11	9.5	3.5	6.5	11.9		11.9		0
12	10.0	3.0	6.5	5.0		5.0		0
13	12.0	3.0	7.5					0
14	8.0	0.0	4.0	2.0		2.0		0
15	7.0	2.5	4.8	3.0		3.0		0
16	8.5	5.5	7.0	2.0		2.0		0
17	8.5	5.0	6.8	1.0		1.0		0
18	7.5	1.0	4.3	27.0		27.0		0
19	7.5	4.0	5.8	17.0		17.0		0
20	10.0	4.5	7.3	16.0		16.0		0
21	10.0	4.0	7.0	7.0		7.0		0
22	10.0	4.0	7.0	4.0		4.0		0
23	8.5	5.0	6.8	7.0	T	7.0		0
24	5.5	.5	3.0	7.0E	2.0E	9.0E		M
25	5.0	0.0	2.5	8.0	T	8.0		M
26	5.0	-.5	2.3	11.0		11.0		0
27	8.0	0.0	4.0	8.0		8.0		0
28	8.0	N	M	2.0		2.0		0
29	8.0	5.0	6.5					0
30	10.0	2.0	6.0					0
TOTAL	274.5	121.0		302.4	2.0	304.4	0 0 0	
MEAN	9.1	4.2	6.7					

MONTHLY MAXIMUM TEMPERATURE WAS 13.5 ON DAY 6
MONTHLY MINIMUM TEMPERATURE WAS -.5 ON DAY 26
HIGHEST RAINFALL WAS 41.7 ON DAY 5
HIGHEST SNOWFALL WAS 2.0 ON DAY 24
MAX TOTAL PRECIP WAS 41.7 ON DAY 5
NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 27
NUMBER OF DAYS WITH MEASUREABLE RAINFALL IS 27
NUMBER OF DAYS WITH MEASUREABLE SNOWFALL IS 1

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 339.1

HEATING DEGREE-DAYS ESTIMATED - MEAN MONTHLY TEMP USED FOR 1 MISSING TEMPS

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR DECEMBER 1988

B.C. REGIONAL ID : 7140J

AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H	Z R	H A	SNOW GND
1	13.0	4.5	8.8	13.0		13.0				0
2	7.0	4.0	5.5	1.0		1.0				0
3	12.0	2.5	7.3							0
4	12.0	2.5	7.3	8.6		8.6				0
5	9.0	3.0	6.0	16.8		16.8				0
6	9.0	7.0	8.0	13.0		13.0				0
7	8.5	6.0	7.3	5.2		5.2				0
8	8.5	6.5	7.5	11.0		11.0				0
9	9.0	7.0	8.0	5.5		5.5				0
10	9.5	6.0	7.8	4.0		4.0				0
11	9.5	5.5	7.5	25.0		25.0				0
12	11.0	5.5	8.3	28.0		28.0				0
13	7.5	5.0	6.3	1.0		1.0				0
14	6.5	-2.5	2.0							0
15	5.0	-3.0	1.0							0
16	6.5	-2.5	2.0							0
17	8.0	-2.0	3.0							0
18	8.0	-2.0	3.0	13.0		13.0				0
19	4.0	-1.5	1.3	3.0		3.0				0
20	4.0	1.5	2.8	11.0		11.0				0
21	6.0	0.0	3.0	1.0		1.0				0
22	5.0	2.5	3.8	9.4		9.4				0
23	6.0	1.0	3.5	3.2		3.2				0
24	6.0	1.0	3.5	.8		.8				0
25	6.0	-2.0	2.0							0
26	1.0	-3.0	-1.0		2.0	2.0				0
27	4.0	-2.0	1.0							M
28	3.5	-2.5	.5	11.5E	5.5E	17.0E				M
29	9.0	0.0	4.5	44.0		44.0				M
30	8.5	2.0	5.3	3.4		3.4				0
31	6.0	2.0	4.0							0
TOTAL	228.5	52.0		232.4	7.5	239.9	0	0	0	
MEAN	7.4	1.7	4.6							

MONTHLY MAXIMUM TEMPERATURE WAS 13.0 ON DAY 1

MONTHLY MINIMUM TEMPERATURE WAS -3.0 ON DAY 15 26

HIGHEST RAINFALL WAS 44.0 ON DAY 29

HIGHEST SNOWFALL WAS 5.5 ON DAY 28

MAX TOTAL PRECIP WAS 44.0 ON DAY 29

NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 23

NUMBER OF DAYS WITH MEASUREABLE RAINFALL IS 22

NUMBER OF DAYS WITH MEASUREABLE SNOWFALL IS 2

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 417.2

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR JANUARY 1989

B.C. REGIONAL ID : 7140J

AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H R	Z H R	A	SNOW GND
1	6.0	0.0	3.0	19.2	T	19.2				0
2	9.5	-1.0	4.3	13.8		13.8				0
3	10.0	4.5	7.3	5.2		5.2				0
4	6.5	3.0	4.8	1.0		1.0				0
5	5.0	2.0	3.5							0
6	3.0	-.5	1.3	.8	T	.8				0
7	3.0	-4.0	-.5		.5	.5				0
8	4.0	-4.0	.0		16.5	16.5				M
9	4.0	.5	2.3	12.6		12.6				M
10	3.0	-2.5	.3							M
11	5.5	-1.0	2.3	6.0		6.0				M
12	7.5	1.0	4.3	15.0		15.0				M
13	5.0	1.0	3.0	1.0E	1.0E	2.0E				M
14	3.0	-1.0	1.0	10.0E	13.0	23.0E				M
15	7.0	-1.0	3.0	20.0E		20.0E				M
16	8.5	5.0	6.8	4.6		4.6				M
17	8.5	5.0	6.8	32.0		32.0				M
18	8.0	2.0	5.0							0
19	8.0	.5	4.3							0
20	6.0	2.0	4.0	10.0		10.0				0
21	7.0	0.0	3.5	2.0E	.5	2.5E				0
22	7.0	-1.0	3.0							0
23	2.0	-4.5	-1.3							0
24	2.0	-2.0	.0	.5	T	.5				0
25	4.0	0.0	2.0	8.2		8.2				0
26	6.0	1.0	3.5	3.2		3.2				0
27	7.5	2.0	4.8							0
28	7.0	1.0	4.0	1.4		1.4				0
29	10.5	5.0	7.8	2.6		2.6				0
30	10.0	6.5	8.3	6.8		6.8				0
31	3.0	-1.5	.8		T	T				0
TOTAL	187.0	18.0		175.9	31.5	207.4	0	0	0	
MEAN	6.0	.6	3.3							

MONTHLY MAXIMUM TEMPERATURE WAS 10.5 ON DAY 29
MONTHLY MINIMUM TEMPERATURE WAS -4.5 ON DAY 23
HIGHEST RAINFALL WAS 32.0 ON DAY 17
HIGHEST SNOWFALL WAS 16.5 ON DAY 8
MAX TOTAL PRECIP WAS 32.0 ON DAY 17
NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 23
NUMBER OF DAYS WITH MEASUREABLE RAINFALL IS 21
NUMBER OF DAYS WITH MEASUREABLE SNOWFALL IS 5

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 454.8

SURREY MUNICIPAL HALL CLIMATE SUMMARY FOR FEBRUARY 1989

B.C. REGIONAL ID : 7140J

AES HEADQUARTERS ID : 1107876

DAY	TEMPERATURE MAX	MIN	MEAN	TOTAL RAIN	SNOW	PCP	T H R A	SNOW GND
1	-10.5	-12.5	-11.5		T	T		M
2	-9.0	-14.0	-11.5					M
3	-4.0	-12.0	-8.0					M
4	0.0	-9.0	-4.5					0
5	4.5	-9.0	-2.3					0
6	3.5	-9.0	-2.8					0
7	4.5	-7.0	-1.3					0
8	4.5	-6.5	-1.0					0
9	8.0	-3.5	2.3					0
10	9.0	-4.0	2.5					0
11	M	Y	M					0
12	M	-4.0	M					0
13	6.0	-4.5	.8					0
14	5.0	-4.0	.5					0
15	6.5	-4.0	1.3		3.0E	3.0E		0
16	3.0	0.0	1.5		4.0	4.0		M
17	3.0	-1.0	1.0	.5	17.0	17.5		M
18	3.5	-1.0	1.3	7.5		7.5		M
19	4.0	-1.0	1.5	9.0		9.0		M
20	7.5	1.0	4.3					M
21	7.0	3.0	5.0	11.4		11.4		M
22	10.0	5.0	7.5	11.4		11.4		0
23	10.0	2.0	6.0	8.0		8.0		0
24	9.0	Y	M					0
25	9.0	-4.0	2.5					0
26	9.0	-5.0	2.0	.4		.4		0
27	5.0	1.0	3.0					0
28	4.5	0.0	2.3		2.0	2.0		M
TOTAL	112.5	-103.0		48.2	26.0	74.2	0 0 0	
MEAN	4.3	-4.0	.2					

MONTHLY MAXIMUM TEMPERATURE WAS 10.0 ON DAY 22 23

MONTHLY MINIMUM TEMPERATURE WAS -14.0 ON DAY 2

HIGHEST RAINFALL WAS 11.4 ON DAY 22

HIGHEST SNOWFALL WAS 17.0 ON DAY 17

MAX TOTAL PRECIP WAS 17.5 ON DAY 17

NUMBER OF DAYS WITH MEASUREABLE PRECIPITATION IS 10

NUMBER OF DAYS WITH MEASUREABLE RAINFALL IS 7

NUMBER OF DAYS WITH MEASUREABLE SNOWFALL IS 4

NUMBER OF HEATING DEGREE-DAYS FOR MONTH IS 501.2

CLIMATE SUMMARY FOR MARCH 1 - 16, 1989
AS RECORDED AT SURREY MUNICIPAL HALL[1]

DAY	TEMPERATURE (C)		PRECIPITATION
	MAX	MIN	(mm)
1	-1.0	-6.0	0.2
2	4.0	-4.5	-
3	5.0	-7.5	-
4	5.0	-6.0	-
5	8.0	-5.0	3.6
6	9.5	-5.0	16.8
7	9.0	0.0	1.4
8	9.5	2.0	0.4
9	10.0	2.0	2.2
10	11.5	5.0	9.0
11	13.0	5.0	11.0
12	13.0	3.0	8.8
13	13.0	2.0	6.4
14	13.0	2.5	0.6
15	13.0	-1.0	3.2
16	13.0	2.0	-

[1]Preliminary data; values unconfirmed.