Fraser River Estuary Study
Water Quality

Impact of Landfills

J. W. Atwater, P. Eng.
Environmental Protection Service
Environment Canada

Vancouver, British Columbia
April, 1980

Government of
Canada

Province of
British Columbia

Canadian Cataloguing in Publication Data

Atwater, J. W.
Impact of landfills.

(Fraser River estuary study, water quality)

Background report to the Fraser River estuary study of the Fraser River Estuary Study Steering Committee.
Bibliography: p.
ISBN 0-7719-8353-0

1. Sanitary landfills - Environmental aspects - British Columbia - Lower Mainland region. 2. Leaching. 3. Water quality - British Columbia - Fraser River. 4. Wood waste - British Columbia - Lower Mainland region. I. Fraser River Estuary Study Steering Committee. Water Quality Work Group. II. British Columbia. III. Canada. IV. Title. V. Series.

TD795.7A88 628.4'4564 C80-092101-1

PREFACE

The Fraser River Estuary Study was set up by the Federal and Provincial Governments to develop a management plan for the area.

The area under study is the Fraser River downstream from Kanaka Creek to Roberts Bank and Sturgeon Bank. The Banks are included between Point Grey and the U.S. Border. Boundary Bay and Semiahmoo Bay are also included but Burrard Inlet is not in the study area.

The study examined land use, recreation, habitat and water quality, and reports were issued on each of these subjects.

Since the water quality report was preliminary, a more detailed analysis of the information was undertaken by members of the Water Quality Work Group. As a result, eleven background technical reports, of which this report is one, are being published. The background reports are entitled as follows:

- Municipal effluents.
- Industrial effluents.
- Storm water discharges.
- Impact of landfills.
- Acute toxicity of effluents.
- Trace organic constituents in discharges.
- Toxic organic contaminants.
- Water chemistry; 1970-1978.
- Microbial water quality; 1970-1977.
- Aquatic biota and sediments.
- Boundary Bay.

Each of the background reports contains conclusions and recommendations based on the technical findings in the report. The recommendations do not necessarily reflect the policy of government agencies funding the work. Copies of these reports will be available at all main branches of the public libraries in the Lower Mainland.

Five auxiliary reports are also being published in further support of the study. These cover the following subjects:

- Site registry of storm water outfalls.
- Dry weather storm sewer discharges.
- Data report on water quality.
- Survey of fecal coliforms in 1978.
- Survey of dissolved oxygen in 1978.

Copies of these reports will be available from the Ministry of Environment, Parliament Buildings, Victoria, British Columbia.

To bring this work together the Water Quality Work Group has published a summary report. This document summarizes the background reports, analyzes their main findings and presents final recommendations. Some of the recommendations from the background reports may be omitted or modified in the summary report, due to the effect of integrating conclusions on related topics. Copies of the summary report are in public libraries, and extra copies will be available from the Ministry of Environment in Victoria to interested parties.

ABSTRACT

Presented in this report is a summary of available information on landfills located within the area contiguous to the Fraser River Estuary.

The information is reviewed within the context of leachate generation and pollutant loads associated with the leachate, present and future, and the potential impact of the pollutants on the Fraser River Estuary.

Four classes of landfills are dealt with in the chapters: large active municipal landfills, large closed municipal landfills, wood waste landfills, and small municipal and miscellaneous landfills.

Landfilling was found to be the prime method of solid waste disposal, with only about 10% of the wood waste currently going to incineration. Estimates of pollutant loads resulting from the first three landfill classes are made and presented in the report.

The landfills were seen to be significant sources of organic material, ammonia, and total solids, although not major sources of trace metals. Leachate from wood waste landfills was seen to cause significant degradation of water quality where directed into small tributaries and drainage ditches flowing into the Fraser River.

ACKNOWLEDGEMENTS

The author wishes to acknowledge the assistance of the many individuals who contributed to the preparation of this report. Much of the Introduction and Large Active Landfill chapters was written from material prepared by P.W. Soper and R.D. Cameron as consultants to the Environmental Protection Service. Guy Hebert, George Derkson and Bob Hallam of EPS provided both text and information used in the preparation of the chapters on Wood Waste Landfills and Small Municipal and Miscellaneous Refuse Dumps. Neil Holman and Otto Langer took the photographs. K.H. Kupka, Director Environmental Services Branch, provided advice, and co-ordinated the editing and publication of this report.

Many thanks go to the owners and operators of the various landfills who freely provided information on their sites and who took the time to review the text: Phil Herring, City of Vancouver; Stan Vernon and Bill Redman, GVRD; Rick Pierce, Fraser River Harbour Commission; John Troubridge, District of Surrey; Jim Leeder, Leeder Landfill; Fred Milligan, Crown Zellerbach; and the Staffs of the Municipality of Burnaby and District of Maple River.

The Waste Management Branch provided monitoring data for many of the landfills, as well as background information on these sites. This material was invaluable.

Finally, the author wishes to thank the members of the agencies who made up the Water Quality Group for their efforts in reviewing the text and providing comments. The efforts of Dave Douglas of the Waste Management Branch were particularly appreciated.

TABLE OF CONTENTS

		Page
PREFACE		
ABSTRACT		i
ACKNOWLEDGEMENTS		ii
TABLE OF CONTENTS		iii
	List of Figures	viii
	List of Tables	xi
	List of Plates	xiv
SUMMARY		xv
CONCLUSIONS		xviii
RECOMMENDATIONS		xxii
1	INTRODUCTION	1
1.1	Purpose and Scope	1
1.2	Study Area	2
1.3	Physiography of the Lower Fraser River and Estuary	2
1.4	Overview of Landfilling in the Lower Fraser River and Estuary	4
1.4.1	Present and Future Quantities of Waste	5
1.4.2	Landfills	7
1.5	Current Regulations	9
2	LEACHATE CHARACTERISTICS	12
2.1	Introduction	12
2.2	Leachate Generation	12
2.2.1	General Mechanisms of Leachate Production and Migration	15
2.2.2	General Refuse	19
2.2.3	Wood Waste	26
2.2.4	Construction Debris	30
2.2.5	Incinerator Residue	31
2.3	Attenuation and Containment Sites	33
2.4	Leachate Treatability	34

		Page
3	LARGE MUNICIPAL LANDFILLS CURRENTLY ACTIVE	36
3.1	Burns Bog - Vancouver Delta Landfill	36
3.1.1	General	36
3.1.2	Physical Description	36
3.1.3	Operation	40
3.1.4	Leachate	41
3.1.5	Impact on Water Quality	48
3.1.5.1	Point of egress	48
3.1.5.2	Defined receiving water, past and present	50
3.1.5.3	Future sewer connection	53
3.1.5.4	Loading estimates	54
3.1.6	Intended Use	54
3.2	Richmond Landfill	56
3.2.1	General	56
3.2.2	Physical Description	56
3.2.3	Operation	63
3.2.4	Leachate	65
3.2.5	Impact on Water Quality	70
3.2.5.1	Point of egress	70
3.2.5.2	Defined receiving water	71
3.2.5.3	Fraser Estuary	75
3.2.5.4	Leachate loadings	76
3.2.6	Impact on Other Environments	76
3.2.7	Intended Use	78
3.3	Port Mann - Surrey Landfill	79
3.3.1	General	79
3.3.2	Physical Description	79
3.3.3	Drainage	82
3.3.4	Operation	86
3.3.5	Leachate	86
3.3.6	Loading	91
3.3.7	Impact on Water Quality	91
3.3.7.1	Point of egress	91

		Page
3.3.7.2	Impact on biota at point of egress	91
3.3.8	Sewer Connection	94
3.3.9	Intended Use	94
3.4	GVRD - Coquitlam Landfill (Braid Street)	94
3.4.1	General	94
3.4.2	Physical Description	95
3.4.3	Operation	95
3.4.4	Leachate	99
3.4.5	Impact on Water Quality	107
3.4.5.1	Point of egress and defined receiving environment	107
3.4.6	Intended Use	107
3.5	Leeder Landfill	108
3.5.1	General	108
3.5.2	Physical Description	108
3.5.3	Operation	113
3.5.4	Leachate	114
3.5.5	Impact on Water Quality	115
3.5.5.1	Point of egress	115
3.5.6	Intended Use	122
4	LARGE MUNICIPAL LANDFILLS (CLOSED)	123
4.1	Kerr Road Landfill	123
4.1.1	General	123
4.1.2	Physical Description	123
4.1.3	Operations	128
4.1.4	Leachate	128
4.1.5	Impact on Water Quality	134
4.1.6	Intended Use	134
4.2	Terra Nova Landfill	134
4.2.1	General	134
4.2.2	Physical Description	134
4.2.3	Operation	137
4.2.4	Leachate	140

Page

4.2.5	Impact on Water Quality	146
4.2.6	Intended Use	150
4.3	Stride Avenue Landfill	150
4.3.1	General	150
4.3.2	Physical Description	150
4.3.3	Operation	154
4.3.4	Leachate	154
4.3.5	Impact on Water Quality	155
4.3.6	Intended Use	159
5	WOOD WASTE LANDFILL	160
5.1	Introduction	160
5.2	Wood Waste Disposal	160
5.3	Wood Waste Disposal Sites	161
5.4	Wood Waste Leachates	161
5.5	Impact on Water Quality	165
5.5.1	Fraser River	165
6	SMALL MUNICIPAL LANDFILLS AND MISCELLANEOUS DUMPS	167
6.1	Introduction	167
6.2	Small Municipal Landfills	167
6.2.1	Introduction	167
6.2.2	Maple Ridge Landfill	167
6.2.3	Johnston Road Landfill - District of Surrey	176
6.2.4	Bear Creek Landfill - District of Surrey	179
6.2.5	Elgin Landfill - District of Surrey	181
6.2.6	Port Coquitlam Landfill	181
6.2.7	Sperling Avenue Landfill - Corporation of Burnaby	181
6.2.8	Semiahmoo Bay and 24th Avenue Landfills - District of Surrey	182
6.3	Miscellaneous Dumps	182
6.3.1	Introduction	182
6.3.2	Permitted Industrial Disposal Sites	183

		Page
6.3.3	Refuse for Land Development	188
6.3.4	Convenience Dumping	188
6.3.5	Impact of Miscellaneous Refuse Dumps	190
7	DISCUSSION	192
7.1	Large Municipal Landfills - Active	192
7.1.1	pH	197
7.1.2	Heavy Metals	198
7.1.3	COD - Chemical Oxygen Demand	200
7.1.4	Ammonia and Alkalinity	201
7.1.5	Toxicity	201
7.2	Large Municipal Landfills - Closed	202
7.2.1	Loadings	203
7.2.2	Toxicity	207
7.3	Wood Waste Landfills	207
7.4	Small Municipal Landfills and Miscellaneous Refuse Dumps	209
7.5	Impact on the Lower Fraser River and Estuary	209
7.5.1	Large Municipal Landfills - Active	209
7.5.2	Large Municipal Landfills - Closed	214
7.5.3	Wood Waste Landfills	215
7.5.4	Small Municipal Landfills and Miscellaneous Refuse Dumps	220
REFERENCES		221
APPENDICES		
APPENDIX A	Leachate Treatability	227
APPENDIX B	Monitoring Data, Burns Bog Landfill	240
APPENDIX C	Monitoring Data, Richmond Landfill	246
APPENDIX D	Wood Waste, EPS Aerial Survey 1977	256
APPENDIX E	Location Maps and Site Plans, Closed Small Municipal Landfills	264
APPENDIX F	Leachate Toxicity, Large Active and Wood Waste Landfills	277
APPENDIX G	Excerpts from "The Aqueous Chlorination of Organic Compounds: Chemical Reactivity and Effects on Environmental Quality."	281
APPENDIX H	Variance in Biological Sensitivity to Bioassay	284

LIST OF FIGURES

Figure		Page
1.1	STUDY AREA AND LARGE LANDFILL LOCATIONS - FRASER RIVER ESTUARY	3
2.1	LEACHATE BOD5 VERSUS TIME FOR LANDFILL SPRINGS	21
2.2	LEACHATE TOTAL SOLIDS VERSUS TIME FOR LANDFILL SPRINGS	23
2.3	CONSTITUENT EXTRACTION FOR REFUSE RECEIVING 45 INCHES OF RAIN PER YEAR - UBC LYSIMETERS	24
3.1.1	BURNS BOG LANDFILL SITE PLAN	37
3.1.2	GEOLOGICAL FENCE DIAGRAM - CITY OF VANCOUVER - BURNS BOG	39
3.1.3	BURNS BOG LANDFILL - DRAINAGE PATTERNS AND MONITORING STATIONS (PRE-1978)	44
3.1.4	BURNS BOG LANDFILL - LEACHATE CONTROL WORKS (POST-1978)	45
3.2.1	RICHMOND LANDFILL SITE PLAN	57
3.2.2	GEOLOGICAL FENCE DIAGRAM - RICHMOND LANDFILL 125 ha SITE	58
3.2.3	GEOLOGICAL CROSS-SECTION - RICHMOND LANDFILL 125 ha SITE	59
3.2.4	RICHMOND LANDFILL - TYPICAL HYDROGRAPHS FOR REFUSE, PEAT, AND SILTY-SAND UNITS	61
3.2.5	RICHMOND LANDFILL DITCH FLOWS	62
3.2.6	RICHMOND LANDFILL - COMPLEX WATER BUDGET INFLOWS AND OUTFLOWS	67
3.3.1	SITE PLAN OF THE ORIGINAL AND PRESENT PROPERTY LINES - PORT MANN LANDFILL - DISTRICT OF SURREY	81
3.3.2	BORE HOLE LOCATIONS - PORT MANN LANDFILL - DISTRICT OF SURREY	83
3.3.3	GEOLOGICAL FENCE - PORT MANN LANDFILL - DISTRICT OF SURREY	84
3.3.4	GEOLOGICAL CROSS-SECTIONS - PORT MANN LANDFILL - DISTRICT OF SURREY	85
3.3.5	DRAINAGE LAYOUT AND MONITORING LOCATIONS - PORT MANN LANDFILL - DISTRICT OF SURREY	87

Figure		Page
3.4.1	GVRD - COQUITLAM LANDFILL LOCATION MAP	97
3.4.2	GVRD - COQUITLAM LANDFILL LEACHATE CONTROL WORKS	98
3.4.3	APPROXIMATE CELL LAYOUT - GVRD - COQUITLAM LANDFILL	100
3.4.4	ALKALINITY CONCENTRATION INCREASE WITH FILL AGE - COQUITLAM LANDFILL	105
3.5.1	LEEDER LANDFILL LOCATION MAP	110
3.5.2	LEEDER LANDFILL SITE PLAN AND MONITORING SITES	111
3.5.3	GEOLOGICAL FENCE DIAGRAM - LEEDER LANDFILL	112
3.5.4	GEOLOGICAL CROSS-SECTION LINE A-A (EAST-WEST) - LEEDER LANDFILL	116
3.5.5	GEOLOGICAL CROSS-SECTION LINE B-B (NORTH-SOUTH) - LEEDER LANDFILL	117
4.1.1	CROSS-SECTIONS - KERR ROAD LANDFILL - CITY OF VANCOUVER	125
4.1.2	GEOLOGICAL CROSS-SECTION - KERR ROAD LANDFILL - CITY OF VANCOUVER	126
4.1.3	ORIGINAL AND PRESENT CONTOURS - KERR ROAD LANDFILL - CITY OF VANCOUVER	127
4.1.4	LEACHATE CONSTITUENT CONCENTRATIONS - KERR ROAD LANDFILL - CITY OF VANCOUVER	129
4.1.5	LEACHATE METAL CONCENTRATIONS - KERR ROAD LANDFILL - CITY OF VANCOUVER	130
4.2.1	GEOLOGICAL CROSS-SECTION - TERRA NOVA LANDFILL	136
4.2.2	TERRA NOVA LANDFILL SITE PLAN	138
4.2.3	FILL POSITIONS - TERRA NOVA LANDFILL	141
4.2.4	PLOTS OF WEIR FLOWS AGAINST TIME - TERRA NOVA LANDFILL	143
4.3.1	LOCATION MAP - STRIDE AVENUE LANDFILL	152
4.3.2	SITE PLAN - STRIDE AVENUE LANDFILL	153
4.3.3	PLOT OF WEIR FLOW AGAINST TIME - STRIDE AVENUE LANDFILL	157
6.2.1	LOCATION MAP SHOWING THE DISTRIBUTION OF SMALL MUNICIPAL LANDFILLS IN THE FRASER RIVER ESTUARY AREA	168
6.2.2	LOCATION MAP - MAPLE RIDGE LANDFILL	171
6.2.3	SITE PLAN - MAPLE RIDGE LANDFILL	172

Figure		Page
6.3.1	AUTHORIZED AND UNAUTHORIZED MISCELLANEOUS REFUSE DISPOSAL SITES IN THE FRASER RIVER DELTA AREA	184
7.1	WOOD WASTE LEACHATE LOCATIONS OF TRIBUTARY DRAINAGE	217

LIST OF TABLES

Table		Page
1.1	ESTIMATED MUNICIPAL COLLECTION REFUSE QUANTITIES - PROJECTIONS TO 1995	6
2.1	SOLID WASTE CLASSIFICATIONS	13
2.2	LEACHATE COMPOSITION - MUNICIPAL REFUSE	20
2.3	LEACHATE TOXICITY IN LYSIMETERS AND SITE NO. 1 AND SITE NO. 2 LANDFILL SPRINGS	25
2.4	FISH TOXICITY AND ANALYTICAL RESULTS - PREDOMINANTLY HEMLOCK WOOD WASTES	28
2.5	RANGE OF LEACHATE CHARACTERISTICS - INCINERATOR RESIDUE	32
2.6	LOS ANGELES WASTE DISPOSAL SITE CLASSIFICATION	35
3.1.1	BURNS BOG LANDFILL - SYNOPSIS OF MONITORING SITES	49
3.1.2	BURNS BOG LANDFILL - MONITORING DATA FROM STATIONS 13 AND 17	51
3.1.3	BURNS BOG LANDFILL - MONITORING DATA FROM STATIONS 14 AND 16	52
3.1.4	BURNS BOG LANDFILL - CONSTITUENT LOADINGS	55
3.2.1	SUMMARY OF LEACHATE MONITORING RESULTS - RICHMOND LANDFILL	68
3.2.2	B.C. RESEARCH COUNCIL 1978 MONITORING PROGRAM - RICHMOND LANDFILL	73
3.2.3	ESTIMATED DAILY LEACHATE LOADING TO THE FRASER RIVER - RICHMOND LANDFILL	77
3.3.1	MONITORING WELL ANALYSES - PORT MANN LANDFILL - DISTRICT OF SURREY	89
3.3.2	ANALYTICAL DATA LEACHATE DITCH - PORT MANN LANDFILL - DISTRICT OF SURREY	90
3.3.3	LEACHATE CONSTITUENT LOADINGS - PORT MANN LANDFILL - DISTRICT OF SURREY	92
3.3.4	SUMMARY OF LOADING DATA - PORT MANN LANDFILL - DISTRICT OF SURREY	93
3.4.1	CUMULATIVE FILL VOLUMES - GVRD - COQUITLAM LANDFILL	101

Table		Page
3.4.2	MONITORING DATA FROM SAMPLING POINT 4 - COQUITLAM LANDFILL	104
3.4.3	ESTIMATED DAILY CONTAMINANT MASS LOADINGS - COQUITLAM LANDFILL	106
3.5.1	MONITORING DATA, SURFACE WATERS, 1972-1978 - LEEDER LANDFILL	119
3.5.2	MONITORING DATA, WELLS, 1972-1978 - LEEDER LANDFILL	120
4.1.1	LEACHATE MONITORING DATA, 1978-1979 - KERR ROAD LANDFILL - CITY OF VANCOUVER	132
4.1.2	LEACHATE CONSTITUENT LOADINGS TO THE FRASER RIVER - KERR ROAD LANDFILL - CITY OF VANCOUVER	133
4.2.1	ANNUAL WASTE QUANTITIES - TERRA NOVA LANDFILL	139
4.2.2	COMPARISON OF LEACHATE CONSTITUENTS - TERRA NOVA LANDFILL	142
4.2.3	CONSTITUENT LOADINGS TO THE FRASER RIVER FROM POPEYE CREEK - TERRA NOVA LANDFILL	145
4.2.4	COMPARISON OF CONSTITUENT CONCENTRATIONS IN MILL CREEK UPSTREAM (SITE 18) AND DOWNSTREAM (SITE 9) - TERRA NOVA LANDFILL	147
4.2.5	COMPARISON OF CONCENTRATIONS OF CONSTITUENTS IN MILL CREEK UPSTREAM OF TERRA NOVA LANDFILL WITH THOSE IN TRIBUTARIES	148
4.2.6	CONSTITUENT LOADINGS FROM TERRA NOVA LANDFILL AND MILL CREEK	149
4.3.1	CONSTITUENT CONCENTRATIONS, 1972/73 AND 1979 - STRIDE AVENUE LANDFILL	156
4.3.2	DAILY AND ANNUAL LEACHATE LOADINGS - STRIDE AVENUE LANDFILL	158
5.1	SUMMARY OF CONSTITUENT DAILY LOADINGS FROM WOOD WASTE LEACHATES	164
5.2	COMPARISONS OF METAL CONCENTRATIONS FROM LEACHATE SPRINGS AT A REFUSE LANDFILL AND A WOOD WASTE DISPOSAL AREA	166

Table		Page
6.2.1	PCB WELL MONITORING DATA - MAPLE RIDGE LANDFILL (July 1975 to February 1978)	174
6.2.2	MAPLE RIDGE LANDFILL, PCB SURFACE FLOW MONITORING DATA (January 1976 to February 1978)	175
6.2.3	MAPLE RIDGE LANDFILL - LEACHATE LOADINGS	177
6.2.4	MONITORING DATA - GVRD 1972 - JOHNSTON ROAD LANDFILL	178
6.2.5	SAMPLE ANALYSES FOR BEAR CREEK LANDFILL	180
6.3.1	INDUSTRIAL REFUSE SITES	185
7.1	LEACHATE CONSTITUENTS, AVERAGE CONCENTRATIONS, LARGE ACTIVE LANDFILLS	194
7.2	LEACHATE CONSTITUENT LOADINGS, LARGE ACTIVE LANDFILLS	195
7.3	COMPARISON OF THE PERCENT YIELD OF LEACHATE CONSTITUENTS (Tonnes of Constituents in Leachate stream/year/tonne of refuse placed per year)	196
7.4	AVERAGE LEACHATE CONSTITUENT CONCENTRATIONS FOR LARGE MUNICIPAL CLOSED SITES	204
7.5	COMPARISON OF LOADING AND PERCENT (%) YIELDS FOR SIX LEACHATE CONSTITUENTS AT THE LARGE MUNICIPAL CLOSED LANDFILL SITES	205
7.6	COMPARISON OF CONSTITUENT LOADINGS IN LEACHATE FROM LARGE MUNICIPAL LANDFILLS, ACTIVE AND CLOSED	206
7.7	SUMMARY OF DAILY LOADINGS FROM WOOD WASTE LEACHATES WITHIN THE STUDY AREA	208

LIST OF PLATES

Plate		Page
3.1.1	AERIAL PHOTO OF BURNS BOG LANDFILL, SHOWING FILL AREA AND COVER DREDGE OPERATION	42
3.2.1	NELSON ROAD OUTLET, RICHMOND LANDFILL SITE LEACHATE PLUME ENTERING THE FRASER RIVER	72
3.3.1	PORT MANN LANDFILL SITE	80
3.4.1	GVRD - COQUITLAM LANDFILL SITE	96
3.5.1	GENERAL VIEW OF THE LEEDER LANDFILL SITE	109
4.1.1	GENERAL VIEW OF THE KERR ROAD LANDFILL SITE	124
4.2.1	GENERAL VIEW OF THE TERRA NOVA LANDFILL SITE	135
4.3.1	GENERAL VIEW OF THE STRIDE AVENUE LANDFILL SITE	151
6.2.1	GENERAL VIEW OF THE MAPLE RIDGE LANDFILL SITE	169
6.3.1	BATCH PLANT AND SOLID WASTE DISPOSAL ALONG THE FRASER RIVER FORESHORE	189
6.3.2	CONVENIENCE REFUSE DUMPING AT GARRY POINT IN RICHMOND	191
7.1	FRASER RIVER FORESHORE HABITAT LOSS DUE TO WOOD WASTE LEACHATE	216

SUMMARY

It is estimated that the leachates coming from all the landfills in the study area contain between 4% and 8.5% of the COD, 6%-7% of the ammonia, 9% of the iron, and about 2% of the zinc entering the lower Fraser each day from all effluent and storm water sources. The principal impacts from these leachates will occur in two areas: increased ammonia levels at the Annacis Island STP from leachate additions which may affect the toxicity of the total Annacis discharge, and degradation of a number of small tributary waters as a result of wood waste leachate.

Landfilling has been, and in all likelihood will continue into the 1980's to be, the principal means of refuse disposal in the Lower Fraser Valley. It is anticipated that in 1980, some 800 000 tonnes of predominantly municipally collected refuse and 427 000 tonnes of wood waste will be landfilled in the five active municipal sites and the innumerable wood waste and small industrial sites. For each tonne of refuse placed it can be anticipated that 5 to 10 kg of solids having a chemical oxygen demand of 7.5 to 15 kg will be leached out, most of which without treatment will find its way into the Fraser River and its tributaries.

It is estimated at this time that the leachate emanating daily from the municipal refuse landfills has a COD of about 5070 kg and contains over 555 kg of ammonia. The large municipal landfills now closed together with the Maple Ridge Landfill contribute about 5% of the total.

The COD of the leachate coming daily from the wood waste fills is estimated to be between 5000 and 14 800 kg/day.

Loadings estimates from the miscellaneous sites, the small closed municipal sites and the convenience dumps have not been attempted; however, while it could be anticipated that the loadings would be small in comparison to the other landfills, site-specific impacts can result.

The impact of municipal refuse leachate on receiving waters in the study area is not well documented and, in most cases, has not been investigated. The leachates are all acutely toxic to fish with 96-hour LC_{50}'s in all cases falling in the range of 24% to 95%. Except for some suggestion of metal accumulation in the sediments adjacent to Richmond Landfill, knowledge of specific effects of the discharges on the Fraser River has to-date been restricted to one of aesthetics.

In recent years efforts have been made to collect the surface leachate discharges from the large municipal landfills and once collected, divert them to the Annacis Island STP, Richmond Landfill excepted. By 1980, leachate could account for more than 16% of the ammonia coming from Annacis Island STP. This may affect the toxicity of the Annacis discharge. At the same time, it is estimated that the organics in the leachate could account for more than 5% of the COD in the Annacis discharge. A large portion of leachate organics is biodegradable; however, there is a fraction which increases with leachate age, that resists biodegradation.

The chlorination of the effluent at the Annacis Island STP will result in the formation of chlorinated organics in trace amounts. The spectrum of chlorinated organics found in the Annacis effluent may increase with the addition of organic precursors in leachate. The total concentration of chlorinated organics would not necessarily increase due to the affinity that the chlorine would have for the increased ammonia concentrations.

The landfill leachates do not on a daily basis contribute large quantities of metals to the Lower Fraser River with the exception of possibly iron (292 kg), manganese (41 kg), zinc and aluminum (5.2 kg). The total measured daily loading of all the other trace metals is less than 2.0 kg. It is suggested that the low metal loadings are a function of the relatively high pH's of 7-8, in the municipal leachates.

Where wood waste leachate has been discharged directly to the Fraser River there has been in some instances discernible effect on the foreshore. However, an overall effect on the receiving water has not been noted. The impact of wood waste leachate on a number of small tributary streams and drainage waters to the Fraser River has been much more apparent and, at its worst, has rendered the waters of two streams acutely toxic. As a general statement, the generation of leachate has not been taken into consideration in the use or placement of wood waste as fill, nor with few exceptions have attempts been made to control wood waste leachates once they are generated. As a consequence, wood waste leachate discharges into existing natural drainage courses.

The problems related to wood waste leachate exist throughout the study area. It is felt that the problems can be better resolved through management guidelines rather than specific leachate control works.

Convenience dumping by companies and individuals is common along many parts of the Fraser River.

The report deals specifically with the impact of leachate on the water quality of the Lower Fraser River and its tributaries. As such, the habitat loss as a function of landfilling some 70 hectares/year with refuse and wood waste has not been discussed. Concerns related to habitat loss are discussed in the "Fraser River Estuary Study, Report of the Habitat Work Group".

CONCLUSIONS

1. Because of net water infiltration, leachate is produced at every one of the assessed landfills. This leachate ultimately finds its way into the Fraser River.

2. It is estimated that the leachates coming from all the landfills in the study area contain between 4% and 8.5% of the COD, 6-7% of the ammonia, 9% of the iron, and about 2% of the zinc entering the Lower Fraser each day from all effluent and storm water sources.

3. In the past, municipal and industrial landfills have been located without due recognition being given in every case to the complexity of the hydrogeological setting. As a result the generation and egress of leachate was not always properly controlled.

4. Estimates were made as to the mass of constituents in the leachates emanating from the majority of the refuse landfill sites. The accuracy of the estimates is qualified, due to the incomplete data base. Generally, there were analyses available on the composition of the various leachates; however, the calculation of mass loadings was hampered by a scarcity of leachate flow data, and therefore the loadings were derived for the most part from estimated flows. At one or two sites not every leachate pathway had been recognized and at these, both the analytical and flow data were minimal.

5. It is estimated that leachate from the municipal refuse landfills accounts for 2% of the COD, 6-7% of the ammonia, 6% of the iron, and about 2% of the zinc entering the Lower Fraser from all the effluent and storm water sources. Ninety-five percent of the leachate is from the large active fills, with the remaining 5% from the Maple Ridge site and the three large closed sites.

6. The total mass of trace metals, such as arsenic, cadmium, copper, chromium, nickel and lead, entering the Fraser in the leachates from the municipal refuse landfills, is estimated to be less than 1 kg/day. No estimate is available as to the mass of persistent organics, such as chlorinated hydrocarbons which might be entering the Fraser in the leachates.

7. The collection and discharge of leachate from municipal refuse landfills to the Annacis Island STP is becoming the dominant leachate handling method in the study area. The leachate from the GVRD Coquitlam Landfill presently goes to Annacis and, by 1980, so will the leachates from the Burns Bog and Port Mann Landfills. At that time, it is estimated that 16% of the ammonia and 5% of the COD in the Annacis discharge will be from leachate. At this time no decision has been made on the leachate handling methods to be employed at Richmond Landfill.

8. It can be expected that the concentration of un-ionized ammonia in the Annacis discharge will increase with diversion of leachate into that system. This increase may affect the toxicity of the Annacis discharge.

9. The chlorination of the effluent at the Annacis Island STP results in the formation of chlorinated organics in trace amounts. The spectrum of chlorinated organics found in the Annacis effluent may increase with the addition of organic precursors in the leachate. The total concentration of chlorinated organics would not necessarily increase due to the affinity that the chlorine would have for the increased ammonia concentrations.

10. The landfilling of wood waste occurs throughout much of the study area and is particularly evident adjacent to the Fraser River.

11. The mass of COD discharged in the leachate from the wood waste landfills is estimated to be 5000 to 14 800 kg/day, between 2% and 5.5% of the total COD load to the Lower Fraser from all the effluents and storm water discharges. Estimates of the mass of constituents in the leachate emanating from the wood waste landfills are based on assumptions as to the amounts of wood waste in place, area of drainage, and percent of leachate material in the wood waste. These mass estimates are at best approximate.

12. As a general statement, the generation of leachate has not been taken into consideration in the use or placement of wood waste as fill, nor with few exceptions, have attempts been made to control wood waste leachates once they are generated.

13. Where wood waste leachate has entered small tributary streams, the assimilative capacity has often been exceeded and in two specific cases the streams have been rendered acutely toxic to rainbow trout.

14. Where wood waste leachate has been discharged directly to the Fraser River there has been, in some instances, a discernible effect on the foreshore. However, an overall effect on the receiving water has not been noted.

15. The concentration of constituents in the leachates from the closed large municipal refuse sites has decreased with time. However, it appears that a plateau level is reached at which concentrations remain relatively constant for some time. These sites had either ceased operation before the permit system was established, or the permit had expired with the closure of the site. As a consequence, the control of these continuing leachate discharges may be difficult to enforce.

16. Eight small municipal landfills (one active, seven closed) were identified, all of which are located on small tributaries. An impact has been noted on Bear Creek; however, any impact from the landfills on the other streams is not known.

17. Industrial refuse disposal sites are generally small in size and contain diverse waste. Monitoring has only been carried out at selected sites.

18. Convenience dumping of refuse by companies and individuals is common along the Lower Fraser River. A wide range of activities is involved extending from individual litterings to the operation of dump areas.

RECOMMENDATIONS

1. Measures must be taken to protect the tributary waters and foreshore of the Fraser River from degradation due to wood waste leachate. These measures should be in the context of an overall reassessment of wood waste management in the study area.

2. The effect of leachate addition on the overall toxicity of the Annacis Island STP effluent discharge should be fully studied. With the decision having been made to divert leachates from the Port Mann and Burns Bog Landfills, the study should assess conditions both before and after the diversions occur. This study should be completed before any steps are taken to initiate further leachate diversion.

3. More emphasis should be placed on quantifying leachate flows, in order to assess accurately overall contaminant loadings. This work should be undertaken in conjunction with leachate analysis. For example, the recently installed leachate control works at the City of Vancouver, Burns Bog Landfill should provide an accurate measure of leachate flow.

4. Further study should be carried out on the apparent unique character of the Fraser River Estuary landfills. The biological-chemical decomposition processes and the resultant contaminant make-up of the leachate are of special interest. Particular emphasis should be placed on identifying organic contaminants and on assessing their impact on the Fraser River Estuary.

5. Assessments of proposed landfill sitings should place more emphasis on the hydrogeologic factors.

6. If the landfilling of wood waste is to remain the dominant disposal method, field studies should be undertaken to determine accurately the constituent loadings coming from wood waste.

7. Leachate control works for sites discharging to municipal sewers, should be designed so that all leachates and contaminated runoffs are collected.

8. Enforcement efforts should be taken to stop the convenience dumping of refuse along the Fraser River.

9. The effect of leachate discharges on municipal sewerage works should be assessed.

1 INTRODUCTION

1.1 Purpose and Scope

The filling of land results in obvious physical changes to the land surface and often the alteration of groundwater and surface waters where the generated leachate is not controlled. The purpose of this report is to identify and detail the past and present landfills that exist on the Lower Fraser and its tributaries, and where possible, quantify the effect that the leachate from these landfills has on the Fraser River Estuary. That effect can be manifested through discharges to tributaries, the river itself, or through sewage treatment plants.

The obvious physical changes of the land surface are not inventoried, condemned or defended in this report, although they are undeniably acknowledged to occur. It is estimated that some 70 hectares of land are covered each year with materials that result in the generation of leachates. Countless other hectares are covered by materials such as clean river sand that do not result in leachates of concern.

Land is filled to provide repositories for wastes, to change surface elevations, or to stabilize soft ground. The net result is an alteration of habitat which is almost always a detriment to one use and a benefit to another.

Landfills can affect water quality and the aquatic biota through the generation of leachate. Leachate is unlike almost any other water pollutant in its variability and dependence on extraneous physical conditions for its eventual characteristics. As a result, considerable attention has been given in this report to describing leachate and the mechanism of leachate generation as well as the current state of leachate treatment.

This report is presented in 7 chapters, plus appendices. Chapter 1, Introduction, outlines the study area and details the scope of landfilling in the Fraser Estuary and the possible trends in terms of the quantities and methods of solid waste disposal for the area. Chapter 2, Leachate Characterization, is presented so that the reader may gain some appreciation as to how leachate is generated, the controlling factors, and the limitations to scientific predictions in this area. A summary on the "state of the art" of leachate treatment and some leachate treatment costs are provided in Appendix A. Chapters 3, 4, 5 and 6 are devoted to the inventory and description of the 4 landfill groups: Large Municipal Landfills - active; Large Municipal Landfills - closed; Wood Waste Landfills; and, Small Municipal Landfills and Miscellaneous Dumps. Additional information on the 4 groups can be found in Appendices B, C and D. Chapter 7, Discussion, provides a summary of the report as well as a forum for reviewing potential impacts of leachate discharges.

1.2 Study Area

Figure 1.1 is a map of the study area. Landfills located in the Vancouver Lower Mainland but excluded from this study include the Premier Road Landfill and the Semple Road Landfill both of which are on land draining into Burrard Inlet. The locations of the largest landfills in the study area are shown on Figure 1.1.

1.3 Physiography of the Lower Fraser River and Estuary

British Columbia's major river, the Fraser River, has a watershed area of 90 000 square miles and an average flow rate of 96 300 cfs. The annual flow which ranges between a low of approximately 28 000 cfs and a maximum in excess of 500 000 cfs, carries an estimated 20 $\times 10^6$ tons/year of sediments. The lower 19 miles of the Fraser River are estuary, adjacent to which is located British Columbia's largest population centre, the Greater Vancouver area.

The Fraser River Delta was formed by the fanning out of river sediments from the upland of New Westminster about 10 000 years ago. At that time, the last Pleistocene ice had disappeared from the Fraser Canyon and local post-glacial rebound was essentially complete. The delta has encroached into the Strait of Georgia at an estimated rate of 450 x 10^6 ft^3/year, and has built deposits of 300 to 700 feet

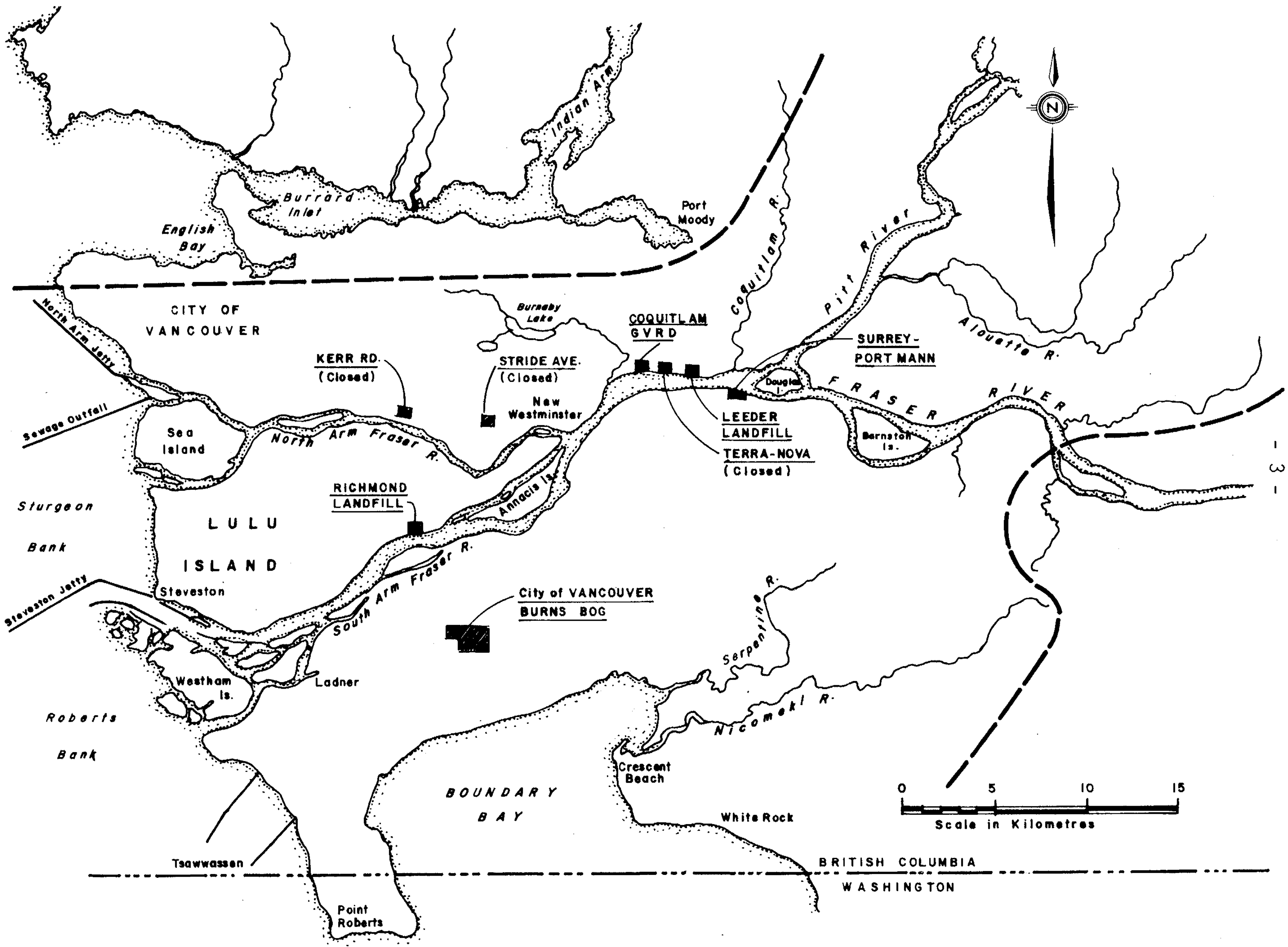


FIGURE 1.1 STUDY AREA AND LANDFILL LOCATIONS - FRASER RIVER ESTUARY

thickness over the Pleistocene sediments (Hoos et al, 1974). Large areas of the deltaic deposits are now overlain by a poorly drained, thick vegetative deposit generally termed peat bogs. Many of the major landfill sites are located on peat bog sites, primarily due to the sites' low economic value and the requirement for considerable filling prior to their development.

1.4 Overview of Landfilling in the Lower Fraser River and Estuary

Landfilling has been almost exclusively the method of solid waste disposal practiced in the Lower Fraser River, even though there are alternative methods of solid waste disposal such as thermal destruction, composting, or the modified landfill practices of shredding and bailing. Solid waste disposal alternatives have been instituted to a limited extent in other parts of B.C. (e.g., Cowichan incinerator); however, such has not been the case in the Lower Fraser region. A major factor in this respect would appear to be the abundance of marginal lands with low economic value and the consequent low landfill operation cost.

Typically in the past, landfill sites have been chosen almost solely on the basis of economics. Generally, marginal land within a reasonable haul distance and an out-of-view location are sought. As a consequence, landfill sites which are selected solely on the basis of economic and social criteria, may be less than ideally sited from an environmental perspective. Landfilling practices have in recent years received much more environmental scrutiny, with the result that environmental design is now becoming a priority consideration in the selection of new sites, and upgrading measures are being required on existing sites. Of particular concern is the egress of leachate either as diffuse or point source discharges.

The Fraser River Estuary, with its large areas of poorly drained low-lying land, has been and continues to be utilized for solid waste landfilling. With the encroachment of urban development on existing landfills and the capacity of sites outside the Fraser Estuary being exhausted, there is a continually increasing volume of refuse out of proportion to the community growth being directed to the Fraser Estuary landfills.

There has, in recent years, been an increasing emphasis on solid waste management in the form of resource and energy recovery, and source reduction, as a result of interest generated by the "energy crisis". To date, the impact of such solid waste management and concomitant disposal alternatives, has had little, if any, effect on the overall solid waste picture in the study area. However, it is expected that the quantities of wood chips and hog fuel that are presently landfilled will decrease because of their increasing utilization and resource value.

1.4.1 Present and Future Quantities of Waste. The solid waste stream is inherently a function of a wide range of variables, not the least of which is the industrial component. Per capita generation rates have been estimated at between 1.6 and 4.5 kg per capita per day (Wilson, 1977).

In their 1976 'Request for Proposals for Solid Waste Disposal Services' by the Greater Vancouver Sewerage and Drainage District (GVSDD), municipally collected* per capita waste generation rates were estimated to range from a low of 0.65 kg/day for North Vancouver City, to a high of 1.08 kg/day for North Vancouver District (GVRD, 1976). Table 1.1 shows extended projections of annual tonnages to 1995, based on the 1974 estimated per capita generation rates and population projections. During 1977, it is estimated that some 770 000 tonnes of solid wastes were landfilled in the five large municipal landfills plus Maple Ridge, some 40% more than anticipated by the GVRD for 1980. Some 470 000 tonnes of wood waste were landfilled that same year mostly in areas outside those fills.

---

* Municipal collection is estimated to be approximately 70% of the total waste going to municipal landfills.

TABLE 1.1 ESTIMATED MUNICIPAL COLLECTION REFUSE QUANTITIES* - PROJECTIONS TO 1995

Municipality	1974 Avg. daily lbs/capita	Estimated Annual Tonnage** 1975	1980	1985	1990	1995
Vancouver	1.98	156 600	164 200	171 700	176 400	180 000
Burnaby	1.75	44 700	55 000	64 100	70 900	77 000
New Westminster	2.03	16 800	18 100	19 600	20 700	22 600
Coquitlam	1.52	17 400	21 800	26 300	30 400	34 400
Port Coquitlam	1.61	6 800	9 300	12 400	15 700	19 300
Richmond	1.75	26 200	34 000	42 500	49 000	54 300
Delta	1.75	20 400	22 400	24 300	27 800	31 300
Surrey	1.75	38 300	51 000	63 800	78 500	93 300
White Rock	1.75	4 200	5 800	6 100	6 200	6 400
Langley City	1.75	1 700	2 000	2 400	2 900	3 400
Total Municipal		333 100	383 600	445 600	478 500	522 000
Private Deliveries Commercial, Industrial @ 40% of Municipal		133 200	153 400	178 200	191 400	208 800
Estimated Overall Total		466 300	537 000	623 800	669 900	730 800

* Based on 1976 GVRD Study for Municipalities discharging to landfills located in Fraser River Estuary Study Area (GVRD, 1976).

** Based on GVRD population forecast.

Residential waste collected by municipal forces can be predicted with a reasonable degree of accuracy. Industrial and commercial wastes, on the other hand, vary significantly with time and location, and it is this factor which likely accounts for the large difference between actual and projected quantities. In the case of wood by-product waste, the day-to-day supply and demand based on market conditions will determine whether certain fractions are a waste or a product.

It is anticipated that the amount of wood waste to be landfilled will reduce to about 320 000 tonnes/annum with planned power boiler expansion on Vancouver Island (Appelby, 1978), and it is conceivable that the remaining material could support a power boiler in the Lower Mainland, and thus be removed entirely as a landfill waste, although the ash would still require landfilling.

Regarding the estimated per capita waste generation, it can be concluded that it is variable and dependent upon the industrial component and economic factors. Any projection of future solid waste quantities must deal with these variables as well as with population projections. Future economic factors which may shift a present waste into a future resource, coupled with improved conservation practices, may partially counteract any future solid waste increases due to population growth.

1.4.2 Landfills. The word 'landfill' is a rather broad and generalized term for any activity involving the filling of land. Webster's Dictionary defines landfill as "a system of trash and garbage disposal in which the waste is buried between layers of earth to build up low-lying land -- called also 'sanitary landfill' and an area built up by landfill". Landfilling can be motivated by the need for disposal of solid waste, the desire to reclaim land, or both. For example, the City of Vancouver Burns Bog Landfill site is a waste disposal operation that will ultimately result in reclaimed land for diversified recreational activities. At the Richmond Landfill, the peat bog is being reclaimed for a future deep sea port and industrial park development by utilizing municipal, commercial, and industrial refuse as fill.

The term 'sanitary landfill' is often used synonomously with refuse landfills. The term acknowledges that the solid waste being handled and disposed can have associated undesirable environmental effects and health hazards. Consequently, operational practices are designed to eliminate or minimize such effects. A sanitary landfill is defined by the American Society of Civil Engineers as: "A method of disposing refuse on land without creating nuisances or hazards to public health or safety, by utilizing the principles of engineering to contain the refuse to the smallest practical area, to reduce it to the smallest practical volume, and to cover it with a layer of earth at the conclusion of each day's operation or more frequent intervals as may be necessary".

It is readily acknowledged that such a coupling of ideal setting and operational practices is difficult to achieve. Nevertheless, most large municipal refuse landfills are engineered and operated in a manner which, for the most part, approaches the idealized and generalized sanitary landfill objectives. In this regard the so-called "open dump" method of the past, in which the refuse was seldom covered and often openly burned, is almost extinct, primarily because of established pollution control regulations precluding such unacceptable disposal practices. It must be recognized that in the past the principal assessment criteria of landfill operations have often been directed towards the operational factors such as face size and covering, which do not necessarily address the more subtle long-term environmental considerations. Many landfill operations, because of future use, structural goals and/or environmental reasons, handle only restricted solid waste materials such as wood by-product wastes, demolition materials, and granular fill materials. Such landfilling activities, although generally devoid of the health implications, can have associated environmental impacts. Of particular concern in this regard are the wood waste dumps.

1.5 Current Regulations

Control of pollution from landfill leachate comes under the Provincial Pollution Control Act, with specific objectives outlined in the Pollution Control Objectives for municipal-type waste discharges in British Columbia (1975).

In concept, these Objectives recognize that refuse leachate presents a potential threat to receiving waters and although the assimilative capacity of the environment may be used within limits to protect against the development of unacceptable conditions, it is further recognized that some receiving areas may have already been used beyond acceptable limits.

The Objectives set out a series of parameters for the maintenance of receiving water quality which are applicable outside a defined, initial dilution zone. Leachate from existing landfills, where the concentration of pollutants exceeds acceptable limits, are to be up-graded in accordance with the Objectives. For new landfills, Location and Control Objectives are defined which, under good waste management and operating practices, are considered to normally prevent any unacceptable changes occurring in the receiving environment. However, where site-specific concerns persist, the Director of the Waste Management Branch may require an environmental assessment study to be undertaken with the details of the study program being subject to his approval.

With regard to ground water quality, the Objectives state that initial dilution zones cannot be specifically defined. However, in cases where leachate causes degradation of groundwater, or surfaces beyond the Permittee's property to cause nuisance or pollution, suitable corrective control techniques will be required. Further, where leachate control or treatment is necessary, or where pollution from leachate is suspected or anticipated, the Permittee may be required to perform sampling and monitoring of the leachate, although the Director will normally implement any receiving area monitoring.

Wood waste leachates are covered under Pollution Control Objectives for the Forest Products Industry of British Columbia. With an approach similar to that adopted for municipal-type leachates, these Objectives specify receiving water quality control such that a negligible increase over background is allowable, outside the initial dilution zone. Although definitive objectives have not yet been formulated for wood wastes leachates, the Objectives refer to interim regulations being devised and implemented at the discretion of the Director of Waste Management, thereby allowing site-specific assessment on the need for leachate control or treatment.

The federal government, under two specific sections of the Fisheries Act, has the reponsibility to ensure protection of Canada's fisheries against pollution and habitat destruction.

The Environmental Protection Service (Department of Environment) administers Section 33(2) of the Act which reads as follows:

> No person shall deposit or permit the deposit of a deleterious substance of any type in water frequented by fish or in any place under any conditions where such deleterious substance or any other deleterious substance that results from the deposit of such deleterious substance may enter any such water.

The Fisheries and Marine Service (Department of Fisheries & Oceans) administers Section 31(1) of the Act which reads as follows:

> No person shall carry on any work or undertaking that results in the harmful alteration, disruption or destruction of fish habitat.

Under the Act, "fish"includes, fish, shellfish, crustaceans, marine animals and the eggs, spawn, spot and juvenile stages of the above. Fish habitat includes spawning grounds and nursery, rearing, food supply and migration areas on which fish depend in order to carry out their life process. Deleterious is defined as follows:

> Any substance that, if added to any water, would degrade or alter or form part of a process of degradation or alteration of the quality of that water so that it is rendered or is likely to be rendered deleterious to fish or fish habitat or to the use by man of fish that frequent that water.

Landfills, whether industrial or domestic, can contravene the Fisheries Act through the generation of any leachate that is deleterious to fish providing the leachate may be deposited in any fish frequented water. A deleterious leachate is one that is toxic to fish or contains substances that will degrade water for fish.

The use of wastes to fill in sloughs, marshes, foreshore areas, etc., can also contravene the Act by means of harmfully altering fish habitat. Since many low-lying areas near water bodies can be considered habitat, the discharge of wastes to these areas can cause conflicts.

In practise, the federal government administers its pollution concerns through the provincial Waste Management Branch (WMB) permit system. The former refer their specific concerns on potential pollution problems as they relate to fish to the WMB who, in turn, in theory, incorporate those concerns prior to issuing a pollution control permit. However, the federal government can unilaterally take action against the pollution of waters frequented by fish. The federal government is responsible for taking action against any activity that is harmful to fish habitat. In most instances, an offer to consult with the provincial authorities is made before taking action.

The federal government, under the Migratory Birds Convention Act is enpowered to maintain adequate habitat for migratory birds. The Canadian Wildlife Service (Department of Environment) administers this section of the Act and is part of the WMB referral system. Landfills by their physical nature and leachate generation have the potential to destroy or reduce the quality of bird habitat.

2 LEACHATE CHARACTERISTICS

2.1 Introduction

This generalized discussion provides the background information as to how leachate is generated, what are the controlling factors, and what are typical leachates.

As the principal impetus of this report concerns the assessment and resultant impact of leachate discharges, it should be recognized that leachates from landfills are a unique source of water pollution. Most sources of water pollution emanate from specific industries or municipalities and, as such, are readily quantifiable as to their contaminant make-up, flows, etc. By contrast, refuse landfills receive the full spectrum of solid waste residues which, when emanating from a highly developed and industrialized metropolitan area, are complex and difficult to characterize. The leachates from those landfills reflect that spectrum. Table 2.1, an outline of waste classifications, shows the range of solid wastes that are landfilled.

Once landfilled, this multiplicity of solid wastes can undergo a complex mix of biological, physical, and chemical decomposition processes and interactions. The resultant leachate outflow is a product of these reactions and is dependent on the nature of the site water balance and operational factors. Typically, the leachate outflow is a complex and diffuse discharge to groundwater and/or surface water, and the rate and character of the flow are functions of the many site variables. Unlike most other sources of water pollution, the discharge and egress of leachate does not cease when the landfilling operation ceases, but rather can continue for many years as the fill contents react and decompose.

2.2 Leachate Generation

When a continuous net water inflow occurs at a landfill, whether from precipitation, surface or sub-surface flow or the application of liquid by other means, leachate generation is assured. The subsequent discharge of leachate may occur almost immediately or may be delayed by

TABLE 2.1 SOLID WASTE CLASSIFICATIONS

DOMESTIC, COMMERCIAL AND INSTITUTIONAL
- paper
- food waste
- textiles
- glass and ceramics
- plastics
- rubber
- leather
- metals
- wood
- yard waste
- bricks, stones, ashes

MUNICIPAL WORK FORCE
- dead animals
- street sweepings
- catchbasin cleanings
- ditch cleanings
- sewage grit and sludge

AGRICULTURAL
- field
- processing
- livestock raising

INDUSTRIAL
- mining, metallurgical
- food processing
- petroleum petrochemical
- forestry (wood chips, hog fuel, sawdust, harbour floatage)

DEMOLITION
- lumber, timbers
- broken concrete, asphalt
- bricks
- masonry
- granular materials

SPECIAL
- hazardous and toxic liquids and sludges
- radioactive wastes
- pathogenic
- pathologic
- international (ships, planes)

Source: From Wilson, 1977, with modifications by Soper, 1978 tens or even more than one hundred years depending upon local conditions. In most cases, preventing the movement of water into the fill will stop leachate production. An exception to this could occur in very deep fills where sufficient pressure is developed to consolidate the refuse in the lower layers. This would cause the moisture received with the refuse to migrate from the fill.

Control of infiltrating water has long been the major design criterion for a "sanitary" landfill. The result has been that regulatory agencies for many years have insisted upon the daily application of "suitable" soil and the provision of adequate surface slopes. This was intended to prevent rain water infiltration. Recent experience has shown that where differential settlement, burrowing animals, and penetrating root systems exist, virtually any soil cover will allow rain water to infiltrate and thus contribute to leachate generation.

Present day opinions indicate that in areas of significant rainfall, leachate generation cannot be prevented by the use of a soil cover. One eminent landfill designer uses a rule of thumb which states that the design rate of infiltration would be equal to 50% of the annual precipitation in excess of 508 mm (20 inches). For safety, the value so obtained should then be doubled. Therefore, all annual rainfall in excess of 508 mm (20 inches) should be considered as potential leachate. It is apparent however that local circumstances should be considered rather than a rule of thumb. For example, in sandy soil and where landfill surface areas are great, virtually all precipitation will eventually become leachate.

In recent years, preoccupation with the concept of sealing landfills to prevent water infiltration has led to the use of synthetic liners and cover materials. Materials used have included polyvinyl chloride (PVC), high density polyethylene (HDPE), chlorinated polyethylene (CPE) and butyl rubber and others. The useful life of these products has not been satisfactorily determined. This one dominant factor should be considered in the application of synthetic seals. Failure of the sealant due to lack of proper site maintenance, particularly in areas of pronounced rainfall, could lead to a substantial rate of leachate generation which future generations would have to cope with. Therefore, synthetic seals are not regarded as a permanent solution to the problem of leachate control.

Leachate production at virtually all landfills located in the Lower Mainland of British Columbia is seen to be a fact of life, considering today's technology. It is therefore necessary that landfills be designed with this in mind.

2.2.1 General Mechanisms of Leachate Production and Migration. During the early life of a landfill (usually six months to one year) biological activity, supported by the organics, nutrients and moving liquid in the refuse, will generally be aerobic. This early activity is followed by a transition period and subsequently by a stable anaerobic stage which does not cease until biological activity has stopped. During all the stages, biological activity breaks down the complex carbohydrates, fats and proteins into simpler molecules. Volatile organic acids form a major proportion of these molecules in both the aerobic and anaerobic systems. These acids, in conjunction with the carbon dioxide (CO_2) produced in both respiratory processes, reduce the pH and consequently increase the capability of the moving liquid to dissolve constituents with which it comes into contact.

During the early stages of leaching, the leachate will reach a maximum concentration which is controlled by the particle size of the refuse, the rate of water movement and by the equilibrium constants for the multitude of organic and inorganic species present. This maximum concentration will usually correspond with the first appearance of leachate if the refuse is relatively dry. The low refuse moisture content means that a relatively long contact time has occurred between the liquid and solid phases. With a higher refuse moisture content, the maximum concentration will tend to occur sometime after the first appearance of the leachate. Smaller refuse particle size will increase the peak concentration because of the greater surface area to volume ratio of the refuse. Increased rates of infiltration will tend to reduce peak concentrations through dilution.

The peak concentrations however are not reduced in proportion to the flow-through volume because more refuse surface area is exposed to water movement at the higher infiltration rates. While peak concentrations may remain high for sometime, a gradual decrease in concentration occurs, which, after many years, theoretically drops to background levels. Experience with landfills has shown that for practical purposes, background levels are never achieved. Persistent low level concentrations continue to appear for many years as can be seen at virtually any landfill which has been closed and which continues to show discolouration, notably from dissolved iron. At a closed local dump, concentrations of most constituents have after 12 years decreased by a factor of 5-10; however, as an example, total ammonia levels remain at about 50 mg/l.

Another major characteristic of landfills which shows the change from aerobic to anaerobic conditions is that of gas composition. During aerobic breakdown the major gaseous end product is carbon dioxide (CO_2), while anaerobic breakdown produces mainly CO_2 and methane (CH_4). The change from aerobic to anaerobic conditions is also characterized by a change in chemical species in the leachate from oxidized to reduced forms. This is exemplified by the appearance of sulphates in the aerobic phase and sulfides during anaerobic breakdown. Leachate flow rates and concentrations are significantly affected by extended dry periods and by sudden rainstorms. While flow rate, as expected, does respond in a damped fashion to the infiltration rate, the concentrations tend to follow a pattern which is not so obvious. That is, high concentrations tend to occur with high flow rates after a rainstorm, rather than as diluted concentrations which might be expected. The major reason for this behaviour probably relates to the long contact time between liquid in the landfill and the solids during a dry period. This has been found to provide the opportunity for leachate concentrations to increase within the landfill. A sudden rainstorm will tend to act as a large "plunger" and force this highly concentrated leachate out of the fill. In addition, refuse surfaces which have not been previously exposed to the movement of water, may become exposed if there is a sudden increase in infiltration. This could also increase contaminant concentrations in the leachate.

The rate of infiltration may also have an effect on the total mass of contaminants discharged. At low infiltration rates, it appears that biological activity is enhanced so that readily soluble intermediate and end products are extracted. At higher infiltration rates, while more refuse may be exposed to water movement, either the micro-organisms or the nutrients necessary for their survival are washed out. In this situation the contaminants removed from the refuse are those that are dissolved chemically or are physically carried out by the passing liquid. High rainfall rates will therefore tend to produce a lower total mass of contaminants than will the lower rainfall rates.

Fill depth is an important factor in leachate concentration. Increased refuse depth has been shown to reduce the concentrations of leachate contaminants. This has been attributed to self attenuation within the landfill. While this effect has not been fully explored nor satisfactorily explained, it is reasonable to assume that it would diminish with great depths.

It must be emphasized that fill depth greatly influences the time when the leachate will first appear, because the refuse must receive enough liquid to reach field capacity. This capacity is the moisture content at which liquid is free to be moved by gravitational forces. Depending upon the initial refuse moisture content, commonly used figures for field capacity range from about 200 to 500 mm per m of depth. A 30 m deep landfill would produce leachate after six years at an annual precipitation rate of 1500 mm, but would not produce leachate for 36 years at an annual precipitation rate of 250 mm. This example illustrates that under certain conditions leachate may not be produced until after a landfill has been closed. After closure, leachate production could continue for many more years. This latent effect has significant legal implications which have not been considered by those who must solve the leaching problem after the landfill has been completed.

The migration of leachate from a landfill can occur in two ways: surface discharge or a sub-surface discharge. Generally if the underlying soil is relatively permeable (sand or gravel), leachate will move vertically downward until reaching a groundwater table. It will then move in the direction of groundwater flow and possibly may appear in down gradient wells, springs or surface waters. At very low groundwater hydraulic gradients, leachate can migrate in a direction opposite to that of groundwater flow.

Where the subsoil is relatively impermeable (silts and clays), a minor volume of leachate will move vertically downward. Usually, the groundwater will rise within the fill, a so-called groundwater mound, until the outside elevation reaches that of the outside slopes of the landfill. This results in the appearance of leachate springs. It should be noted that if a relatively impermeable intermediate cover material is used in construction, then the leachate springs usually appear on the landfill side slopes at the elevation of the intermediate cover. Leachate from this type of landfill often occurs as a surface discharge. Landfills placed on highly compressible soils such as peat will tend to produce surface discharges because settlement of the underlying soil quickly produces a relatively impermeable barrier. During this settlement, liquid may be forced from the underlying soil and also produce a surface discharge. Landfills placed below the groundwater table may exhibit leachate migration characteristics of either of the above, depending upon the rate of groundwater movement, depth below the water table and precipitation rate.

Many factors influence the rate of leachate production, contaminant concentrations, time of first leachate appearance, time of contaminant concentration decay, and the location of leachate discharge. Every landfill has its own unique characteristics.

2.2.2 General Refuse. The characteristics of leachate from a municipal refuse landfill can vary dramatically from one landfill to another depending upon the factors described in the previous section. Unfortunately, there is no present method of predicting what leachate contaminant concentrations can be expected at any given time, nor is it possible to predict with certainty what the length of time will be during which contaminant concentrations will remain great enough to be of concern. It is therefore necessary to maintain a somewhat general approach when describing leachate characteristics.

Table 2.2 provides an insight into the broad range of contaminant concentrations which have been measured both from landfills and from simulated landfills. The table also includes data related to leachate springs from three local landfills and thus provides some perspective regarding variations with site age.

The maximum values shown in the first data column of Table 2.2 are peak contaminant concentrations in leachate from landfill lysimeters. These values represent the maximum concentrations which might be measured at the bottom of freshly placed refuse in the field. It is readily apparent that the leachate concentrations would far exceed any effluent discharge regulations which might apply. As shown in the bottom line of Table 2.2, the leachate would also be extremely toxic.

The second data column shows concentrations from a leachate spring at one of the local fills. The refuse contributing to this spring has probably been in place for about six years. The third and fourth data columns refer to a leachate spring at another local site. While it is not possible to state exactly how long the refuse contributing to this leachate has been in place, it probably averages about one to one and a half years. The fifth column shows data from a leachate seep. It is estimated that this leachate is from refuse less than one month old.

Using the approximate ages given for the springs, Figure 2.1 illustrates the differences which might be expected between data from lysimeters and from a full-scale landfill. It can be seen that some

TABLE 2.2 LEACHATE COMPOSITION - MUNICIPAL REFUSE

Parameter	Range Values Landfills and Test Lysimeters	Leachate Spring Site 1	Leachate Spring Site 2	Leachate Spring Site 2	Leachate Seep Site 3
Color (Chloroplatinate)	0 - 120 000	NA	NA	NA	NA
pH	3.7 - 8.5	7.1	6.2	6.3	5.3
Total Carbon mg/l	715 - 22 350		930	1830	NA
BOD mg/l	9 - 55 000	120	1140	2480	NA
COD mg/l	0 - 90 000	903	1860	4720	16 350
Acidity mg/l as CaCO	0 - 9590	185	540	790	NA
Alkalinity mg/l as CaCO	0 - 20 900	3400	1350	3050	1315
Hardness mg/l as CaCO	0 - 22 800	NA	NA	NA	4100 (calc)
T.Solids mg/l	1000 - 45 000	4636	3190	6490	NA
T.D.Solids mg/l	0 - 42 300	4502	3070	6470	NA
Tannin-Like Compounds (mg/l)	62 - 1278	62	NA	NA	NA
Chloride mg/l	34 - 2800	2400	125	390	875
Cyanide mg/l	0 - 0.11	NA	NA	NA	NA
Fluoride mg/l	0 - 2.13	0.3	NA	NA	NA
Nitrogen - Total mg/l N					
- NH mg/l	0 - 1106	427	0.3	37.5	190
Phosphorous - mg/l PO	0 - 154	6	14.0	9.3	53
Sulphate mg/l	1 - 1826	5	250	83	1455
Sulphide mg/l	0 - 0.13	NA	0.02	30	NA
Calcium mg/l	5 - 4000	175	535	1065	1310
Magnesium mg/l	16.5 - 15 600	126	39	84	196
Potassium mg/l	2.8 - 3770	600	51	137	NA
Sodium mg/l	0 - 7700	840	128	358	781
Arsenic mg/l	0 - 11.6	0.04	0.006	NA	(.15 to 0.2)
Aluminum mg/l	0 - 122	0.3	0.4	1.3	33
Barium mg/l	0 - 5.4	0.08	NA	NA	0.89
Beryllium mg/l	0 - 0.3	0.03	NA	NA	NA
Boron mg/l	0.3 - 73	4.5	5.9	7.4	NA
Cadmium mg/l	0 - 0.19	0.004	0.002	0.001	0.24
Chromium mg/l	0 - 33.4	0.05	0.025	0.085	0.69
Copper mg/l	0 - 10	0.02	0.05	0.01	.06
Iron mg/l	0.2 - 5500	30.3	22.4	1.6	280
Lead mg/l	0 - 5.0	0.06	0.051	0.023	0.39
Manganese mg/l	0.06 - 1400	0.6	4.3	7.8	31.3
Mercury mg/l	0 - 0.064	NA	NA	NA	.1
Molybdenum mg/l	0 - 0.52	0.01	NA	NA	.15
Nickel mg/l	0.01 - 0.8	0.07	0.002	0.012	0.59
Titanium mg/l	0 - 5.0	ND	NA	NA	1.17
Vanadium mg/l	0 - 1.4	ND	NA	NA	0.11
Zinc mg/l	0 - 1000	0.43	1.3	0.6	38.6
Toxicity (96 hr TL) %)	38 - 0.062	7.0	NA	4.2	NA

NA - not analyzed
ND - not detected
Metals are total unless otherwise indicated.

Source: Columns 1,2,3 and 4 - R.D. Cameron, 1978
Column 5 - E.P.S., 1979

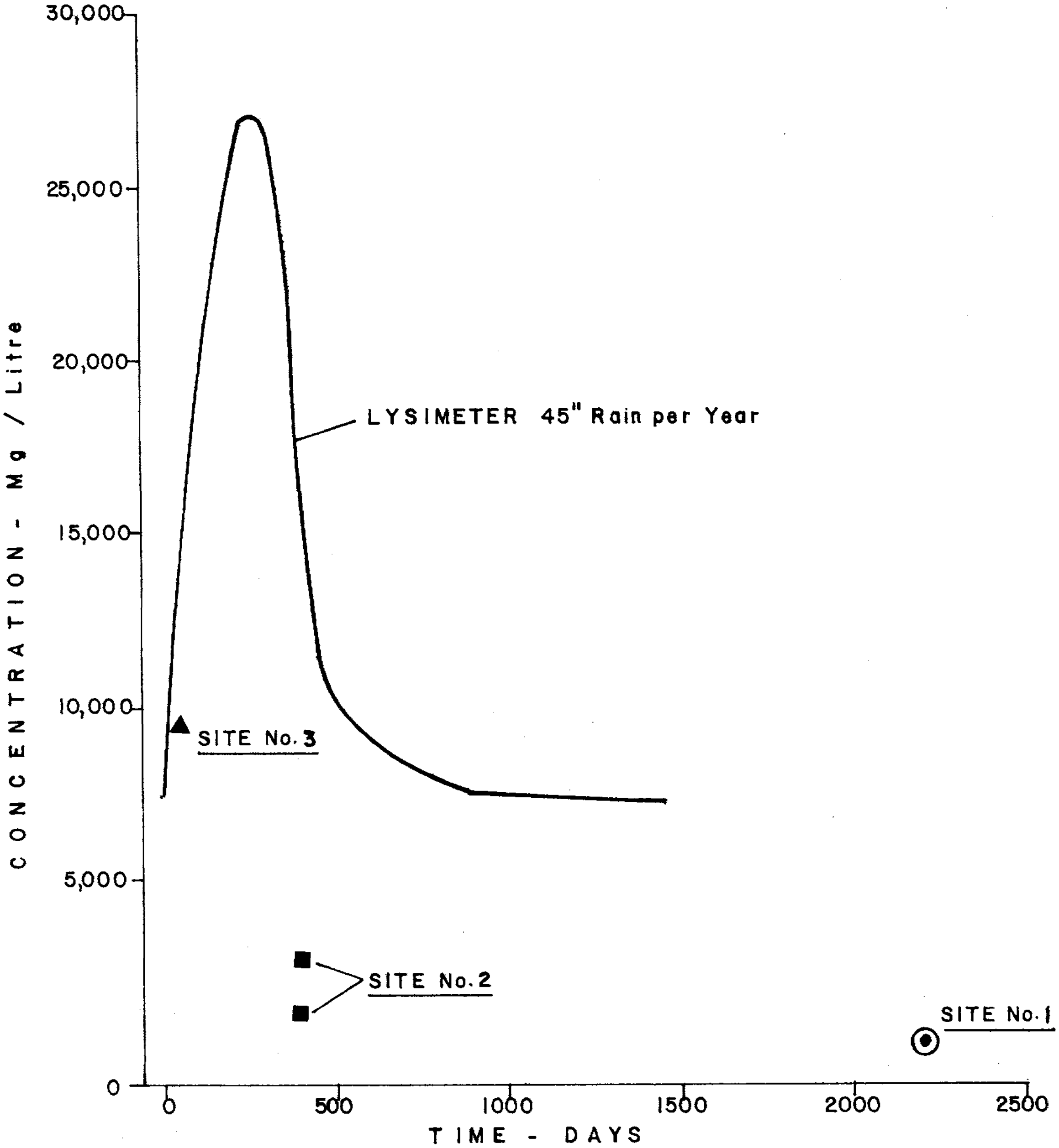


FIGURE 2·1 LEACHATE BOD_5 VERSUS TIME FOR LANDFILL SPRINGS type of attenuating mechanism is rapidly reducing the concentrations of BOD_5 in the leachate from two landfills; whereas, the near surface leachate from the refuse is comparative. As a greater volume of refuse is contributing to the leachate coming from the landfill springs than the refuse volume in the lysimeters, it is likely that the lower concentrations with time are due to self attenuation in the fill as the leachate moves through the refuse. On the other hand, total solids as illustrated in Figure 2.2 are more closely approximated and therefore may be predicted.

Another problem encountered is that of the different rates of extraction of constituents from the refuse. Figure 2.3 clearly illustrates this point. With these different rates of extraction and with changing ratios of the constituents to one another, designing a leachate treatment system or predicting the change in receiving water effects with time is, at best, extremely difficult. In general, readily soluble constituents such as sodium, potassium and chloride will be extracted rapidly and will be among the first contaminants to reach low levels with time. On the other hand, contaminants within the fills which have solubilities that are highly dependent upon pH and pE conditions, such as iron, will continue to be extracted over very long periods.

Toxicity is one of the best indicators of potential leachate effects on receiving waters. Although little work has been done in this area, some data are available. Table 2.3 shows toxicity data from landfill lysimeters and from landfill springs. The major reason for the different rates of reduction of toxicity from the lysimeters is likely due to rainfall effects. Tank H received 15 inches of precipitation per year, while Tank K received 90 inches. The greater rainfall rate has reduced contaminant concentrations from Tank K in approximately the same proportion as the toxicity reduction. The lower toxicity of the leachate from the landfills is probably closely related to the reduced contaminant concentrations due to attenuation within the refuse. On the other hand, the relatively high toxicity of the Site 1 leachate after

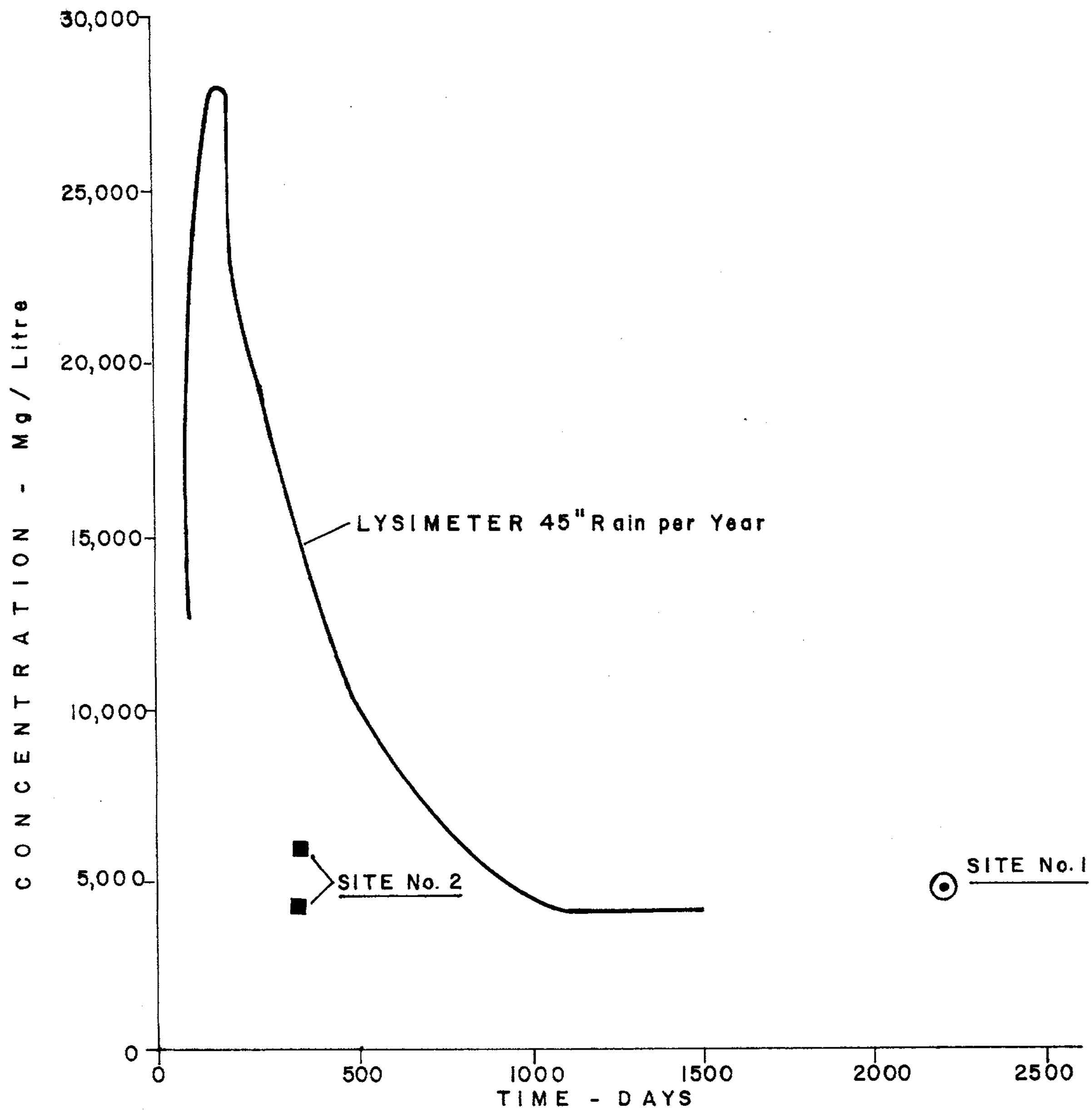


FIGURE 2·2 LEACHATE TOTAL SOLIDS VERSUS TIME FOR LANDFILL SPRINGS

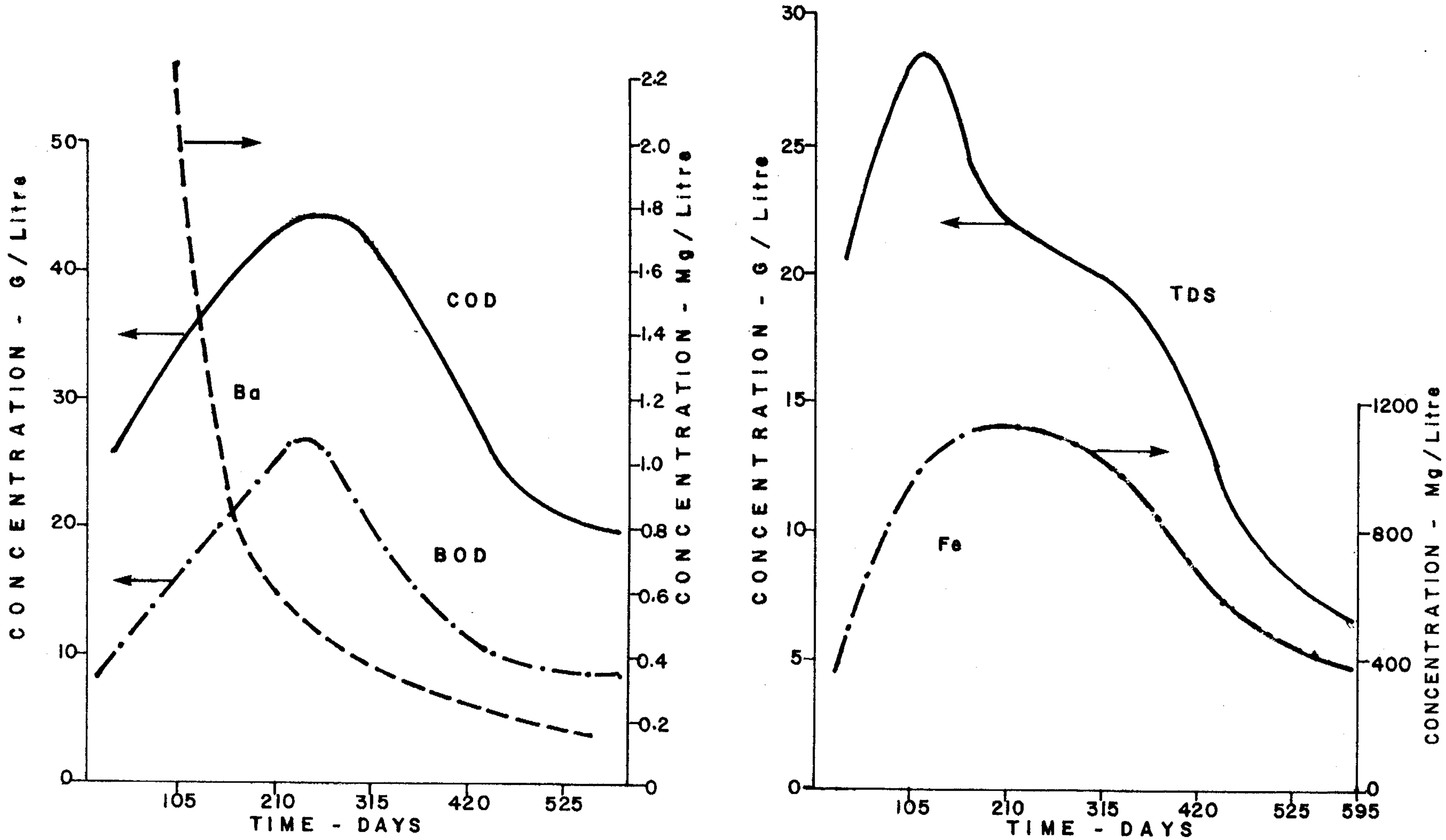


FIGURE 2·3 CONSTITUENT EXTRACTION FOR REFUSE RECEIVING 45" RAIN PER YEAR - UBC LYSIMETERS

TABLE 2.3 LEACHATE TOXICITY IN LYSIMETERS AND SITE NO. 1 AND SITE NO. 2 LANDFILL SPRINGS

Sample Type	Approximate Leachate Age	Toxicity (LC_{50}) % by volume
Lysimeter H	3 weeks	0.27
Lysimeter H	40 months	0.7
Lysimeter K	6 months	0.7
Lysimeter K	45 months	34.5
Site #1 Landfill	6 years	7.0
Site #2 Landfill	12-18 months	4.2

six years, shows that factors other than contaminant attentuations are affecting the toxicity of this leachate. Toxicity therefore, while an important parameter, can be expected to be as variable with time as are the contaminants previously discussed.

The preceding discussion has been limited to a consideration of "typical" municipal refuse. It has to be appreciated that if specific contaminants, such as those arising from industrial sludges, are added in substantial quantities to the refuse, leachate contaminant concentrations can be expected to increase. For example, if highly acidic or alkaline wastes are added to refuse the biological activity may be inhibited and, because many metals are tied up by microorganisms, metal concentrations in the leachate may increase. On the other hand, if a material having an active biological population combined with nutrients and organic material such as domestic sewage sludge or septic tank pumpings is added to refuse, a substantial reduction in leachate contaminant concentrations can occur. Research indicates that metal contaminant concentrations may increase and decrease by at least one order of magnitude respectively with these additions. This further complicates any predictions which might be made about leachate quality and its effects on the receiving environment.

2.2.3 Wood Waste. Leachate from wood waste landfills while usually having a more objectionable appearance than leachates from municipal landfills, will have a reduced adverse environmental effect because of lower contaminant concentrations and slightly lower toxicity over time. As biological activity has little effect on the quality of leachate released from wood waste landfills, contaminant concentrations are almost entirely controlled by the solubility of the wood extracts and the metals which may be contained in or on the wood. The behaviour of contaminant concentrations with time is similar to that for municipal refuse in that a type of exponential decay occurs. With wood waste, however, the decay is much more rapid.

Table 2.4 shows how both contaminant concentrations and toxicity are reduced with time for leachate from lysimeters containing predominantly hemlock wood wastes. The comparison should be made for each set of lysimeter analyses, i.e., z and 1-z rather than between individual lysimeters. It is to be noted that contaminant concentrations and toxicity from different types of wood wastes will vary. Several investigations have examined some of the aspects of leachability and toxicity of wood wastes, and although the results are not completely consistent the following data provide some perspective:

Cold Water Soluble Extracts

Cedar..........................10% to 20%
(mostly from heartwood)

Hemlock................................6%

Douglas Fir............................6%

Toxicity has been attributed to resin acids by one researcher (Steelson, 1974), while others, working mainly with Western Red Cedar, have found tropolones to be the major contribution to toxicity (Peters et al, 1976). Tropolones were reported to represent 5% of the water soluble fraction. The latter have reported the following toxicity data for Western Red Cedar cold water extracts:

Extract	96-hr LC$_{50}$	Test Species
Heartwood	120 mg/l*	Coho Fry
Heartwood (tropolones)	0.33 mg/l	Coho Fry
Bark	28.0 mg/l	Coho Fry
Foliage	2.0 mg/l	Coho Fry

* at 120 hrs.

TABLE 2.4 FISH TOXICITY AND ANALYTICAL RESULTS - PREDOMINANTLY HEMLOCK WOOD WASTES

PAIRS

Parameter	Sample Numbers					
	J-1	Z	N	1-N	1-Z	1-J
Toxicity	0.48	1.12	4.0	< 100	< 100	NA
pH	4.46	4.33	4.55	5.34	5.40	6.90
BOD_5	2160	1900	651	5.0	5.1	5.0
COD	6380	6220	5307	1060	711	400
TOC	3000	2950	1975	436	292	172
Alkalinity	NA	NA	NA	54.5	45.0	80.0
Acidity	1340	1620	200	[illegible]	35.0	11.0
T. Solids	4100	5410	5935	1100	812	509
V. Solids	2970	2970	3855	811	522	279
TDS	4080	5370	5922	1095	807	301
P	5.5	6.1	11.2	0.75	0.33	1.41
SO_4	1.6	8.25	67.0	< 2.0	6.0	NA
NO_3	NA	< 0.5	NA	< 0.05	< 0.05	< 0.05
Cl	474	1410	1220	7.4	6.4	3.9
F	0.03	0.18	0.15	< 0.1	0.1	< 0.1
Cn	NA	NA	< 0.05	NA	NA	NA
NH_3-N	< 0.3	2.2	0.56	0.3	< 0.3	< 0.7
Total N	7.6	10.1	9.9	3.34	2.33	7.46
Colour	8500	NA	NA	4000	2500	300
Tannin	1670	873	628	113	61.3	35.0
Na	209	450	480	38.2	60.0	9.0
K	89	170	152	16.7	13.2	6.14
Ca	31	109	131	13.0	10.1	9.43
Mg	36	99	100	3.3	1.7	7.17
Al	6.9	7.9	8.7	3.22	2.95	0.73
As	0.006	< 0.006	0.037	< 0.006	< 0.006	< 0.006
Ba	< 0.1	0.61	0.95	NA	NA	NA
Be	< 0.05	< 0.05	< 0.05	NA	NA	NA
B	1.44	0.76	0.60	0.39	0.22	0.41
Cd	0.007	0.008	0.006	0.001	0.001	< 0.002
Cr	0.03	0.06	0.039	0.006	0.004	0.002
Cu	0.04	0.07	0.03	0.086	0.060	0.090
Fe	27.6	55.2	16.0	6.0	7.5	15.5
Pb	0.07	0.07	0.084	0.003	0.005	< 0.01
Mn	8.8	30.2	25.2	0.83	0.59	0.78
Hg	< 0.004	< 0.005	< 0.001	< 0.0001	< 0.0002	NA
Mo	0.2	< 0.1	< 0.01	NA	NA	NA
Ni	< 0.02	0.01	0.06	0.012	0.008	< 0.005
Ti	NA	< 1.0	0.16	NA	NA	NA
V	< 0.3	< 0.3	< 0.05	NA	NA	NA
Zn	0.42	1.92	1.43	0.136	0.099	0.078
Days after first leachate appearance	223	126	37	1843	1759	1552

- All values except pH and toxicity in mg/l; Toxicity (LC_{50}) as % V/V.
- Cameron (1978)b
- Metals as totals.

There is conflicting evidence regarding the chemical oxygen demand fraction of water soluble extracts. One study showed the following order of COD: Douglas Fir > Western Red Cedar > Hemlock (Schermer et al, 1976); while another showed for 12" pieces: Western Red Cedar > Hemlock > Spruce > Yellow Cedar, and for 24" pieces: Western Red Cedar > Yellow Cedar > Hemlock.

One report (Schermer et al, 1976) says that Western Red Cedar extractions have a lower pH than those of Douglas Fir, while another (Thomas, 1977) says that Douglas Fir bark extractives have a lower pH than bark from Western Red Cedar. The latter research also reports higher BOD, TOC and total solids from fir than from cedar.

Another report (Cameron, 1978a) indicates that softwood bark degrades more than hardwood bark. This is interpreted to mean that there is a greater water soluble fraction in the softwood bark.

Data from test lysimeters which have been operated for about four years provide a further perspective. At a 15-inch annual rainfall, about 0.5% of the total mass of wood waste had been leached compared with about 2.3% of the total mass of municipal refuse. At 90 inches of annual rain, the percentages were about 1.7 and 3.7, respectively. These percentages of wood waste leached are much lower than those reported for total water soluble extracts. This is likely due to the probable difference in particle size (Cameron, 1978a).

Environmental conditions play a significant role in wood waste leachate toxicity. The effect of age of fill has been shown in Table 2.4. Iron (III) present in the leachate or in receiving waters was considered by one researcher (Dawson, 1974) to complex with tropolones and, in a 1:3 ratio, to eliminate tropolone toxicity. This same researcher also indicated that cedar extracts are metabolized or bound by contact with soil during overland flows, because in several logged areas there was no evidence of cedar leachates in the streams.

Another researcher (Schermer, et al, 1976) found that toxicity was significantly reduced when leachate was allowed to pass through a soil environment. Whether this was entirely due to attenuation rather than only dilution, was not determined.

Unquestionably, wood waste extracts can be toxic. The level of toxicity being dependent not only on the species involved but also on the type of waste: bark, heartwood, sapwood or foliage. It is also clear that toxicity of the leachate will decline fairly rapidly with time along with the reduction in percent extractibles. It also seems from the scant evidence available, that attenuation of toxicity can occur much more readily from wood waste leachate than from refuse leachate as the leachate moves over or below a soil surface. It is also apparent that leachates from wood waste landfills can, under the right conditions, make a significant contribution to environmental damage, and therefore cannot be overlooked.

2.2.4 Construction Debris. Construction debris is often referred to as "inert" material and consequently has been used at many landfills as a "mattress". In areas where underlying soil is very soft, this mattress forms a base upon which the landfill can be constructed.

The term "inert" is partially misleading as it implies that no leaching takes place. This is not strictly true as small amounts of material can be leached from old wood, concrete, bricks and other demolition material. However, the magnitude of any leachate problem which might arise is likely to be small. It should be noted, however, that some building materials may cause difficulties. One of the most evident is the deposit of substantial quantities of gypsum wallboard in a fill. Under anaerobic conditions extracted sulphate can be reduced to sulphide, thus causing a severe odour problem from the formation of hydrogen sulphide gas.

When construction debris is used as a mattress, channels can form beneath the refuse through which liquid can readily flow. This can result in the development of leachate springs at the top of the outer slope of the landfill. As such flow-through channels do not allow for self attenuation of the leachate in the refuse, springs may remain at relatively high strengths for long periods of time, even though they may be some distance from the landfill working area. It is also possible that the voids may provide an easy access for air into the landfill. If this occurs, aerobic reactions may take place with a consequent increase in refuse temperature due to biological activity. This may be one of the reasons why spontaneous combustion starts in the older portions of a landfill.

The use of construction debris, particularly concrete, may have a beneficial effect on leachate quality. In this situation, leachate having a low pH may dissolve and extract oxides and hydroxides thus raising the pH and encouraging the precipitation of metals. This type of behaviour is thought to occur at many landfills where leachate analysis has shown a relatively high pH combined with low metal concentrations.

In general, properly selected demolition debris will probably be more beneficial than detrimental to leachate quality. In areas where sub-soil is extremely soft, such as in peat, its use may be necessary.

2.2.5 Incinerator Residue. Residue from incinerators whether bottom ash or fly ash, if disposed in a landfill, will be subject to the same physical, chemical and biochemical reactions as those which occur in refuse landfills. In the past, incinerator residues were thought to be inert, but recent evidence shows that this is not the case. Incinerator residues can, in fact, have significant proportions of both unburned organic material and water soluble material.

Unburned organics have been found in two studies (Cameron, 1978a) to range from 4% to 43.6% of dry weight of residue, while the water soluble fraction was found to range from 1% to 17%. The range of characteristics expected in a leachate, based on results obtained in one of these studies, is shown in Table 2.5.

Leachate flow rates reported from one landfill varied from 0.5 up to 25 U.S. gallons per minute from a 4-1/2 acre, 40-foot deep fill.

No data were presented in the studies which showed the toxicity of the analyzed leachates. When compared with data from refuse landfills, however, it appears reasonable that the leachate would be toxic.

TABLE 2.5 RANGE OF LEACHATE CHARACTERISTICS - INCINERATOR RESIDUE

Parameter	Concentrations (mg/l)
pH	7.4 - 8.6
Alkalinity	6 - 4800
Nitrate	0 - 16.7
Phosphate	7.8 - 69.6
Iron	0.5 - 5.0
Chloride	3 - 2960
Fluoride	0.9 - 5.7
Sulphate	6 - 626
Calcium	5 - 23
Sodium	570 - 3900
Potassium	21 - 525
Total Dissolved Solids	7900*
BOD_5	125*
COD	1265*
Zinc	0.09*
Copper	1.15*
Chromium	1.53*
Lead	1.16*
Ammonia	47.6*
Total Kjeldahl Nitrogen	124.7*

* Mean values.
Metals as totals.
Source: Cameron, 1978a.

It must be noted that the concentrations shown in Table 2.5 represent those found between 8 and 30 months after leachate production had begun. The high values are therefore likely to be lower than those measured in the initial leachate discharge.

While incinerator residue leachates are likely to have considerably less strength than those from municipal refuse landfills, it is clear that they could have sufficient strength and volume to cause a noticeable addition to a receiving environment as evidenced by the results presented in Table 2.5. With the indicated fairly low values of BOD_5, the leachate might require a physical-chemical rather than biological treatment. Unfortunately, as no work seems to have been done on incinerator residue leachate treatment, conclusions regarding leachate treatment are simply speculation.

2.3 Attenuation and Containment Sites

Landfill operations can be generally classified into two categories based on site-specific hydrogeologic and management factors; namely, attenuation sites and containment sites.

Attenuation sites are landfill operations located such that the hydrologic, geologic and topographic nature of the site provides natural attenuation of any leachate formed. Generally, a contaminated groundwater zone is anticipated beneath and beyond the landfill operation. The boundaries of such a contaminated zone will be a function of the hydrogeologic factors of permeability and hydraulic gradient, which dictate the flow potential and the natural attenuation capabilities. Such attenuation is due to ion exchange adsorption, precipitation and dispersion within the underlying aquifer. The design of an attenuation landfill site requires detailed consideration of the hydrologic features of the site prior to and as altered by the proposed landfill operation.

Containment landfill sites can be classed as those which by natural geology or by environmental design inhibit the sub-surface migration of leachate. Natural geologic protection against vertical migration of leachate may be provided by impermeable rock or low permeability unconsolidated materials such as clays and silt-clays. Designed containment may be through the use of synthetic, impermeable membranes, bentonite clays, concrete or bitumen liners. When the field capacity of the fill is exceeded by water inputs which are greater than evapotranspiration, leachate will form, collect and mound at the base of fill and eventually migrate laterally along the surface. If protection against horizontal leachate migration is not provided naturally, it must be designed. Common means to achieve this include impermeable cutoff walls, level controlled interception ditches, and purge wells.

Assuming that 100% containment is obtained, there still remains the task of determining an ultimate point of disposal for the collected leachate. The two disposal options available are discharge to a receiving water or discharge to a municipal sewer. Due to the character of the leachate, some form of treatment or pre-treatment will often be required to attain pollution control receiving water objectives in the case of direct discharge. Given the varied and frequent high-waste strength of the leachate, treatment can be a formidable task. There is also a need to prevent damage to sewer conduits and/or prevent sewage treatment plant upset in the case of a municipal connection.

Environmental management practices laid down by some jurisdictions have attempted to identify a landfill site classification guideline based on the hydrogeologic nature of the site. One of the oldest established controlling agencies is the Los Angeles Regional Water Quality Board, which classifies disposal sites and specifies the type of materials that are acceptable at each class of site. Table 2.6 outlines the provisions of this classification procedure.

2.4 Leachate Treatability

The treatment of leachate is very much in its infancy. Work to date has indicated that it is both expensive and largely site specific.

A review of the current "state of the art" of treatment systems and some leachate treatment costs are presented in Appendix A.

TABLE 2.6 LOS ANGELES WASTE DISPOSAL SITE CLASSIFICATION

Site Class	Hydrogeologic Character	Acceptable Materials
Class I	- sites located on non-water-bearing rock underlain by isolated or unusable aquifer; - protected from surface runoff and control of surface drainage to acceptable wasteway	- all solid and liquid waste
Class II	- sites underlain by usable, confined or unconfined aquifer with landfill maintained above high ground water elevations; - protected from surface runoff and control of surface drainage to acceptable wasteway	- typical municipal refuse and Class III materials
Class III	- sites located so as to afford little or no protection to usable waters	- non-water soluble non-decomposable inert solids

Source: Dept. of H.E.W., 1969.

3 LARGE MUNICIPAL LANDFILLS CURRENTLY ACTIVE

Five large municipal landfills are in operation (1977) in the study area. They are the Burns Bog or Vancouver Delta Landfill, Richmond Landfill, Port Mann or Surrey Landfill, Braid Street or Coquitlam Landfill, and Leeder Landfill. The first four landfills receive refuse from households, commercial establishments and industries within the study area. The Leeder Landfill receives only selected commercial and industrial solid wastes. During 1977, an estimated 760 000 tonnes of solid wastes were landfilled at the five sites. Each of the five sites is described in this Chapter.

3.1 Burns Bog - Vancouver Delta Landfill

3.1.1 General. The landfill is located in the Municipality of Delta on some 400 ha of the Burns Bog area. Operation of this site was initiated in 1966, following completion of the City of Vancouver Kerr Road site (see Section 4.1). The site is owned and operated by the City of Vancouver under an agreement with the Corporation of Delta. The agreement specifies certain conditions and considerations, including acceptance of Delta refuse and reversion of completed fill to Delta for future recreational development.

3.1.2 Physical Description. The Burns Bog area is located south of the main arm of the Fraser River in the north-eastern portion of the Corporation of Delta. The 404 ha landfill site located north of Highway 499 is being filled from west to east. Approximately 40 ha has been filled to date. Figure 3.1.1 shows the landfill location, site plan, and the approximate filled area.

The geologic history is typical of much of the low lying Fraser River Delta. In recent millenium the bog area was cut off from the main river channel during deltaic deposition. The undrained backwater was infilled with silt and clay deposits from flood plain sedimentation, and was subsequently infilled with organic material. The resultant geologic stratigraphy is a surface peat layer averaging

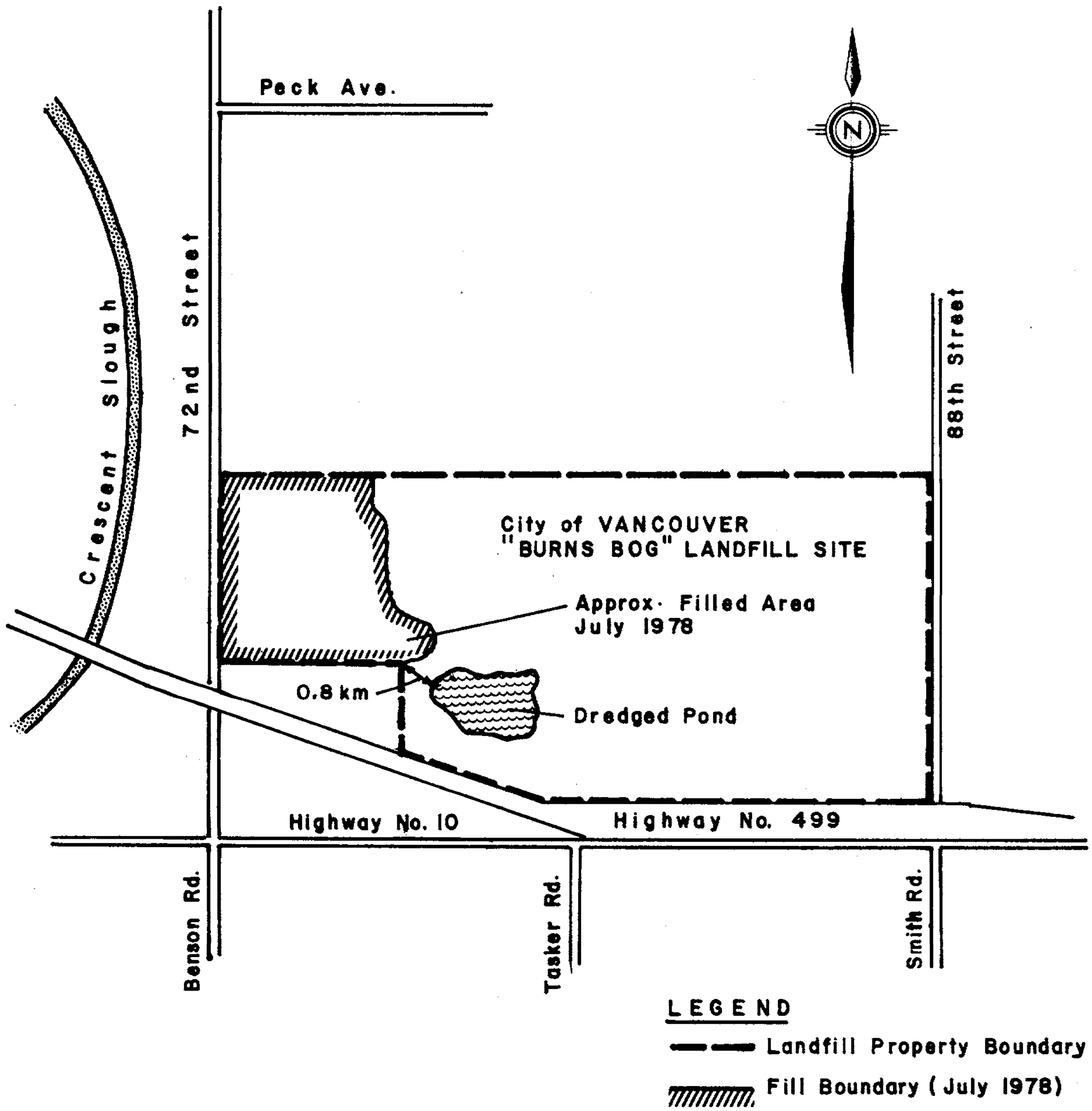


FIGURE 3.1.1 BURNS BOG LANDFILL SITE PLAN

2.75 meters in thickness underlain by a relatively thin silty-clay unit and a permeable sand unit several tens of meters thick. The Provincial Government Department of Highways data indicate that to the south there is an absence of surface peat and that clay is on the surface. Soils maps for the area suggest that the landfill is located on the edge of the main bog, which consists of an organic deposit of decomposed peat rather than sphagnum peat. Figure 3.1.2 shows a geologic fence diagram developed from test auger borings (City of Vancouver, 1973).

The bog area is generally poorly drained. The runoff associated with the approximate 1000 mm/annum rainfall is conveyed by drainage control ditches throughout the area. Groundwater level is at or near the surface over most of the year. Drainage ditch flows are to Crescent Slough which, in turn, discharges to the Fraser River. The flow of the ditches to Crescent Slough is maintained by pumps during high river stages.

Geotechnical studies at the Burns Bog site and other similarly located landfill sites, indicate that the hydrogeologic characteristics of both the native peats and the landfilled peats generally preclude vertical migration of leachates and egress into the main (sands) underlying hydrostratigraphic unit. These phenomena are the result of the natural low permeability in the underlying silt clay unit in the case of native unconsolidated peat bogs, and are coupled with a load-caused reduction in peat permeability in the case of consolidated peats.

As a result of the load-caused reduction in peat permeability, water levels in the landfilled areas tend to rise in the fill. Consequently, the leachate seepage discharge results from positive water levels in the fill relative to the surrounding native bog area. The landfill flow system is generally isolated and distinct from the groundwater flow system of the general area. Based on flow net analyses carried out for the City of Vancouver by Golder Brawner and Associates in 1973 (City of Vancouver, 1973), it was postulated that almost all of the landfill seepage discharge (92% to 99%) would be captured within 3 metres of fill toe by suitably sized and located drainage interception works.

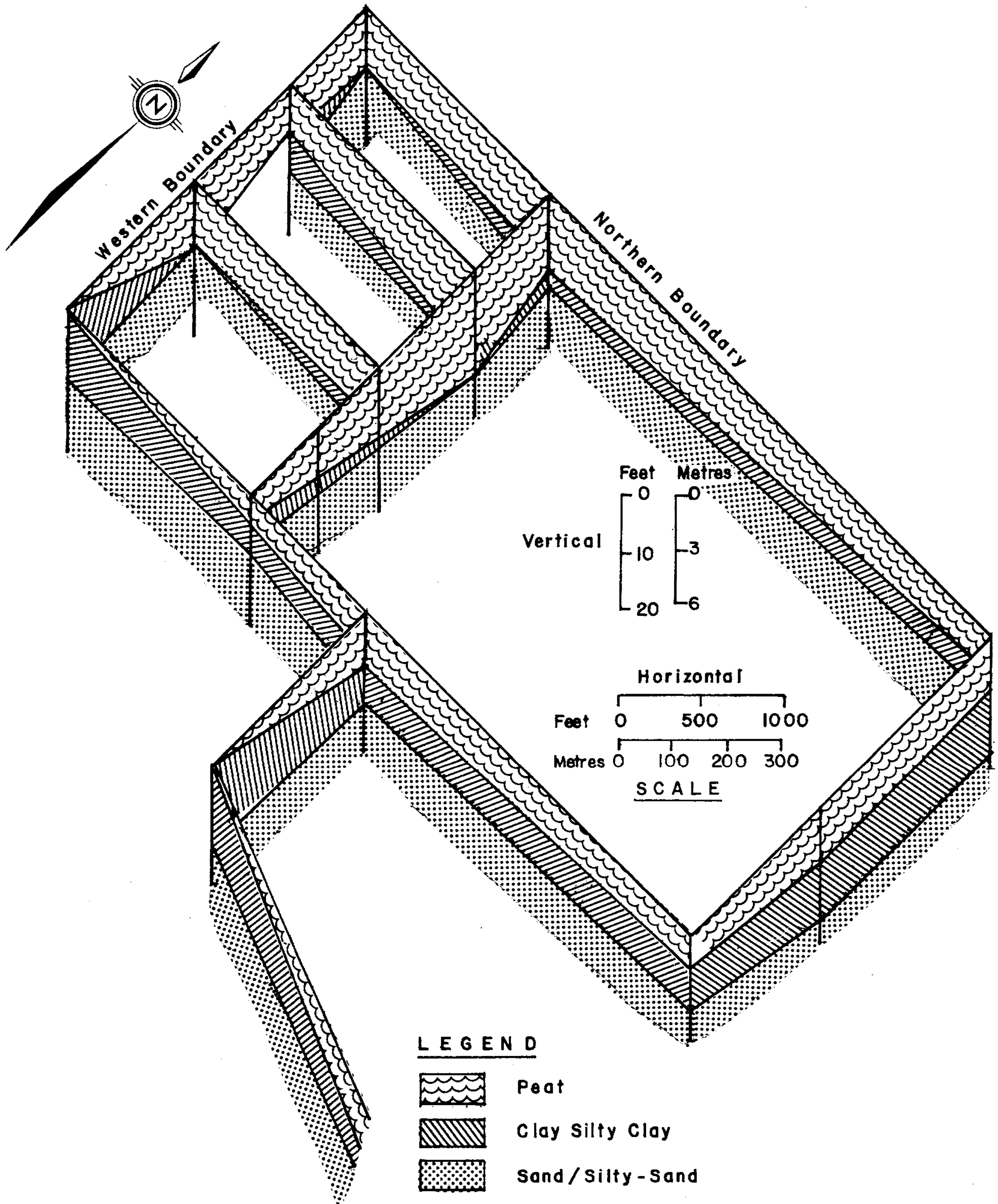


FIGURE 3.1.2 GEOLOGICAL FENCE DIAGRAM - CITY OF VANCOUVER - BURNS BOG

It should be noted that this general seepage flow model assumes the continuous geologic stratigraphy as outlined. The integrity of both the silt-clay unit and the peat unit is believed to be maintained throughout the site, although geologic bore hole investigations have shown that some pockets or lenses of sand exist in the peat unit.

Recent studies have shown that as a result of peat consolidation beneath the filled area, there is a saucer-like depression (City of Vancouver, 1975b). This depression is a maximum of 4.5 metres at the centre, and tapers out to less than a metre at the perimeter. It is postulated that this depression together with the load-caused permeability reduction, will tend to collect and pool leachate.

3.1.3 Operation. Refuse discharged at the Burns Bog site is received from the following sources:

- City of Vancouver
- Municipality of Delta (private contractor)
- Municipality of White Rock (private contractor)
- University of B.C. and Endownment Lands,
- commercial haulers
- private citizens.

Of the approximate 227 000 tonnes of refuse discharged annually, approximately 60% to 70% is received from City of Vancouver sources and the balance is received from the other listed sources.

The landfill is open to the public 14 hours a day, seven days a week with City of Vancouver vehicles discharging on a 24-hour per day basis. The bulk of the refuse discharge (approximately 93%) is confined to Monday to Friday, with the weekend traffic consisting primarily of private citizen discharges of yard refuse. All vehicles are weighed upon entering the site.

Site construction comprises a series of three lifts. The first lift 1.8 metres in depth is mattress made up of primarily demolition materials. The subsequent two refuse lifts, each 3.6 metres deep, settling to 2.4 metres, are constructed by spreading the refuse in uniform layers on a sloped working face and compacting to an average depth of 0.6 metre. The compactive energy is applied by 27-tonne "Hyster" steel wheeled compactors. Fifteen centimetres of sand cover material is placed on the top of the refuse lift. Intermediate cover is placed each working day and working faces and slopes are covered when completed.

Cover material is obtained from a dredging operation (see Plate 3.1.1). It should be noted that the dredging operation itself results in a man-made geologic discontinuity by removal of the peats and clays overlying the silts and sands; however, loss of leachate through this discontinuity can be prevented through the use of the double ditches.

To date, approximately 40 ha of the total 404 ha have been filled and completed. A 1.8 metre peat/clay/silt final cover mixture obtained from the site is placed on the completed fill. Following levelling and grading, a suitable cover crop can be planted. Figure 3.1.1 and Plate 3.1.1 show the area completed to date at the approximate 4 ha per year fill rate.

Liquid and sludge wastes discharged via tank trucks to the fill are estimated to be in the order of 2 273 000 litres per year (Atwater, 1978).

The life expectancy of the fill, with consideration given to increased rates of disposal, has been estimated to be up to 40 years. Obviously the method of lift construction and height will influence the fill life expectancy, but given the present approximate 4 ha per year fill rate, the 40-year estimate appears conservative.

Given the estimated lift height area coverage and fill depth, the in-place density at the Burns Bog site is estimated to be approximately 770 kg per cubic metre.

3.1.4 Leachate. Given the hydrogeologic nature of the site, it is reasonable to postulate that most of the leachate formed as a result of water inputs will be captured by drainage interception works; and, until the sewer connection has been completed, will be conveyed via the drainage ditches to Crescent Slough.

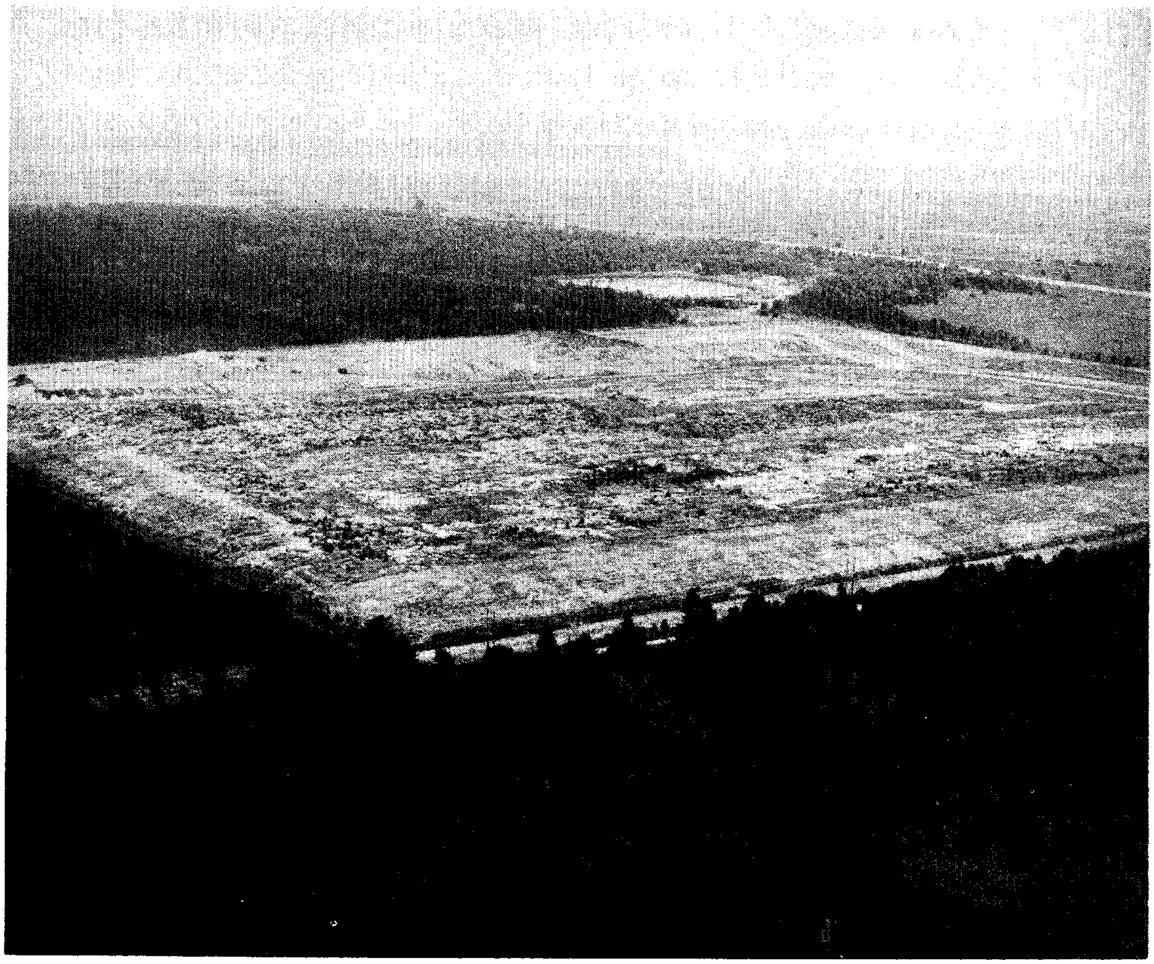


PLATE 3.1.1 AERIAL PHOTO OF BURNS BOG LANDFILL, SHOWING FILL AREA AND COVER DREDGE OPERATION (Top Centre)

The drainage scheme up to the end of 1978 has involved interception and combination of the landfill leachate seepage with the natural drainage and runoff from the surrounding land. The amount of initial dilution had been estimated to be an average of 5:1 (City of Vancouver, 1975b), based on average winter rainfall and the assumption of total runoff from the landfill and contributary surrounding drainage areas. It was also postulated that during higher than average rainfalls the landfill rentention will be greater than the surrounding areas, with the time lag between the two discharges resulting in a greater dilution than the 5:1 average during storm events. Figure 3.1.3 shows the pre-1978 drainage flow patterns at the landfill site.

Under the terms and conditions of a provincial pollution control permit (PR-1611), issued to the City of Vancouver for the Burns Bog operation in 1978, leachate containment and collection works are to be installed at the site with subsequent discharge to a sewer trunk connected to the Annacis Island STP. The works involve a double ditch system which will be expanded with the fill. This system, shown in Figure 3.1.4, will allow hydraulic isolation of the surrounding area drainage and the interception of the leachate seepage, thereby minimizing flows. Based on the consultant's estimates, the flow to be discharged to the sewer will be initially in the order of 0.02 m^3/sec for 40.4 ha, and will increase as the landfill area expands to a theoretical maximum of 0.2 m^3/sec for the total 404 ha. The outside ditch will allow interception and diversion of surface drainage waters. The inside ditch will intercept and collect the landfill leachate. A cut-off ditch on the east boundary of the west to east advancing fill, will allow flow minimization by a drainage module approach. Forty-eight ha are enclosed within the double ditch system at this time. The leachate is collected in wet wells for pumping to a proposed Annacis Island STP interceptor, which will follow an alignment to the north of the site adjacent to Crescent Slough. The collected leachate will continue to be discharged to Crescent Slough until the connection to the interceptor can be completed. The wet well pumping works will allow for control of leachate ditch levels, thereby providing

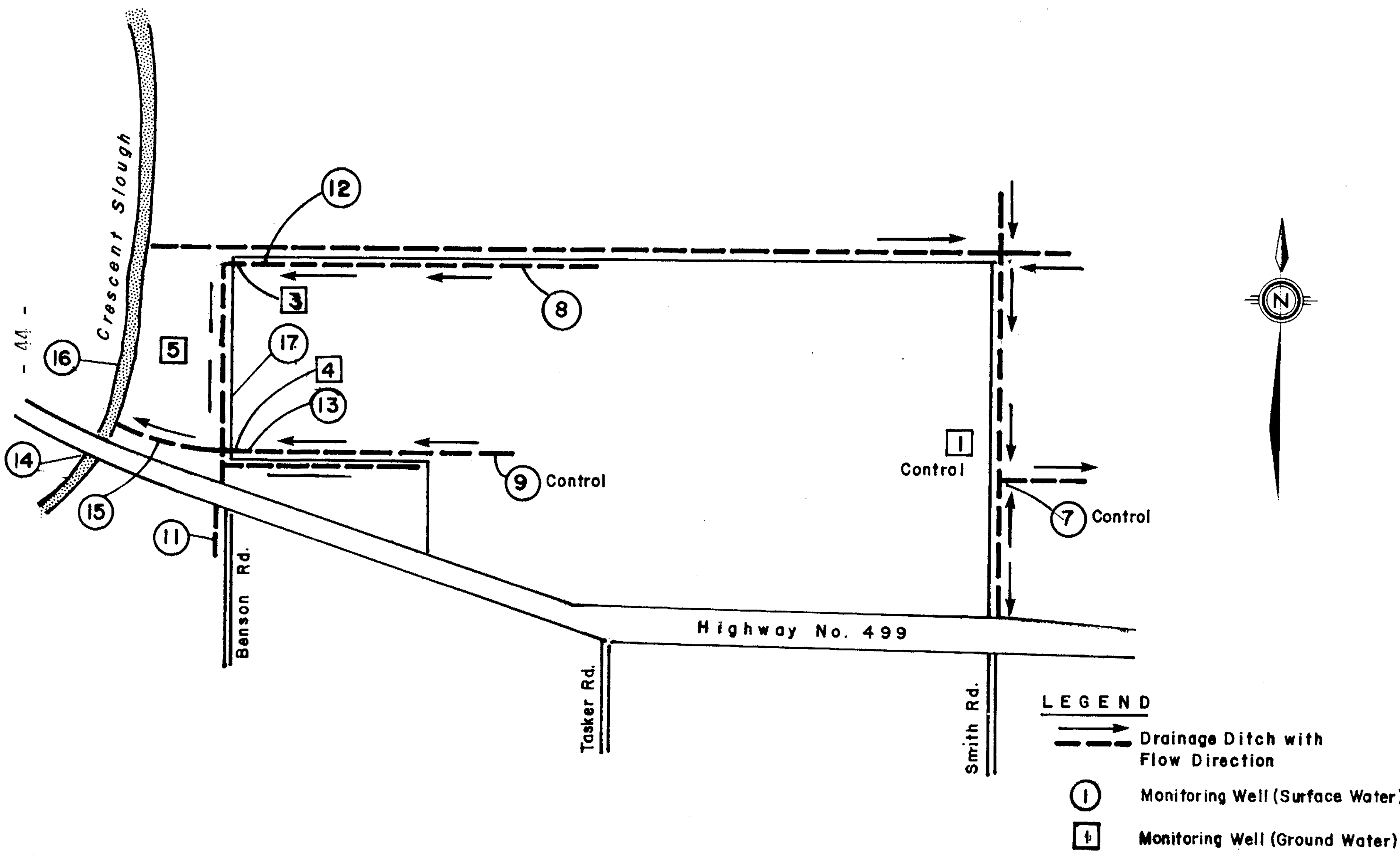


FIGURE 3.1.3 BURNS BOG LANDFILL - DRAINAGE PATTERNS AND MONITORING STATIONS (Pre 1978)

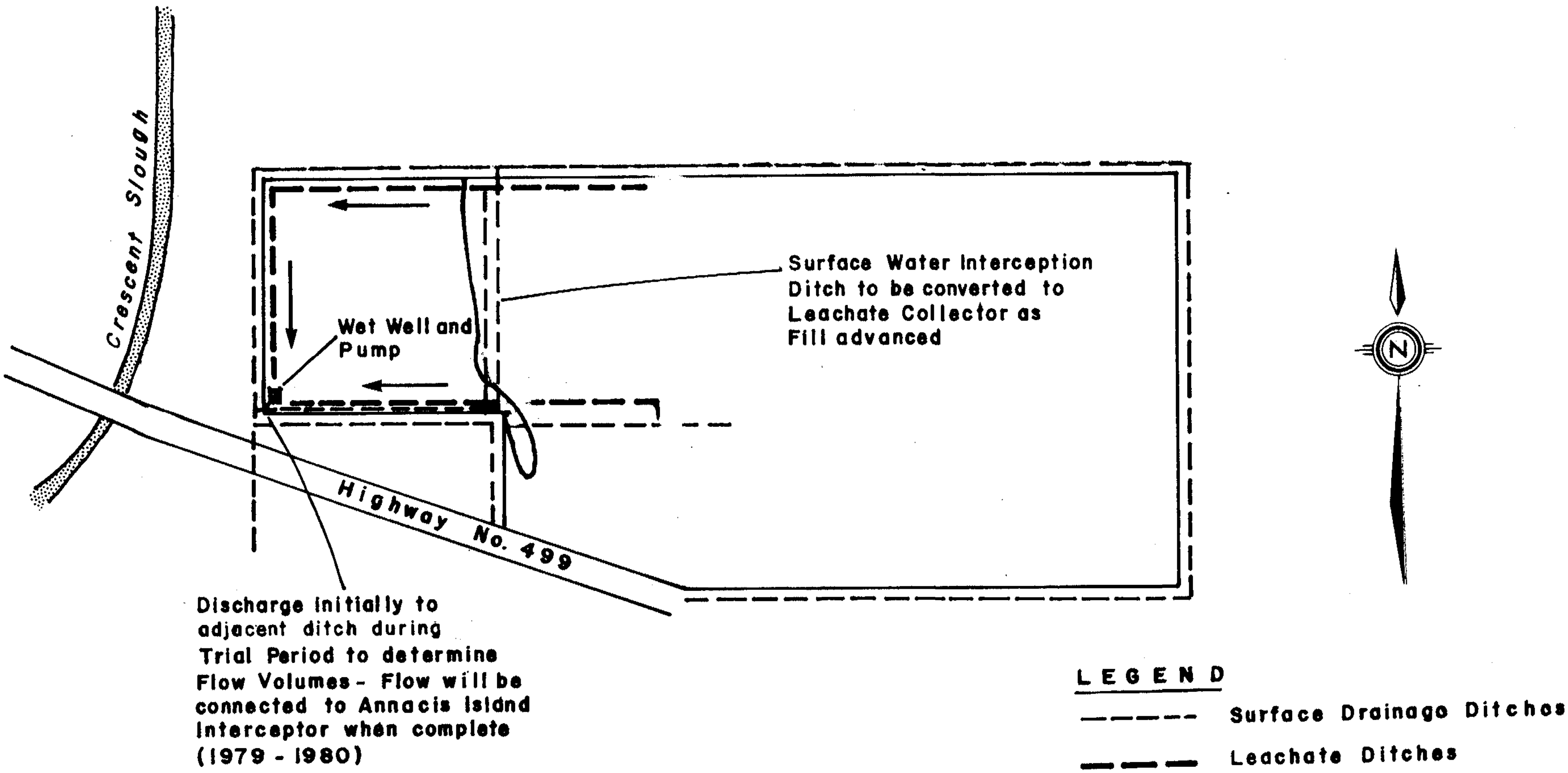


FIGURE 3.1.4 BURNS BOG LANDFILL - LEACHATE CONTROL WORKS
(Post 1978)

a negative hydraulic gradient and precluding any leachate migration to surrounding lands. It is anticipated that the control of the hydraulic gradient will be a critical operating factor. This is because sufficient gradient must be maintained in order to contain the water leachate, but too much gradient will require the unnecessary transport of excess water.

The determination of a precise water budget for the site is difficult due to the uncertain size of the component flows. Water inputs are precipitation and liquids discharged to the fill (tank truck contents, dust control, liquid component of refuse). This latter liquid input is estimated to be less than 10 mm/year over the fill surface. Water losses are evapotranspiration, surface runoff, leachate losses to surface ditches, and leachate loss to ground-water (negligible). Evapotranspiration should account for all of the May to August precipitation and possibly September precipitation in a dry autumn.

Surface runoff is expected to be negligible except for high intensity storms due to porous cover material, irregular fill surface, and minimal slopes. Water loss due to surface runoff would not result in an overall reduction in water volumes; however, it would result in a dilution of leachate as that portion would not have interacted with the fill refuse.

The difference between the water input and loss should occur as water stored in the fill. Refuse entering Burns Bog typically has a moisture content of about 25% on a dry weight basis (Bird & Hale, 1978); whereas, refuse normally has to contain about 65% moisture (field capacity) before leachate flows. The 40% difference in moisture content would require about 4 years worth of infiltrating water at normal precipitation rates. Or, conversely for Burns Bog, since it is 12 years old, about 1/3 of the infiltrating water. The following rough budget can be constructed using precipitation data for November, 1978 to October, 1979.

Water in - water stored - evaporation = leachate out:
758 mm - 1/3 (758-112 mm) - 112 mm = 430 mm.

Since December,1978, the City of Vancouver has been collecting and pumping leachate from the double ditch system in order to determine flow rates. The average flow rate measured is 0.007 m^3/sec, equivalent to 450 mm of moisture over the ditched area, which agrees closely with the water budget. The average precipitation for the area based on historical records is about 1060 mm which would produce about 600 mm of leachate or a flow of 0.0093 m^3/sec. As the city will be filling in the area presently ditched for two or three more years, the leachate flow from Burns Bog could be expected to stay in the range of 0.007 to 0.0093 m^3/sec for that period. After that period, the flow will increase in proportion to the newly ditched area.

Based on the established general hydrogeologic picture, leachate would be produced as a function of refuse contact with the water inflows. The actual contact process would be expected to be a complex process dependent on the nature and permeability of the refuse which, in turn, will be influenced by the operational procedures of lift construction and compaction. Leachate seepage flows will vary considerably over time. It has been suggested that the landfill acts somewhat as an accumulator, with the rainwater retention period within the fill being in excess of one month (City of Vancouver, 1975b). Accordingly, the rate of leachate discharge is not expected to depend directly on rainfall intensity. The hydraulic gradient between the landfilled area and the perimeter ditching is expected to influence directly the rate of leachate outflow.

The quality of the leachate from the Burns Bog site was examined in 1977 in conjunction with renewal of a permit under the Pollution Control Act (PCB, 1977). The large amount of data available to that time was subjected to a statistical analysis; these findings are referred to where possible in the following discussion.

Alteration of on-site and contiguous off-site ditching and growth in landfilled areas have resulted in changes in the importance placed on monitoring locations that have been designated since 1966. Monitoring locations relevant to site conditions prior to the start of construction of leachate control works in 1978 are shown in Figure 3.1.3.

A summary of the function, type and kind of water associated with each monitoring site is presented in Table 3.1.1. As the ground and surface waters of the bog are highly coloured and of variable chemical composition, it is often necessary to describe leachate quality relative to the background water quality rather than in absolute terms.

A comparison of leachate in the landfill wells #3 and #4 with background control wells #1 and #2, showed higher levels of specific conductance, chloride, hardness, ammonia, nitrate, COD and iron; whereas, the bog groundwater wells #1 and #2 had higher concentrations of tannin and lignin, chromium, copper, lead, zinc, and much lower pH values.

A similar comparison of the quality of surface leachate sites #13 and #17 with background ditches from the bog sites #7 and #10, showed the leachate water to be higher in specific conductance, pH, ammonia, Kjeldahl nitrogen, COD, and iron.

Summaries of available data on sites 1, 2, 3, 4, 5, 6, 7, 10, 11 and 15 are presented in Appendix B. The data were taken from an August, 1978, EQUIS printout.

3.1.5 Impact on Water Quality

3.1.5.1 Point of Egress. While the site geology would appear to preclude any downward migration of leachate, some lateral migration of leachate through the ground could have been anticipated prior to the construction of the double ditch system. A comparison of concentrations in the downstream wells, #5 and #6, with the upstream control wells showed increases in concentrations for some parameters; however, there were pairings in which one downstream site was higher than an upstream site but opposite in the other pairing. The parameters that were higher downstream were specific conductance, hardness, ammonia, and phosphorous.

TABLE 3.1.1 BURNS BOG LANDFILL - SYNOPSIS OF MONITORING SITES

Site	Type	Function	Kind of Water
1	Well	Control	Bog water.
2	Well	Control	Bog water.
3	Well	Contamination	Leachate in fill, undiluted (old).
4	Well	Contamination	Leachate in fill, undiluted (old).
5	Well	Contamination	Bog water 150 m west of site. Check for leachate loss from perimeter ditching.
6	Well	Contamination	Bog water 150 m west of site. Check for leachate loss from perimeter ditching.
7	Surface	Control	Bog water outside site.
8	Surface	Control	Bog water upstream of mixing with leachate (N-side).
9	Surface	Control	Bog water upstream of mixing with leachate (S-side).
10	Surface	Control	Bog water.
11	Surface	Control	Drainage water upstream of mixing with water from landfill site.
13	Surface	Contamination	Leachate and bog water mix (S-side of fill).
14	Surface	Contamination	Crescent Slough south of leachate/drainage ditch discharge.
15	Surface	Contamination	Mix of leachate/bogwater and off-site drainage.
16	Surface	Contamination/ Control	Crescent Slough north of leachate/drainage ditch discharge.
17	Surface	Contamination	Mix of leachate and bog water (N + W side of landfill).
19	Well	Contamination	South of active site, located on private land.
20	Well	Contamination	South of active site, located on private land.
21	Well	Contamination	South of active site, located on private land.
22	Well	Contamination	South of active site, located on private land.

Attributing the increased concentrations in the groundwater down gradient from the site solely to landfill leachate may be questionable, as the downstream wells are located in agricultural land subject to heavy applications of fertilizer.

Prior to the newly constructed double ditch system, the single perimeter ditch system shown in Figure 3.1.3 directed the leachate to the southwest corner of the site. The collected flow then discharged to Crescent Slough as shown. Accordingly, representative surface ditch sample points are: Sites 13 and 17 in the perimeter ditch system prior to mixing with drainage water; Site 11 representing natural ditch water prior to mixing with site waters; and Site 15 representing the downstream mixed site waters and natural drainage waters, prior to discharge to Crescent Slough.

Table 3.1.2 outlines the monitoring results for Sites 13 and 17 for the period 1974 through 1978. The values shown indicate a considerable range but compare well with one another. Nitrogen forms appear to be the most significant with ammonia present in concentrations in excess of 100 mg/l.

3.1.5.2 Defined receiving water past and present. The receiving water for the collective landfill leachate discharge is Crescent Slough. It could, however, be argued by definition that the connecting surface drainage ditch between the landfill and Crescent Slough is the receiving environment. Table 3.1.3 shows the results in Crescent Slough of sampling upstream and downstream from the landfill drainage confluence at Sites 16 and 14, respectively. On comparison of mean values, it is seen that there is a general trend toward increasing contaminant concentration downstream. Nitrogen forms again show the most pronounced increase along with colour and some metals. However, a comparison on a sample-by-sample basis does not clearly show this trend. A qualification to this last point is that site #16 may not be located far enough upstream to avoid some contamination on reversing tides. A statistical analysis of these data was not carried out.

TABLE 3.1.2 BURNS BOG LANDFILL - MONITORING DATA FROM STATIONS 13 AND 17

Parameter	Station 13			Station 17		
	No. of Samples	Range Min. - Max.	Average	No. of Samples	Range Min. - Max.	Average
Colour	15	40 - 1000	593	8	75 - 1375	775
pH	28	6.3 - 8.2	7.3	14	7.1 - 7.9	7.5
Spec. Cond.	33	690 - 7680	2156	16	900 - 9640	3002
DO	20	.5 - 6.5	3.1	11	.3 - 6	3.49
Chloride	23	50.2 - 965	246	12	86 - 625	315
Fluoride	22	.07 - .5	.17	9	< .1 - .51	.21
Hardness	7	219 - 611	356	5	83.5 - 408	217
Ammonia	25	1 - 350	71.5	12	5.6 - 228	114
Nitrate	22	.02 - 2.91	.602	10	.03 - 6.42	1.6
Kjeldahl N	17	4 - 326	77	11	9 - 241	118
COD	23	74.9 - 604	223	12	37 - 460	286
Tannin & Lignin	24	3.9 - 38	14.8	5	10.8 - 30	19.8
As (T)	23	.005 - .021	.0017	2	.013 - .016	.0145
Cd (T)	3	< .0005	<.0005	3	.0005	<.0005
Ca (D)	7	57 - 123	85.8	5	14 - 97.5	43.8
Cr (T)	26	0.005 - .082	.015	2	0.006 - .017	.0113
Cu (D)	4	<.001 - .004	<.003	2	.005 - .008	.0065
Cu (T)	20	.002 - .04	.012	4	.003 - .01	.006
Fe (D)	13	.2 - 15.4	6.3	8	1.2 - 15	7.1
Fe (T)	15	1.2 - 14.7	7.6	6	.3 - 10.1	4.7
Pb (T)	23	0.01 - 0.1	.02	5	0.002 - .009	.006
Mg (D)	8	18.3 - 92.5	34.2	6	11.8 - 47.2	29.5
Mn (D)	4	0.1 - .65	.385	2	.16 - .49	.325
Mn (T)	23	.16 - 1.13	.66	5	.09 - .72	< .27
Hg (T) μg/l	4	<.05 - <1	< .30	4	<.05 - 1	.29
Ni (T)	2	.01 - .02	.015	2	.01 - .02	.015
Na (D)	6	59.6 - 242	119	6	92 - 372	210
Zn (T)	19	.01 - .45	.13	3	.04 - .07	.53
Al (D)	7	.01 - 1.4	<.48	4	.02 - <1	< .29

Colour................TCU
Specific Conductance...mmho/cm
pH....................pH units
All others............mg/l

T = Total
D = Dissolved

TABLE 3.1.3 BURNS BOG LANDFILL - MONITORING DATA FROM STATIONS 14 AND 16

Parameter	Station 16			Station 14		
	No. of Samples	Range Min. - Max.	Average	No. of Samples	Range Min. - Max.	Average
Colour	8	100 - 1000	469	15	100 - 1125	570
pH	15	6.3 - 7.6	7.0	31	4.7 - 8	7.2
Spec. Cond.	16	170 - 8800	1483	32	365 - 5120	1755
DO	10	1.7 - 9.6	4.7	22	3 - 7.9	3.78
Chloride	12	8.8 - 432	149	26	64 - 625	292
Fluoride	0	< .1 - .34	<.161	7	.17 - .36	.226
Hardness	5	2.6 - 494	196	8	223 - 536	340
Ammonia	13	.145 - 94	18.8	28	1.6 - 890	52.3
Nitrate	10	< .02 - 2.5	< .60	21	< .02 - 3.6	<.73
Kjeldahl N	12	2 - 115	24.1	16	9 - 198	49
COD	12	49 - 227	149	12	118 - 344	195
Tannin & Lignin	5	7.5 - 20	5.0	7	8.4 - 20	14.3
As (T)	3	0 - .014	.009	5	0 - .012	.008
Cd (T)	3	0 - .0005	<.00033	3	0 - .0005	.0003
Ca (D)	5	13.9 - 104	46.98	8	34.9 - 107	63
Cr (T)	3	0 - .017	.008	5	0 - .02	.011
Cu (T)	3	.01 - .06	.03	5	.01 - .6	.04
Fe (D)	8	.5 - 6.5	4.14	20	.4 - 11	4.1
Fe (T)	6	.7 - 9	6	5	5.1 - 17.5	9.04
Pb (T)	5	0 - .007	.004	7	0 - .012	.006
Mg (D)	6	9.8 - 57	31.6	9	33 - 65.4	44.7
Mn (T)	6	.4 - .83	.543	15	.26 - .73	.47
Hg (T) µg/l	4	< .05 - <1.0	< .29	4	.05 - <1	< .29
Ni (T)	2	.02 - .03	.025	2	.02 - .03	.025
Na (D)	6	11.8 - 310	137	6	220 - 314	243
Zn (T)	3	.05 - .09	.067	5	.06 - .08	.066
Al (D)	4	.03 - 1	< .28	4	.01 - <1	.285

Colour.................TCU
Specific Conductance...mmho/cm
pH.....................pH units
All others.............mg/l

T = Total
D = Dissolved

The waters of Crescent Slough are not known to support salmon; but catfish, trout, and carp do inhabit the slough. The effect of the landfill leachate on this receiving water is undefined.

3.1.5.3 Future sewer connection. When the revised drainage system has been put in place, the discharge will eventually be directed to the Annacis Island STP. For the initial design trial period, the collected leachate is being pumped to the ditch which flows to Crescent Slough in order to determine design flow volumes and variations. Following this assessment period, the collected leachate flow will be connected by force main to the sewer interceptor which will follow an alignment close to Crescent Slough. With this modification, the receiving environment will become the Fraser River at the Annacis Island STP outfall. The alteration of the leachate through the sewerage and the primary treatment works is difficult to determine. Possible factors are adsorption, complexing precipitation and settlement which may occur both in the sewerage conveyance works as well as in the treatment plant. Hence, a portion of contaminants associated with the leachate discharge could be expected to be tied up in the digested sewage sludge. The disposition of leachate contaminants is difficult to partition and should be considered in conjunction with the overall mass balance of the Annacis STP solid and liquid flows.

No information was found in the literature which would suggest leachate constituent removals in a primary sewage treatment plant. However, a considerable amount of work has been done on the physical-chemical treatment of leachate. COD removals were typically about 20% with heavy chemical additions, whereas metal removals were much higher, often 90% or greater (Boyle et al, 1974; Bjorkman, 1979). It is unlikely that the incidental removal of leachate organics would occur to any degree in passing through the sewerage works, whereas some metal removals might be anticipated.

3.1.5.4 Loading estimate. As was stated earlier, flow data are only available for the period extending back to December, 1978. At the same time, the qualitative data base for 1979 is incomplete, notably for ammonia, chloride and calcium.

In calculating the loadings from the Burns Bog Landfill, the following data are used: (1) the 1979 flow data of 0.007 m^3/sec as the leachate flow from the site (This may be low over the long term, but is the only firm value at this time.); (2) the average concentration values of leachate constituents for sites 13 and 17, prior to double ditching; and (3) a dilution ratio of 4.4. The ratio of 4.4 was calculated from mass balances using average COD and total iron values for undiluted leachate (City of Vancouver, 1979), diluted leachate (EQUIS, 1978), and bog water prior to mixing (EQUIS, 1978). City of Vancouver data (1979) indicates that 46% of the leachate flow comes from the north and west perimeter ditch (site 17) and 54% from the south and east perimeter ditch (site 13).

Presented in Table 3.1.4 are the average daily constituent loadings based on the above information. Also provided are comparative 1979 values for COD, total iron and total zinc calculated on the limited monitoring of the double ditch collected leachate.

3.1.6 Intended Use. The intended future development of the Burns Bog Landfill site will be to provide diversified recreational facilities. It is expected that portions of the completed fill will be developed in the near future. Recent discussions have been held to examine the possible development of a motorcycle trail riding facility. Under the terms of the agreement between the City of Vancouver and the Corporation of Delta, the completed landfill will revert to Delta on completion. Hence, the character and development of future recreation facilities are solely the responsibility of the Municipality of Delta.

TABLE 3.1.4 BURNS BOG LANDFILL - CONSTITUENT LOADINGS

Parameter	Constituent Loading (Kg/day) Leachate Diluted 4.4 times		Contributed by Bog Water	Leachate Total	Constituent Loading (Kg/day) 1979 Undiluted Leachate
	Site 13	Site 17			
COD	320	350	300	370	380
Chloride	353	385	56	682	
Fluoride	.24	.26	-	.60	.11
Calcium	124	54	1	177	
Ammonia-N	103	140	14.3	229	
Nitrate-N	.90	2.0	.1	2.8	
Kjeldahl-N	110	144	27.5	227	
Tannin & Lignin	21.2	24.2	29.3	16.1	
Al(D)	.57	.20	.48	.29	.23
As	.013	.018	-	.031	0
Cd	NC	NC	-	-	0
Cr	.021	.013	.007	.027	0
Cu	.017	.007	.05	0	.006
Fe	10.9	5.7	.47	16.1	14.3
Mn	.95	.33	.14	1.14	.7
Ni	.021	.018	NC	.039	.06
Pl	.028	.007	.032	.003	.06
Zn	.19	0.64	.038	.79	.26

D= dissolved

NC=Not calculated

3.2 Richmond Landfill

3.2.1 General. The Richmond landfill site is located in the Municipality of Richmond on the main arm of the Fraser River (Figure 3.2.1). The landfill site is comprised of two parcels; a 125 ha active site and an adjacent 140 ha proposed future landfill site. The site is Federal Crown Land under the control of the Fraser River Harbour Commission (FRHC), and is being developed as a future port facility and industrial park. The FRHC acquired the first 125 ha parcel in 1971 from Richmond Municipality and others through a land exchange arrangement. Richmond Municipality had operated the southern portion of the 125 ha site and the dyke area as a municipal dump since 1965. The FRHC acquired the 140 ha parcel in 1975 as back-up land for their refuse filling operation and integration into the future development plans.

The landfill operation has been carried out since 1971 by Richmond Landfill Ltd., under an agent's agreement with the FRHC. Richmond Landfill Ltd. also own an adjacent 166 ha site, Figure 3.2.1., on which limited filling using demolition and clean fill was initiated in 1976.

3.2.2 Physical Description. The Richmond Landfill site is located on a peat bog adjacent to the main arm of the Fraser River and across from Tilbury Island. The geology is typical of much of the Fraser River Delta: 0 to 4.9 m of peat underlain by 0.9 to 7.3 m of silt and clay underlain by a thick unit (30 m +) of deltaic sands. In the north-eastern corner of the site there is a geologic discontinuity where there is no peat as a result of a former distributary channel. The southeast corner of the 125 ha site also appears to have an absence of surface peats. Figure 3.2.2 shows a geologic fence diagram for the 125 ha site and Figure 3.2.3 a typical geological cross-section of the 125 ha site.

The complex hydrogeologic factors influencing this site have been well documented by various researchers. Generally, the regional water table, like most peat bog sites, is near the surface for most of

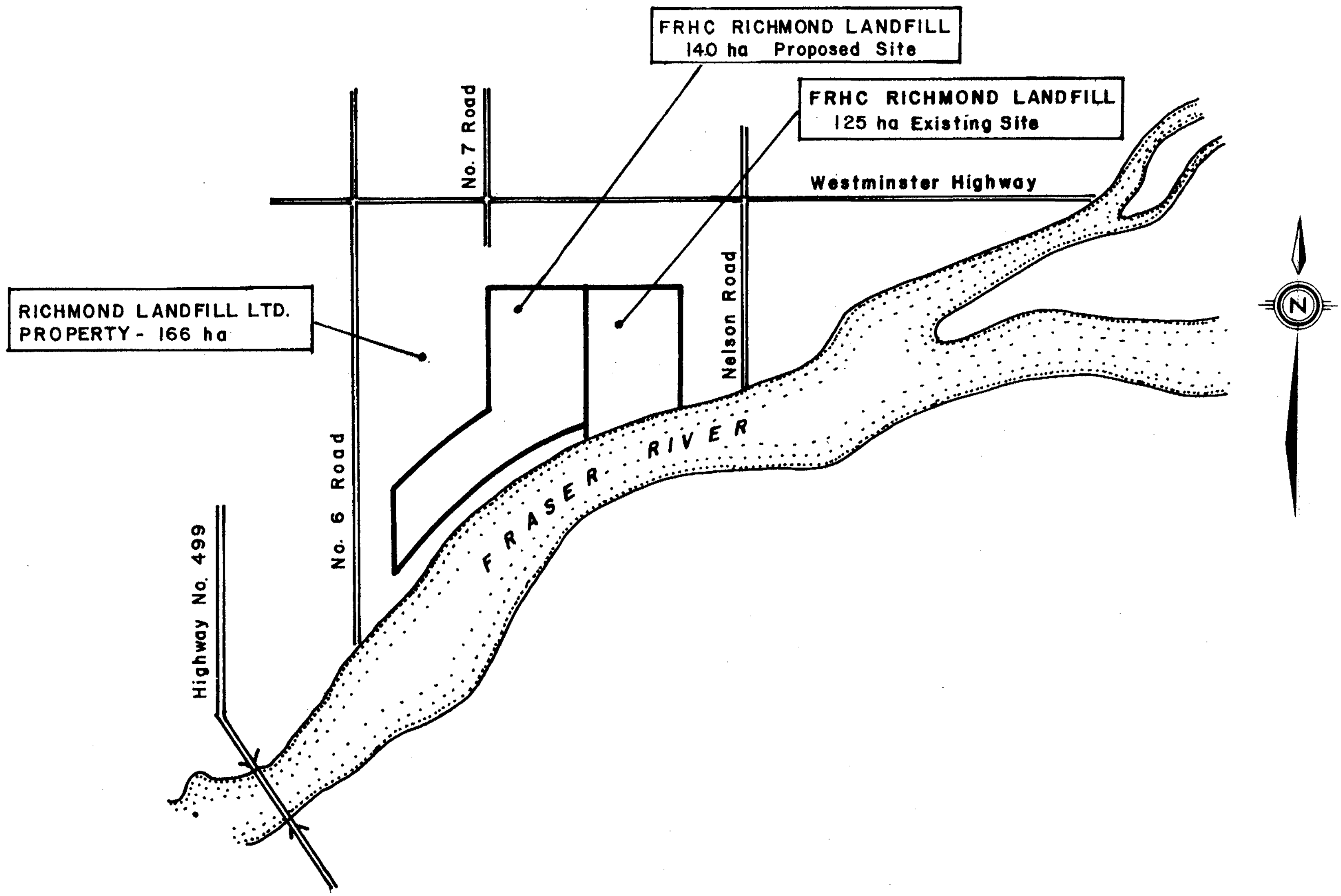


FIGURE 3.2.1 RICHMOND LANDFILL SITE PLAN

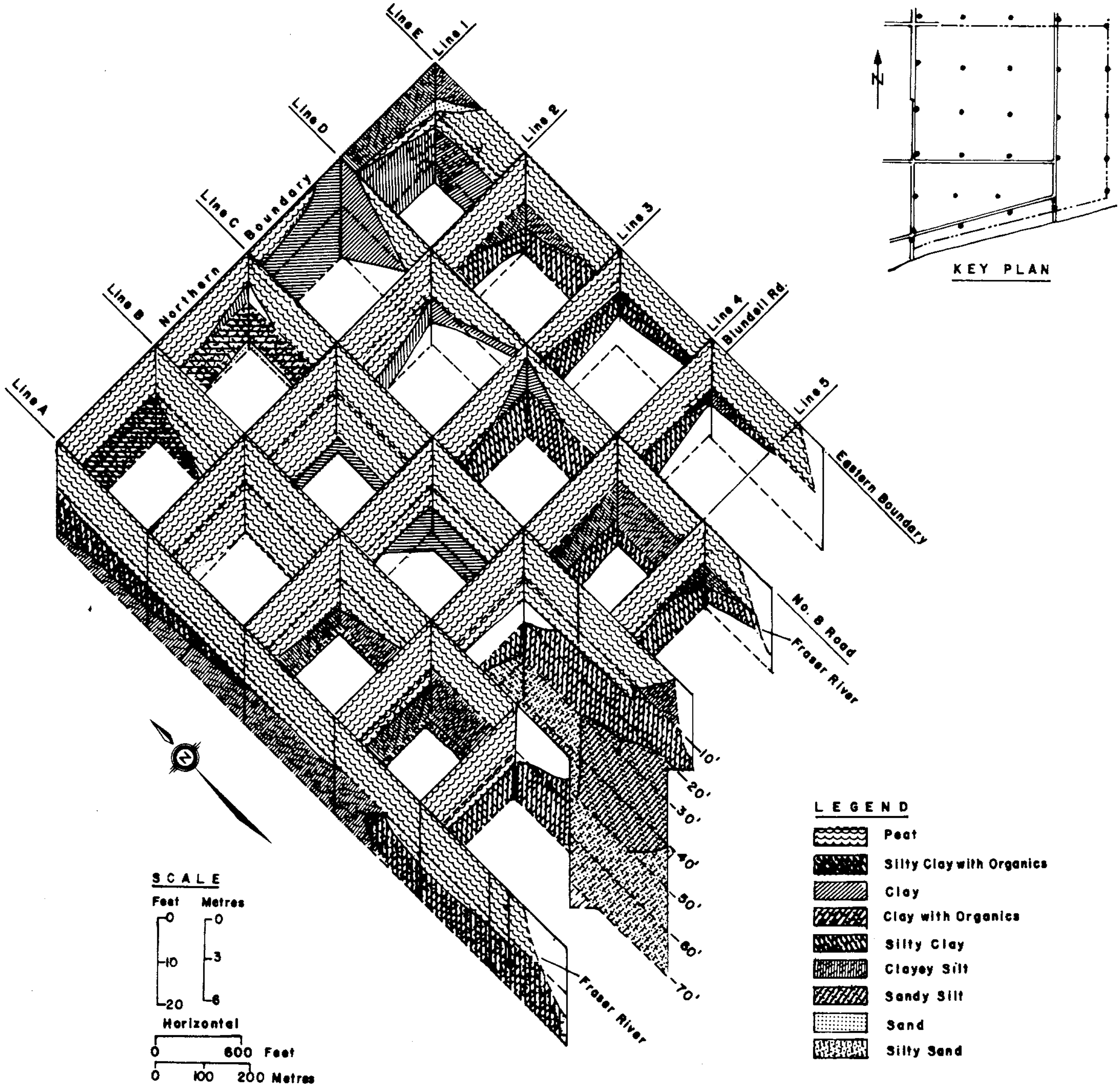


FIGURE 3.2.2 GEOLOGICAL FENCE DIAGRAM - RICHMOND LANDFILL SITE 125 ha.

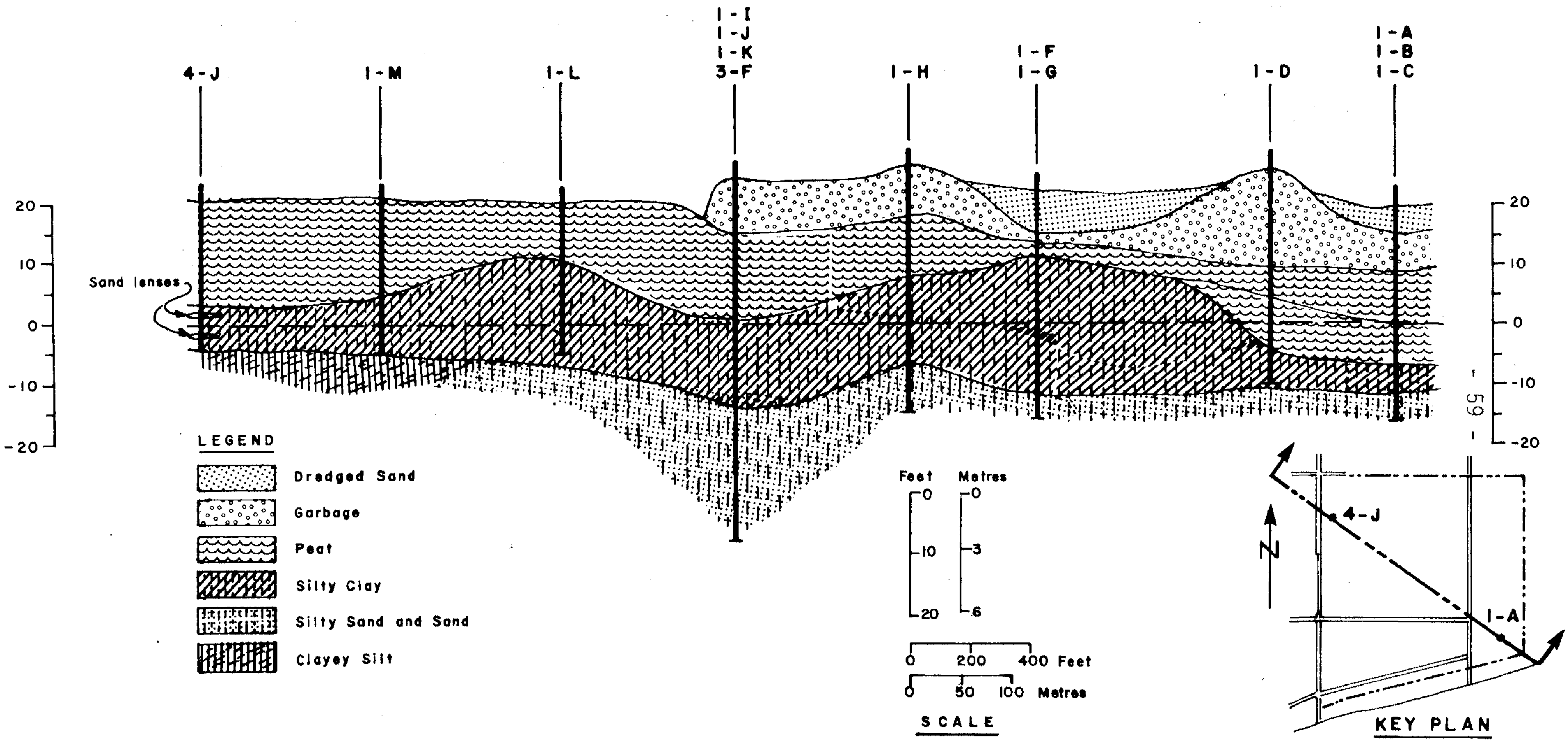


FIGURE 3.2.3 GEOLOGICAL CROSS-SECTION - RICHMOND LANDFILL 125 ha SITE

the year. As a result of compression from refuse filling, the peat horizon on the filled portion of the site is generally below the river level. Consequently the lower portion of the refuse is submerged below the original water table level and becomes inundated with groundwater. The load-caused restricted permeability (10^{-3} to 10^{-5} cm/sec) and subsequent increased piezometric pressures in the peat, yield a hydrologic situation which is predominantly influenced by surface factors. As at Burns Bog, the reduction in peat permeability together with the underlying clay unit, provides an impermeable basin effect, generally preventing the sub-surface vertical migration of leachate. The site leachate will migrate along the load-caused impermeable compressed peat contact. Water inputs to the fill flow through the lower layer of refuse in the form of leachate, and have little or no contact with the natural geologic units. The hydrographs for the silty-sand, peat, and refuse units shown in Figure 3.2.4, clearly illustrate the independent nature of the three main water bearing units. Note that the sand unit and refuse unit both have diurnal tidal response, whereas, the peat has a constant but significantly increased piezometric level due to refuse loading.

A notable exception to the above generality is the distributary channel in the northeast corner of the property and, to some extent, the southeast corner of the 125 ha site where the silty clay unit is expected to form the impermeable layer. In the northeast corner there is no peat on the surface and only a thin silt clay unit exists. Leachate loss to the permeable coarse sands and hence to the deltaic sands and underlying regional aquifer, resulted from excavation through the silty clay unit and penetration of the coarse distributary channel sands. This mechanism of leachate loss was postulated in the EPS report (Soper et al, 1977) and was confirmed by the Golder Associates' study (Golder, 1977). Recently, a cut-off ditch has been constructed along the peat contact of the distributary channel in order to divert leachate flow from this permeable zone. Figure 3.2.5 shows the present leachate ditch alignment and flow patterns.

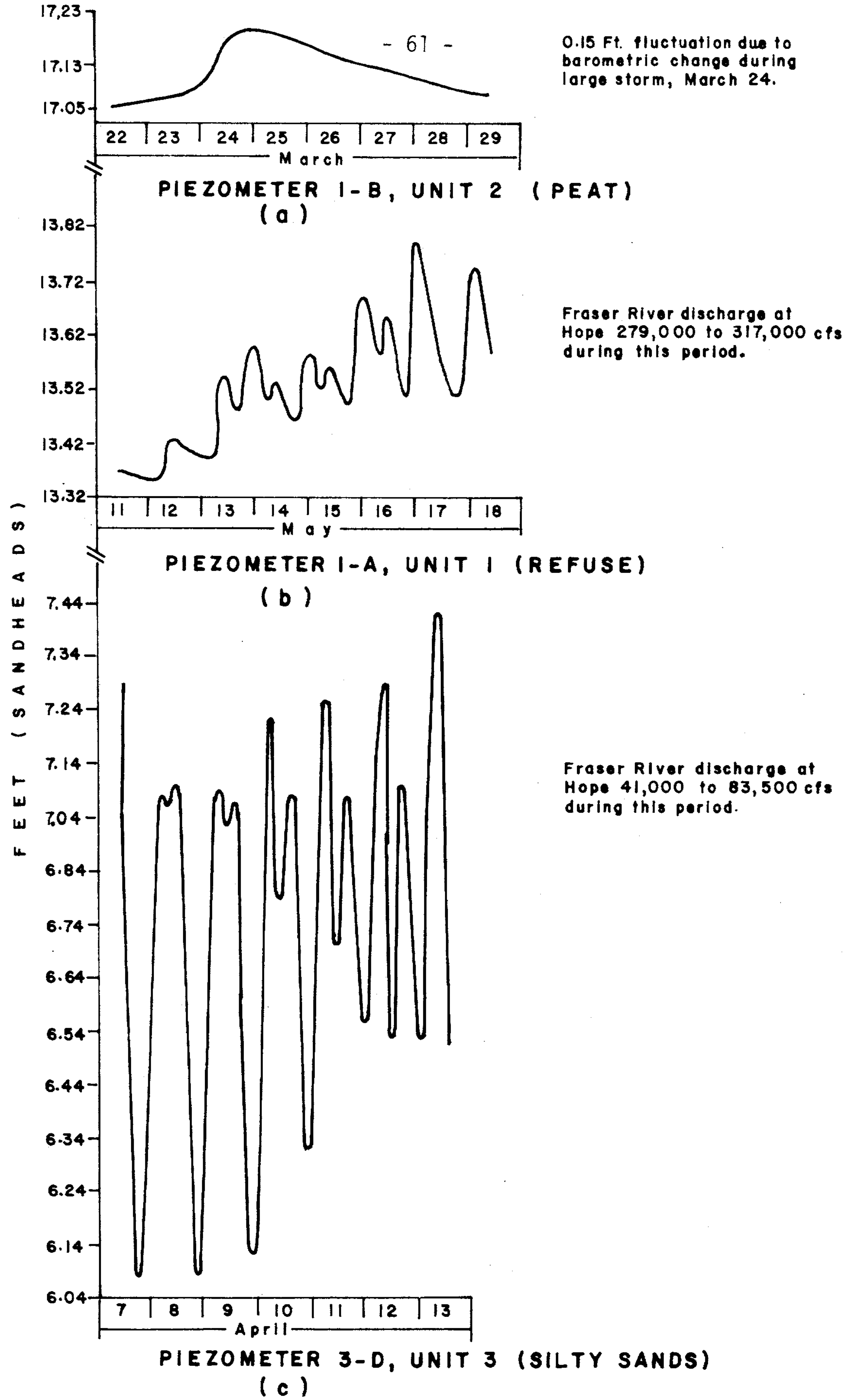


FIGURE 3.2·4 RICHMOND LANDFILL - TYPICAL HYDROGRAPHS FOR REFUSE, PEAT AND SILTY-SAND UNITS

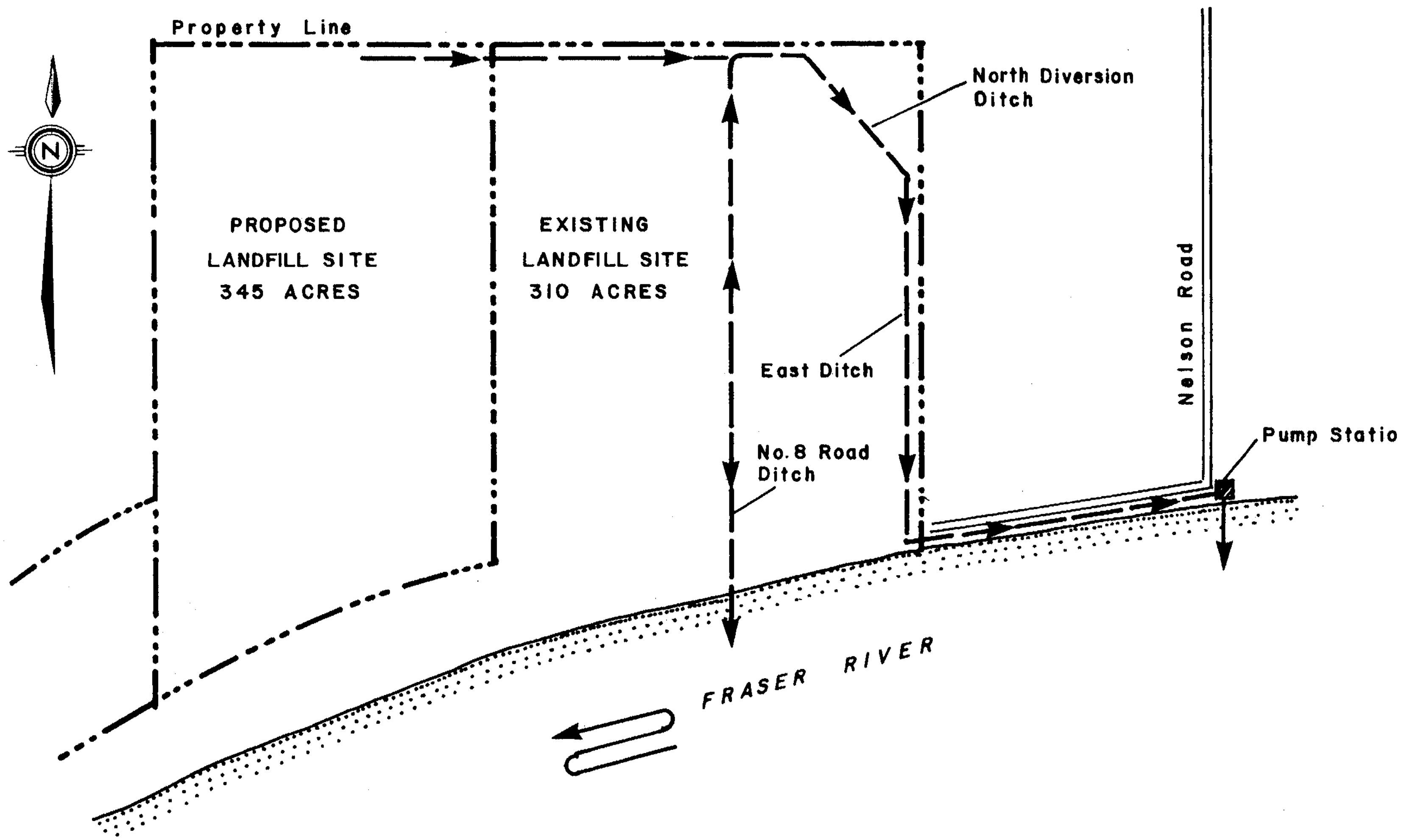


FIGURE 3.2.5 RICHMOND LANDFILL DITCH FLOWS

Notwithstanding the sub-surface loss of leachate through the northeast corner geologic discontinuity, the main leachate flow is a surface phenomenon with the Fraser River being the principal receptor. The surface hydrology of this site has been in a state of flux over the years as a result of the dynamic nature of a landfill operation. Generally, leachate has been intercepted by surface ditches and conveyed either directly or via agricultural and municipal ditching to the Fraser River. At present, the main leachate discharge is via the pump station at the foot of Nelson Road.

3.2.3 Operation. Refuse discharged to the Richmond Landfill site is from the Corporation of Richmond and from commercial haulers and private vehicles from Greater Vancouver as well as from a Fraser River barge unloading operation. As there are no weigh scales at this site, quantification of the source contribution is difficult. Based on estimates (Soper et al, 1977), the total annual tonnage is in the order of 435 000 tonnes. Excluding barge wastes, granular materials and ditch cuttings, the annual estimated tonnage is in the order of 245 000 tonnes. One-third to one-half of this is thought to emanate from the Richmond Municipality, with the balance from the Greater Vancouver area (primarily the City of Vancouver).

The filling operation generally consists of placing first a lift mattress fill, 1.2 to 2.1 metres, followed by a second refuse lift of 2.4 to 4.3 metres. The mattress lift components have, in the past, included the full myriad of refuse components and were not restricted to specifically "non-leachable inert" materials. The upper refuse lift is comprised primarily of municipal and commercial compactor refuse.

Compaction of the mattress fill is achieved by a crawler tractor spreading the refuse materials out onto the native peat bog. The method of compacting the upper refuse lift has varied over the life of the operation. Originally, and as reported in the EPS Assessment Study, compaction was applied only to the top of the lift by a steel wheeled compactor and there was no regular cell construction or face compaction. Richmond Landfill Ltd. has since modified its method of lift construction to the more conventional area cell method with sloped working face compaction.

It is believed that the old method of construction could have markedly affected the in-place density of the fill. The environmental and structural advantages or disadvantages of placing the refuse in this manner are not readily predictable, but it is generally believed that the disadvantages would include reduced structural integrity and a higher rate of refuse leaching.

Liquid wastes were discharged to the site in what was believed to be significant quantities. This practice ceased in February 1977. The sand cover used at the site is primarily obtained from a suction dredge deposit. Wastes discharged via the barge unloading ramp have been sporadic and have been generally dredge spoil and log pond wastes.

The fill rate of the Richmond Landfill site has been approximately 20 ha per year. The active 125 ha site has little capacity remaining and, as a consequence, plans are now underway to utilize the adjacent 140 ha site for refuse filling. This site (Figure 3.2.1) will have a refuse fillable area of approximately 73 ha; the reduction being due to the desired 460 metres setback in line with a railway easement reserved for clean structural fill. The life of the site will depend upon the amount of refuse handled and the operational practices. Negotiations presently underway between the Municipality of Richmond and the FRHC could restrict the deposit of refuse materials. If such restrictions occur, it is estimated that the useful life of this site could be extended up to possibly 15 years. If the present refuse volumes continue, the site life may be as little as five years.

The fill density of this operation is estimated to be in the order of 415 kg per cubic metre; however, this is at best only an estimate because as mentioned before there are no weigh scales.

A Pollution Control Permit (PR-5113) was issued in May, 1978 to impose controls on the operation of the active 125 ha site and to authorize an extension of the landfill into an adjacent 140 ha area.

The terms of the permit required studies to be undertaken on hydrogeologic features of the site, on leachate generation, collection and treatments, and on methods of operation to ensure adequate protection of the environmental and health aspects associated with the landfill. The permit was appealed and an amended permit issued in February, 1979. In accordance with the appeal decision, a notice of intent to cancel the permit accompanied the amended authorization, and the Fraser River Harbour Commission (FRHC) was advised that a new permit was required should a landfill operation at the site continue. In this respect, an application for permit should be supported by the necessary background assessments and should encompass the control of leachate and other related problems of the operation. The permit was cancelled in February, 1980, and the FRHC has been requested to submit plans for the final rehabilitation of the site and the long term control of leachate.

3.2.4 Leachate. Development of an accurate water balance for the Richmond Landfill site is difficult due to the complex and variable nature of the water inputs and outputs. The major influences on the site water budget are precipitation, Fraser River ingress through the permeable dyke, culverts with inoperative flap valves which cut through the dyke, suction dredge deposit of sand cover material, and the addition of pore water from compression of the in-situ peats. Based on estimates of these major influences, the average water input over the entire 125 ha site has been placed at 320 cm per year (Golder, 1977). This corresponds to an average outflow of 0.12 m^3/sec over the entire 125 ha site. As the Fraser River influence and the dredgate recharge are expected to be localized, the water input over much of the site would be expected to be considerably lower. It is reasoned that if only the southern portion of the site is influenced by river ingress, then average outflow associated with the remainder could be in the order of 0.052 m^3/sec. In addition, dredging operations tend to result in high flows over short periods and much of the flow bypasses directly without any refuse contact. The majority of the leachate outflow is discharged via the drainage ditch system to the Fraser River, although, until recently, a portion has been lost to the distributary channel sands and thence to the underlying regional aquifer and the surrounding peat bogs. Figure 3.2.6 shows a typical geologic cross-section with what was the complex water inflows and outflows.

Surface ditch flow patterns have varied over the life of the operation. Originally the No. 8 Road ditch outlet was the principal point of discharge, but through re-alignment and changes, the main outlet is now the east ditch which discharges to the Dyke Road ditch and to the Fraser River via the Nelson Road pump station. The most recent site ditching alteration involved construction of a cut-off ditch along the peat contact in the northeast corner, thereby precluding direct leachate discharge to the distributary channel sands.

Generally, based on the above budget outline, this site like other similarly located sites, has a leachate discharge which is predominantly intercepted and conveyed by surface drainage ditches which discharge to the Fraser River. It is felt, however, that because of the nature of the fill materials and their placement, together with the high site water inputs, a somewhat different type of leachate may result.

There are basically three main locations in which to sample the leachate: within the fill from monitoring wells; from spring discharges to the ditches; and in the ditches themselves. Presented in Table 3.2.1 are typical concentrations from these three locations. An extensive sampling and monitoring program was carried out in conjunction with the EPS Assessment Study in 1975 through 1977, and has continued on a limited basis. Some 40 wells installed in the refuse, peat, and sand hydrostratigraphic units, together with several ditch locations and spring discharges have been monitored. The results of this monitoring indicated that again with the exception of the northeast corner distributary channel, there is good protection against vertical migration of leachate to the underlying sands unit. The well and spring leachate characteristics are somewhat variable being highly dependent on the prevailing water inputs and fill reactions, but generally they contain the highest concentrations of constituents.

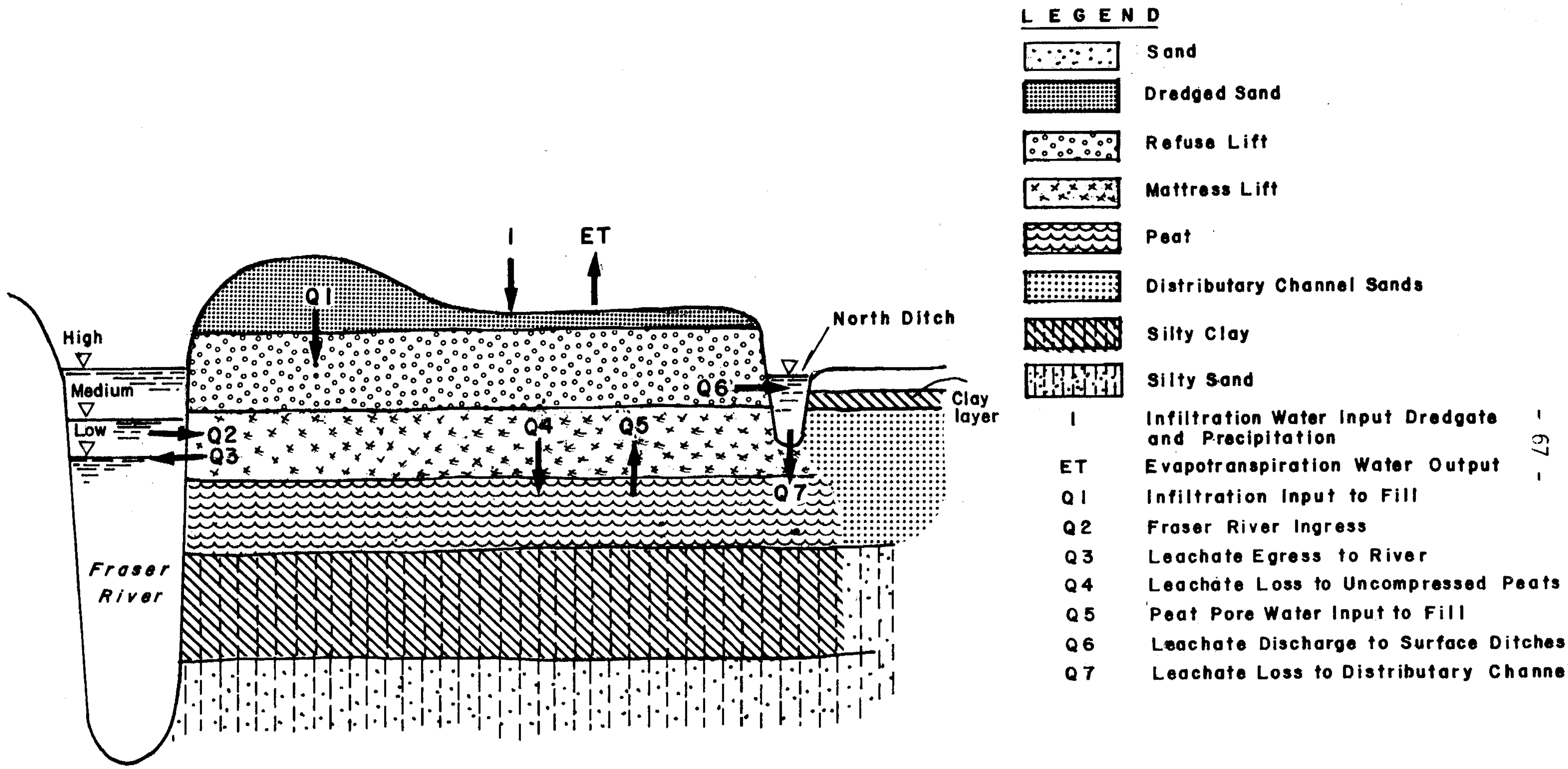


FIGURE 3.2·6 RICHMOND LANDFILL - COMPLEX WATER BUDGET INFLOWS AND OUTFLOWS

TABLE 3.2.1 SUMMARY OF LEACHATE MONITORING RESULTS - RICHMOND LANDFILL

Parameter (mg/l)*	Range of Values		
	Refuse Wells	Leachate Springs	East Road Ditch
COD	82-11000	1860-4720	160-1900
BOD_5	-	1140-2980	-
Total Carbon	26.0-2000	930-1830	-
Total Organic Carbon	18.0-1600	810-1600	95-300
Total Residue	500-6080	3190-6490	3130-4600
Total Volatile Residue	-	1470-2930	-
pH (units)	5.3-11.6	6.2-6.3	6.9-7.5
Acidity ($CaCO_3$)	-	540-790	-
Alkalinity ($CaCO_3$)	47-2200	1350-3050	800-1820
Kjeldahl N.	-	8.8-460	-
NH_3-N	2.0-79	0.3-37.5	13.0-66.0
Total Phosphate	-	3.1-4.7	-
Sulphate	4.0-500	83-250	76.5-77.5
Chloride	5.0-3000	125-390	382-1600
Sulphide	-	0.02-30	-
Boron	-	5.9-7.4	-
Calcium	30-770	535-1065	200-370
Sodium	7.7-1100	128-358	13-306
Potassium	-	51-137	-
Magnesium	6.0-150	39-84	-
Iron	2.8-490	1.6-22.4	1.8-25.8
Manganese	-	4.3-7.8	4.1-4.3
Zinc	0.05-0.97	0.55-1.3	0.09-0.82
Aluminum	-	0.36-1.26	0.43-0.80
Chromium	0.02-0.09	0.025-0.085	<0.02-0.04
Copper	-	0.010-0.050	<0.02-0.02
Nickel	0.5-0.1	0.002-0.012	< 0.20
Lead	-	0.023-0.051	<0.01-1.01
Cadmium	-	0.001-0.002	< 0.01
Selenium	-	0.013-0.018	-
Arsenic	-	0.006	< 0.20
Specific Conductance (μmho/cm)	560-10100	-	-

* except as noted

In addition to the analysis carried out by EPS for the assessment study (Soper et al, 1977), there has been an ongoing collection of leachate data by numerous groups: EPS, 1978 - leachate ditches; Civil Engineering, University of British Columbia, 1977/1978 - leachate springs; B.C. Research, 1979 - leachate ditches and leachate springs. Summary tables of these data along with sample site location maps are presented in Appendix C. A detailed discussion on the 1975 to 1977 EPS leachate data is available in the EPS Assessment Report on the Richmond Landfill.

The analyses indicate that there is a great variability in the leachate composition, depending on source, location and time. Generally it can be stated, as would be expected, that levels in the refuse well and raw leachate spring are the highest; whereas, the ditch samples show the results of considerable dilution by less contaminated site waters (runoff, Fraser River ingress, etc.).

In conjunction with the above generalizations, it is important to note that the Richmond Landfill leachate, although somewhat different from the other fills, has similar characteristics of low trace metal concentrations. In this regard, the high pH and alkalinity values would tend to support a hypothesis that metals are retained within the fill itself and are not carried in solution with the leachate.

Bioassay results collected during and after the EPS Assessment Program indicated that the ditch waters were generally acutely toxic with a 96-Hr LC_{50} in the order of 30% to 40%. Although the metal concentrations are not considered high, there are compounds in the leachate which give rise to concern. Total ammonia which is seen to be at high concentrations in some samples, may be a significant factor, since the presence of the un-ionized fraction can be extremely toxic at low concentrations. Recent research (Cameron, 1978b) has shown good correlation between toxicity and the un-ionized ammonia concentration in leachate from lysimeters. Another factor may be the mobilization of organic compounds from the fill. This would be expected to be enhanced at the higher pH values.

How long a leachate of measurable strength will emanate from the Richmond Landfill is difficult to predict. Given that the old Richmond Municipal dump portion of the landfill continues to produce leachate, a conservative estimate would appear to be in the order of 25 years although the nature of the filling operation and control water inputs will greatly influence the leachate generation.

3.2.5 Impact on Water Quality

3.2.5.1 Point of egress. The impact on water quality due to the Richmond Landfill discharge may be separated into the following receiving environment categories of leachate egress:

- local agricultural and drainage ditches;
- regional aquifer;
- Fraser River.

Over the years, leachate from the Richmond Landfill has directly entered the municipal drainage ditch system such as the East Road ditch, Dyke Road ditch, and Nelson Road connection; or, indirectly through agricultural drainage on the east and north of the site as a result of blocked or restricted drainage. The latter egress has been substantially reduced with time and now occurs infrequently.

The sub-surface egress of leachate via the distributary channel sands to the regional aquifer was postulated by the EPS Assessment Study (Soper et al, 1977) and confirmed by the Golder Associates Study (Golder, 1977). Investigations involved the installation and monitoring of piezometers and a pump test. The pattern of leachate egress does not follow the typical dispersed plume spread due to the variable tidal influences, but it is clear from the results that significant leachate contamination of the regional aquifer had occurred.

It is anticipated, that the installation of the north diversion ditch along the contact of the distributary channel will reduce the leachate egress considerably. In terms of specific environmental impact, the aquifer water quality appears to have been degraded but, as a result of the cutoff ditch, continued contamination should now be effectively halted. The aquifer was not used as a source of drinking water due to its poor quality and the availability of municipal water supplies.

Notwithstanding the above geologic anomaly and the resultant sub-surface leachate discharge, the primary leachate discharge is to the Fraser River either directly via surface ditches or indirectly through a diffuse dyke discharge. Plate 3.2.1 shows the black leachate plume entering the Fraser River at the Nelson Road outlet to the east of the landfill site. The plume is the result of the direct surface ditch discharge of leachate. A diffuse leachate discharge to the river also occurs through the permeable dyke and is evident during low river levels. The No. 8 Road ditch outlet, which in the past was a central discharge point from the site, now involves a small drainage area.

3.2.5.2 Defined receiving water. The Fraser River is the receiving water of primary concern. Leachate enters via the main discharge at the Nelson Road outlet and the lesser discharges at the No. 8 Road ditch outlet and through the dyke.

The surface ditches which convey the leachate to the Fraser River (e.g., Dyke Road ditch) are also receiving waters. In many cases these ditches do support some fish species. In the past, where there were discharges to the local drainage ditch network, particular environmental health concerns were related to potential human and animal contact. At the present however, with ditch realignments, these uncontrolled discharges do not exist.

A leachate ditch monitoring program initiated by the B.C. Research Council in January 1978, at the request of Richmond Landfill Ltd., provided some insight to the major system of leachate egress and the quality of the water in that system. The program involved the establishment of surface ditch sampling sites and monthly monitoring for indicator parameters. The sampling results are summarized in Table 3.2.2. These more recent ditch sampling results again point to the high variability of contaminant concentrations with time and location. The data indicated that the principal north ditch - east ditch - Dyke Road ditch - Nelson Road outlet flow pattern, has a generally decreasing trend in contaminant concentration with distance east, as a result of

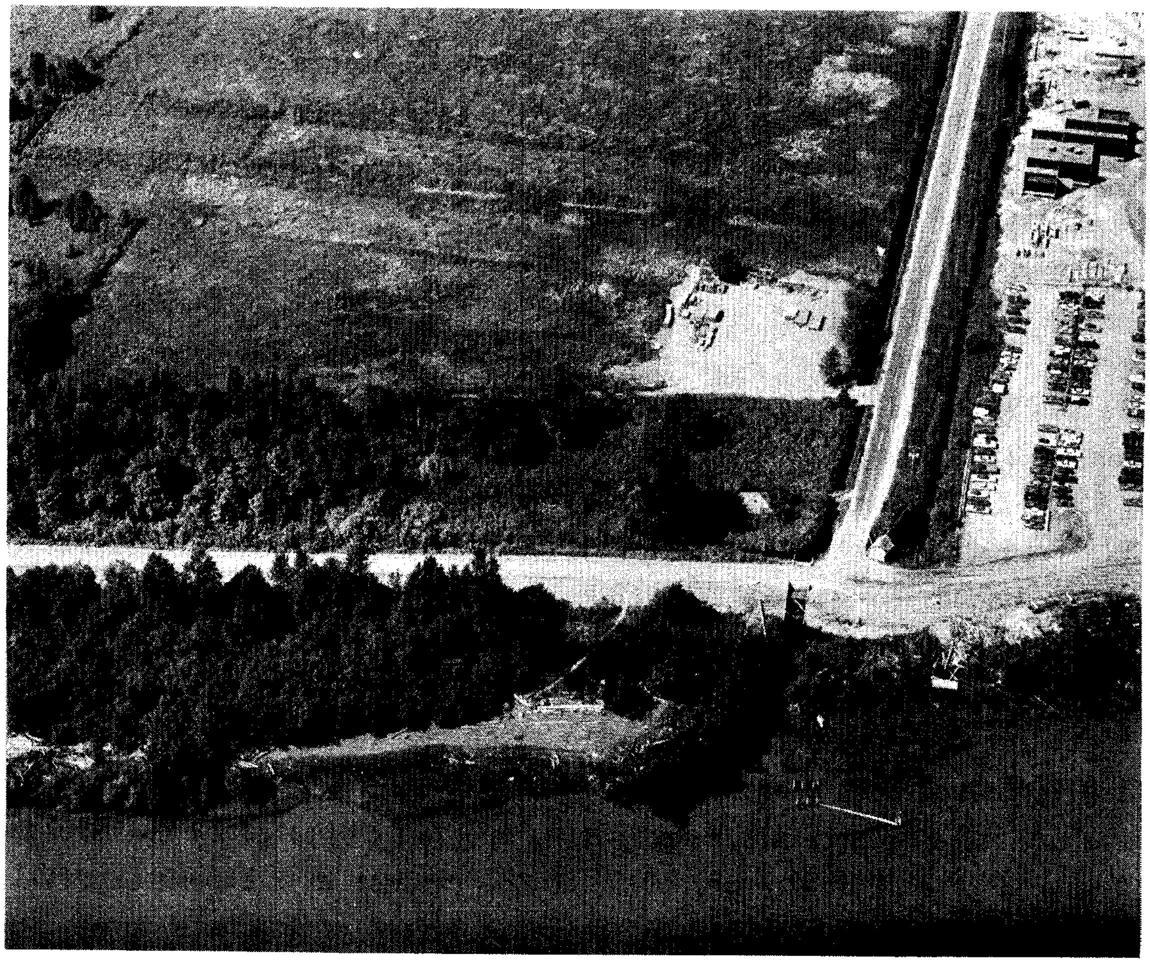
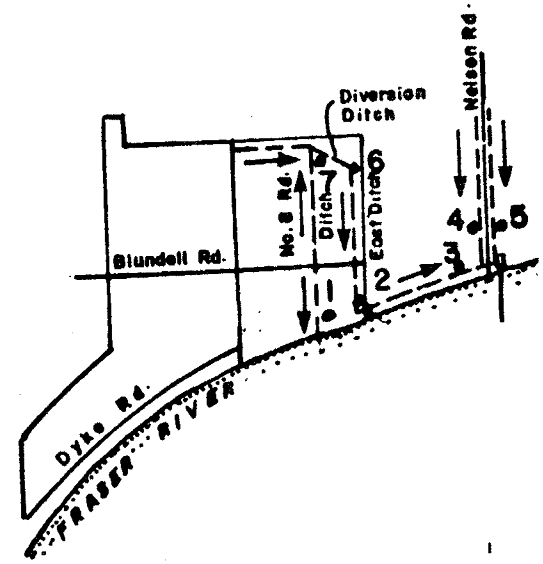


PLATE 3.2.1 NELSON ROAD OUTLET, RICHMOND LANDFILL - BLACK LEACHATE PLUME ENTERING THE FRASER RIVER

TABLE 3.2.2 B.C. RESEARCH COUNCIL - 1978 MONITORING PROGRAM - RICHMOND LANDFILL

Parameter	Site 1				Site 2				Site 3			
	No.	Min.	Max.	Avg.	No.	Min.	Max.	Avg.	No.	Min.	Max.	Avg.
DO (mg/l)	6	.1	11.9	7.1	5	.1	2.6	.74	5	.3	4.4	3.8
pH	6	6.6	7.4	6.9	5	6.9	7.1	7.0	5	6.7	7.2	7.0
Spec. Cond. (μmho/cm)	6	73	2250	793	5	900	3300	2060	5	370	2300	1200
Sodium (mg/l)	6	2.9	225	78.2	5	72	246	183	5	27	220	89
Chloride (mg/l)	6	2.2	342	127	5	30	310	199	5	30	525	155
Iron (mg/l)	6	.1	10	2.2	5	1.0	9.4	4.4	5	2.6	8.7	4.7
Org. Carbon (mg/l)	6	4	10	7.7	5	50	325	175	5	15	200	85
T.Alk. (mg/l $CaCO_3$)	6	41	660	226	5	310	1480	949	5	100	1090	473
Ammonia-N (mg/l)	6	.52	17.8	4.8	5	9.5	48.7	28.6	5	.41	304	11.5
Bioassay (96 hr TL_m)	1			100%	5	17.5	93	53%	5	100	45	89%



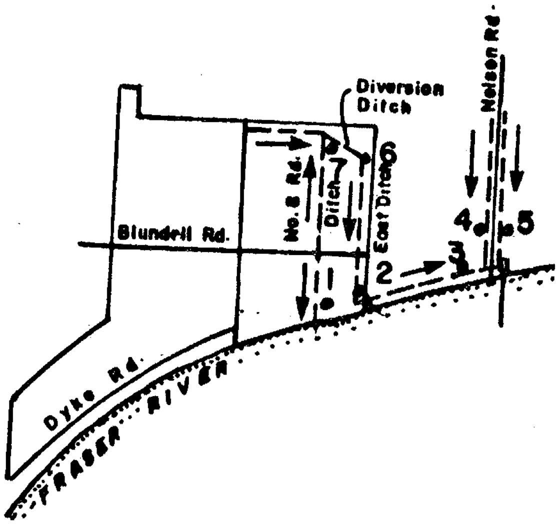


TABLE 3.2.2 B.C. RESEARCH COUNCIL - 1978 MONITORING PROGRAM - RICHMOND LANDFILL (continued)

Parameter	Site 4				Site 6				Site 7			
	No.	Min.	Max.	Avg.	No.	Min.	Max.	Avg.	No.	Min.	Max.	Avg.
DO (mg/l)	5	.4	4.7	2.9	5	.1	1.5	.6	5	.1	4.6	1.74
pH	5	6.6	7.1	7.3	5	7.0	7.1	7.1	5	6.7	7.1	7.0
Spec. Cond. (μmho/cm)	5	250	1400	660	5	1050	3200	2270	5	880	4200	2200
Sodium (mg/l)	5	19.4	117	534	5	72.5	260	194	5	63	225	217
Chloride (mg/l)	5	38.5	111	65.3	5	102	354	227	5	35	440	261
Iron (mg/l)	5	2.6	6.7	5.6	5	1.1	13.2	5.6	5	.1	3.8	.92
Org. Carbon (mg/l)	5	16	120	55	5	63	225	143	5	50	100	65
T.Alk. (mg/l $CaCO_3$)	5	77	660	282	5	495	1440	1075	5	400	1450	845
Ammonia-N (mg/l)	5	.47	19	6.6	5	19	61.6	43.3	5	10.5	54.9	29.2
Bioassay (96 hr TL_m)												

dilution effects. The site leachate ditches represented by sample sites No's. 2, 6 and 7 show generally consistent contaminant levels. Variations at these points are primarily attributable to variable dilution effects due to site water inputs.

The No. 8 Road ditch outlet results show the continuing trend of attenuation due to reduced drainage area, and to the flushing effects of the Fraser River and dredgate water.

3.2.5.3 Fraser Estuary. Much discussion has occurred regarding the Richmond Landfill site and its resultant environmental impacts. Clearly in a receiving environment like the Fraser River with its large flows and resultant dilution, quantification of specific changes in the chemistry of the water column is at best difficult if not impossible.

In terms of specific contaminant concentrations in the leachate discharge, the prominent characteristics are oxygen demand, nitrogen forms, and possibly iron and manganese. As with most landfill leachates in the study area, the concentrations of the trace metals are below those which would be considered to be acutely toxic to fish, given the pH and hardness of the leachate. On the other hand, the metal concentrations in the leachate are such that they are at or exceed threshold limits which could stress fish. Leachate analysis was generally not carried out for specific organics, and while there is the probability of organic contaminants being present in leachate, it is not reasonable to infer an impact in the absence of qualitative and quantitative data.

The 96-hr LC_{50} bioassay results can provide some insight as to the acute toxicity of the leachate but provide little information on chronic or accumulative effects. Recognizing the difficulties in measuring changes in the chemistry of the water column, the principal impact of the Richmond Landfill leachates on the Fraser River that one can see is the black plumes coming from the Nelson Road and No. 8 Road ditches.

A limited benthic sampling program of Fraser River sediments was carried out by EPS adjacent to the Richmond Landfill and the Nelson Road outlet (EPS, 1977). Little can be concluded from the study other than there were generally higher levels of lead and zinc in these sediments than found elsewhere in the river (EPS, 1976, 1977; Westwater, 1972). It was not possible to establish a direct cause and effect relationship between the elevated metal levels and the presence of the leachate. A summary of the data along with sample site location is presented in Appendix C.

To date, the discharge to the unconsolidated peats has not been quantified, but it is of less direct concern due to the renovation and attenuation capabilities of the peat. The distributary channel leachate loss is now expected to be greatly reduced due to the diversion ditch alignment. Resultant attenuation is expected, but it will be slow as a function of contaminant plume dilution and dispersion, rather than contaminant renovation. Accordingly, leachate losses by sub-surface means are viewed as manageable anomalies to the principal discharge to the Fraser River.

3.2.5.4 Leachate loadings. The estimated mass loading of constituents in the leachate from Richmond Landfill are presennted in Table 3.2.3 and are based on average ditch sample results and an annual average discharge flow rate of 0.052 m^3/sec. Due to the probable large variability in both of these values, the mass loading values must be considered as a 'best guess', and should be treated accordingly.

Notwithstanding the above qualifications, it is seen that the Richmond leachate discharge has an annual contaminant mass loading to the Fraser River in the same order of magnitude as the Burns Bog site, considering that the filled area of Richmond Landfill is 3 times as great.

3.2.6 Impact on Other Environments. Beyond the water quality impacts of the leachate discharge there are only minor concerns with the Richmond Landfill operation.

TABLE 3.2.3 ESTIMATED DAILY LEACHATE LOADING TO FRASER RIVER - RICHMOND LANDFILL

	Estimated Average Concentration (mg/l)	Daily Loading (kg/day) Based on 0.052 m^3/sec. Annual Flow Rate
Organic Carbon	120	540
Sulphate	75	336
Chloride	300	1350
Ammonia (N)	50	225
Total Residue	3000	13500
Non-filterable Residue	80	360
COD	700	3150
Copper	$<$ 0.02	$<$ 0.09
Iron	15	67
Lead	$<$ 0.01	$<$ 0.04
Zinc	0.3	1.3
Sodium	250	1120
Manganese	5	22
Aluminum	0.5	2.2
Chromium	0.03	0.13

The native peat bog is inhabited by a variety of animals and wild fowl with many ducks and some deer being observed. In this regard, ducks and muskrat are often observed in the leachate ditches. With the reduction of habitat by filling, it is expected that the existing small deer population will be affected. This, of course, is the case with all habitat encroaching development and is not particularly related to landfilling. Other environmental concerns may include the potential access of livestock and children to the leachate in the local drainage ditch system. Health vectors related to sea gull transport of waste materials and rat populations are common to all landfills and have not been noted as a particular problem at this site.

There is some evidence that landfills may raise the local water table as a result of hydrogeologic alteration. This could possibly affect crop growing conditions. As well, leachate migration to neighbouring lands may be a potential concern to crop growth and stress due to possible phyotoxins (e.g., boron) in the leachate. These local agricultural factors have, to-date, not been in evidence adjacent to the Richmond site.

3.2.7 Intended Use. As previously noted, the intended use of the completed Richmond Landfill site is a deep sea port facility and industrial park under the control of the Fraser River Harbour Commission. The structural integrity of this site is paramount in the day-to-day operation as it relates directly to future potential structural problems; for example, inadequate bearing capacity due to differential settlement. Other areas of concern are explosion hazards in buildings constructed on the fill and the corrosion of foundations and utilities. The specifications of the site operation laid down by the Federal Department of Public Works acknowledge many of the future structural problems and attempt to provide operational procedures which will avoid future problems.

3.3 Port Mann - Surrey Landfill

3.3.1 General. This landfill located in Surrey on the south bend of the Fraser River has had an on-again, off-again operational existence. When filling of the site was initiated in 1969, the life expectancy of the site was some 25 years (AESL, 1974); however, in 1976, this was effectively cut to two years because of the expropriation of some 21 ha by the Canadian National Railway. At that time, the Port Mann site was expected to close at the end of 1978. Surrey Municipality has now determined that by modifying the fill operation, specifically the depth of fill, a further six to eight years of operation (Surrey, 1978) can be anticipated. Plate 3.3.1 is an aerial photograph of the site.

3.3.2 Physical Description. The Port Mann Landfill is located on a peat bog adjacent to the Fraser River and is immediately upstream of the Port Mann Bridge. The Surrey uplands rise abruptly from the south side and, on the north, the site is hemmed in by the CNR Port Mann surge yard. Access to the site is from the west underneath the Port Mann Bridge.

Prior to 1976, the Corporation of the District of Surrey's property extended to within 61 metres of the Fraser River (to the edge of the original CNR mainline right of way). This area was filled prior to 1974 and some 2.7 ha of old fill was included in the CNR expropriation. Figure 3.3.1 shows the site plan of both the original and present property lines and fill areas.

The site is located on a terrace-like transition zone between the Surrey uplands, the south wall of the Fraser Valley in this instance, and the Fraser River. The site is generally underlain by a sequence of 1.2 to 7.0 metres of peat, over organic silts and/or organic sands, over non-organic silts, sands or sands and gravel, or a sequence of peat over sand or sand and gravel (R.M. Hardy, 1978). A peat, silt, sand sequence is generally maintained north of the present site to the river, (AESL, 1974, and W. Jansen, CNR personal comm.) with the silts extending to as much as 21 metres. The information from the Port Mann Bridge studies indicates a sensitive marine clay at 36.6 metres (AESL,

PLATE 3.3.1 PORT MANN LANDFILL SITE

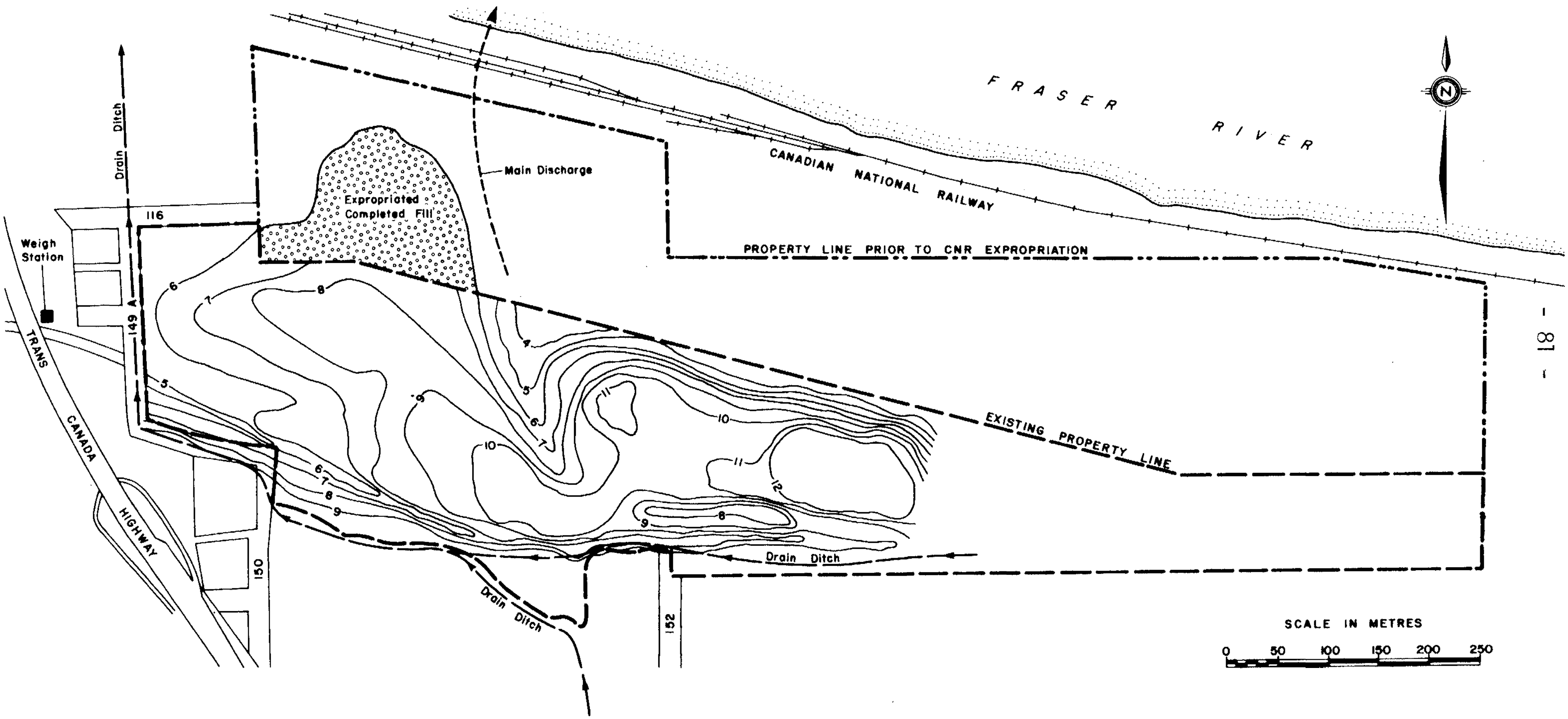


FIGURE 3.3.1 SITE PLAN OF THE ORIGINAL AND PRESENT PROPERTY LINES PORT MANN LANDFILL - DISTRICT OF SURREY

1974), whereas, one deep hole on site found clay at a depth of 23 metres. This geology is generally correlatable to east of the Surrey Bend area (UNIES Ltd., 1973).

Figure 3.3.2 depicts the location of the various drill holes in the site area, while Figure 3.3.3 is a geologic fence constructioned primarily from the R.M. Hardy drill hole information. Included with the geology are the logs of four holes drilled in 1972 (Surrey, PCB files, 1972).

Where refuse has been placed on the peat, an expected settlement has occurred. In those areas where the adjoining peat has not been loaded, the "classic" peat dish has formed. Shown in Figure 3.3.4 are two sections through Surrey's portion of the property; one section where adjacent filling has taken place, and the other where it has not.

The presence of the underlying organic silts in conjunction with the low permeability induced in the peat by refuse loading, will limit the vertical migration of leachate in those areas of the fill. However, in the areas where peats were over sands or gravels, vertical migration of some leachate ahead of refuse placement could have occurred. It would be expected that once the peats are compacted the vertical migration of leachate would be limited. Leachate migration from the fill is expected, for the most part, to be along the peat refuse contact. There is some evidence, where filling is preceding over the old clay capped part of the landfill, that leachate is coming out along this elevated refuse clay contact.

3.3.3 Drainage. Surface water off the south slope opposite the site is picked up by a surface ditch and diverted to the west around the site. It has been suggested that some seepage loss from this ditch to the fill could occur (AESL, 1974).

Surface water off the south slope east of the present fill is, for the most part, culverted through CN property to the Fraser River. However, some of this surface water flows west into and along the north side of the fill, to discharge eventually through a ditch that flows

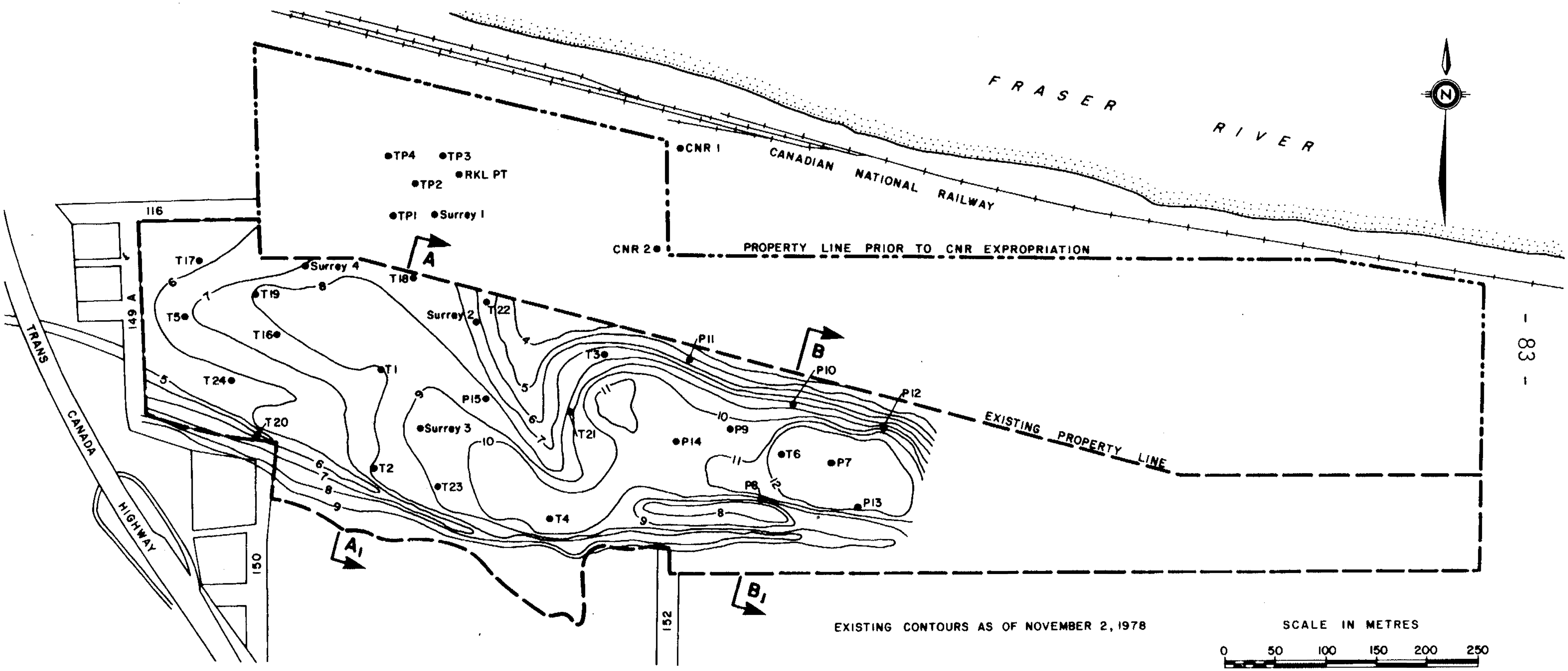


FIGURE 3.3.2 BORE HOLE LOCATIONS - PORT MANN LANDFILL - DISTRICT OF SURREY

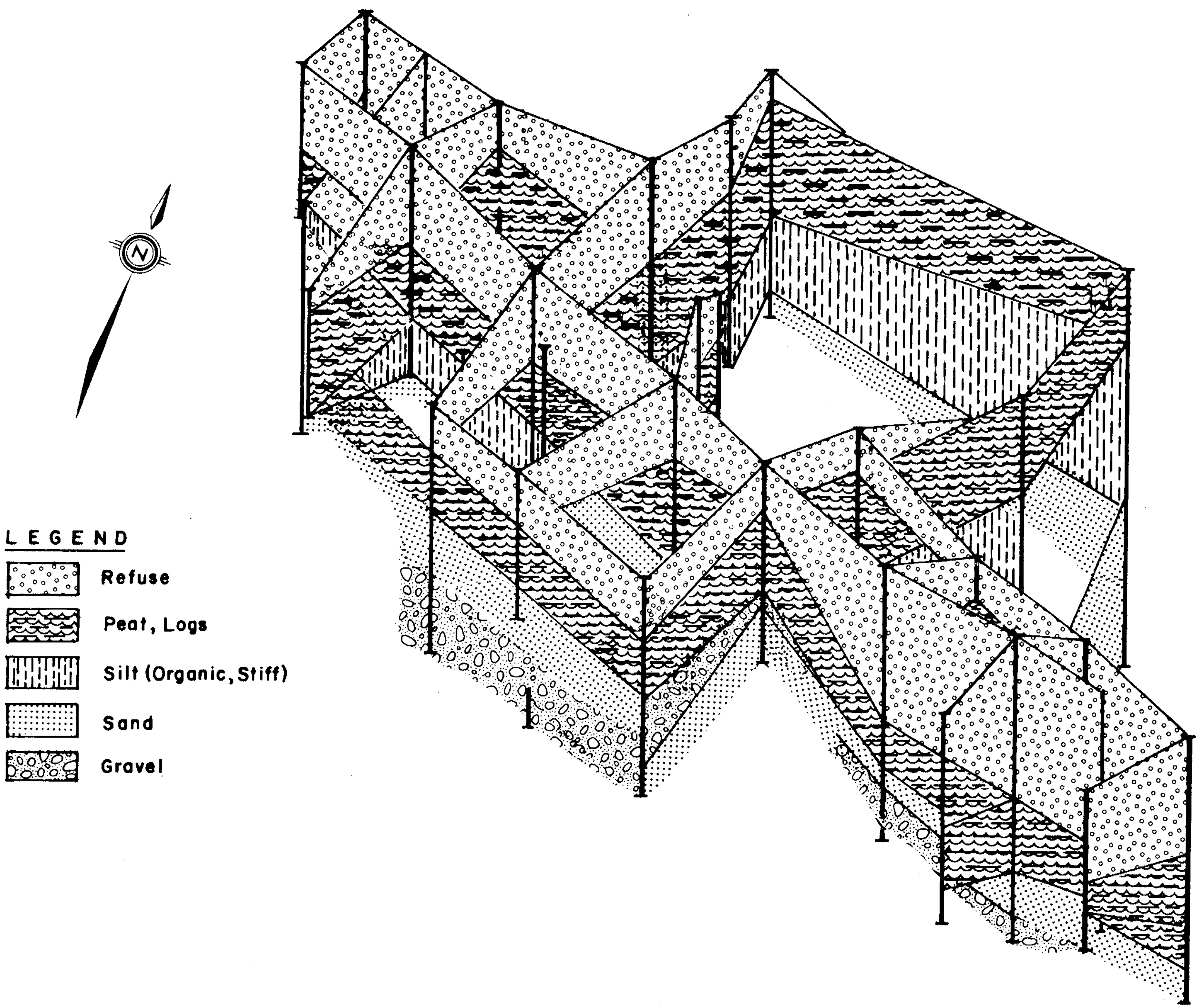


FIGURE 3·3·3 GEOLOGICAL FENCE - PORT MANN LANDFILL, DISTRICT OF SURREY

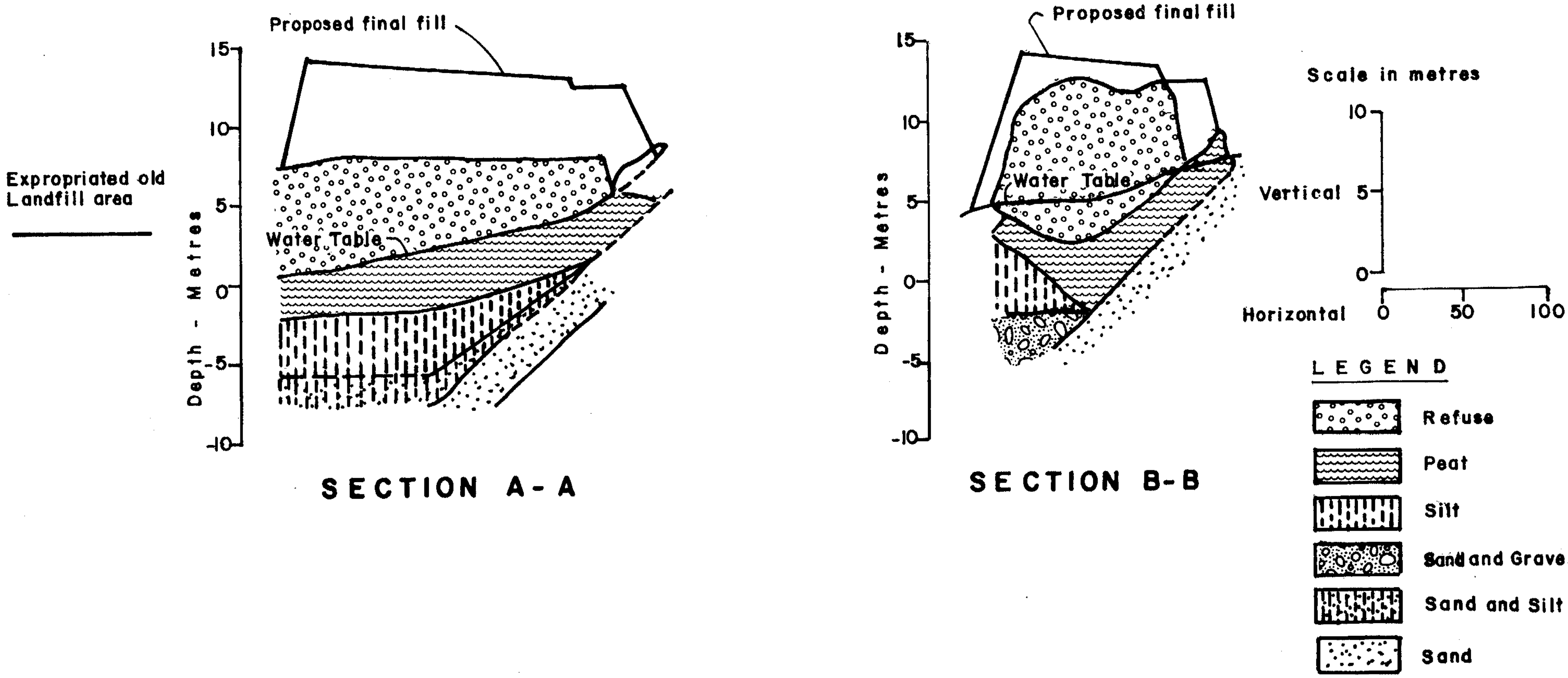


FIGURE 3.3.4 GEOLOGICAL CROSS-SECTIONS - PORT MANN LANDFILL - DISTRICT OF SURREY

north from the landfill site to the Fraser River. Precipitation averages 150 cm a year with 75% falling between October and April. A schematic of the surface drainage and monitoring locations is presented in Figure 3.3.5.

3.3.4 Operation. The filling operation had consisted of the placement of a shallow construction debris hog fuel mattress with three subsequent 1.8 metre lifts of refuse and a final clay-till capping of 0.45 to 0.6 metres. A 150 mm layer of sand was used to cover the working face at the end of each operational day. The proposed operation is not expected to change significantly, except that the height of the refuse will now be to about 12 metres placed in 1.8 metre lifts.

For the years 1969 and 1971, an estimated 40 950 tonnes and 40 500 tonnes, respectively, of refuse were placed; of this, 80% was estimated to be domestic, 15% demolition, and 5% industrial (AESL, 1974). Quantities placed during 1978 were: 45 360 tonnes of municipal refuse, 15% of which was industrial; an estimated 19 100 m^3 of hog fuel and demolition debris; and 19 100 m^3 of sand (used for daily cover) (Surrey, 1978). Assuming a density of hog fuel of 0.15 tonnes/m^3 and construction debris of 0.88 tonnes/m^3, the annual weight of material placed at the Port Mann Landfill, excluding cover, is 65 000 tonnes.

3.3.5 Leachate. Leachate discharges from the Port Mann Landfill through two routes. The principal discharge is through the north flowing ditch, from the leachate pond adjacent to the active site (Figure 3.3.5). This is culverted under the CNR trackage onto the Fraser River foreshore. Almost all the leachate from the active site is believed to exit via this ditch. The second route would be through near-surface groundwater flow into the unfilled peat area and sand fill on the CNR property. From there it moves, eventually, into either the main leachate ditch near the tracks or into the drainage ditch along the west perimeter. It is expected that any leachate entering the west perimeter ditch would have undergone some renovation through the attenuation afforded by 60 to 240 metres of lateral groundwater movement. The leachate from the completed portion of the site is believed to discharge by this means.

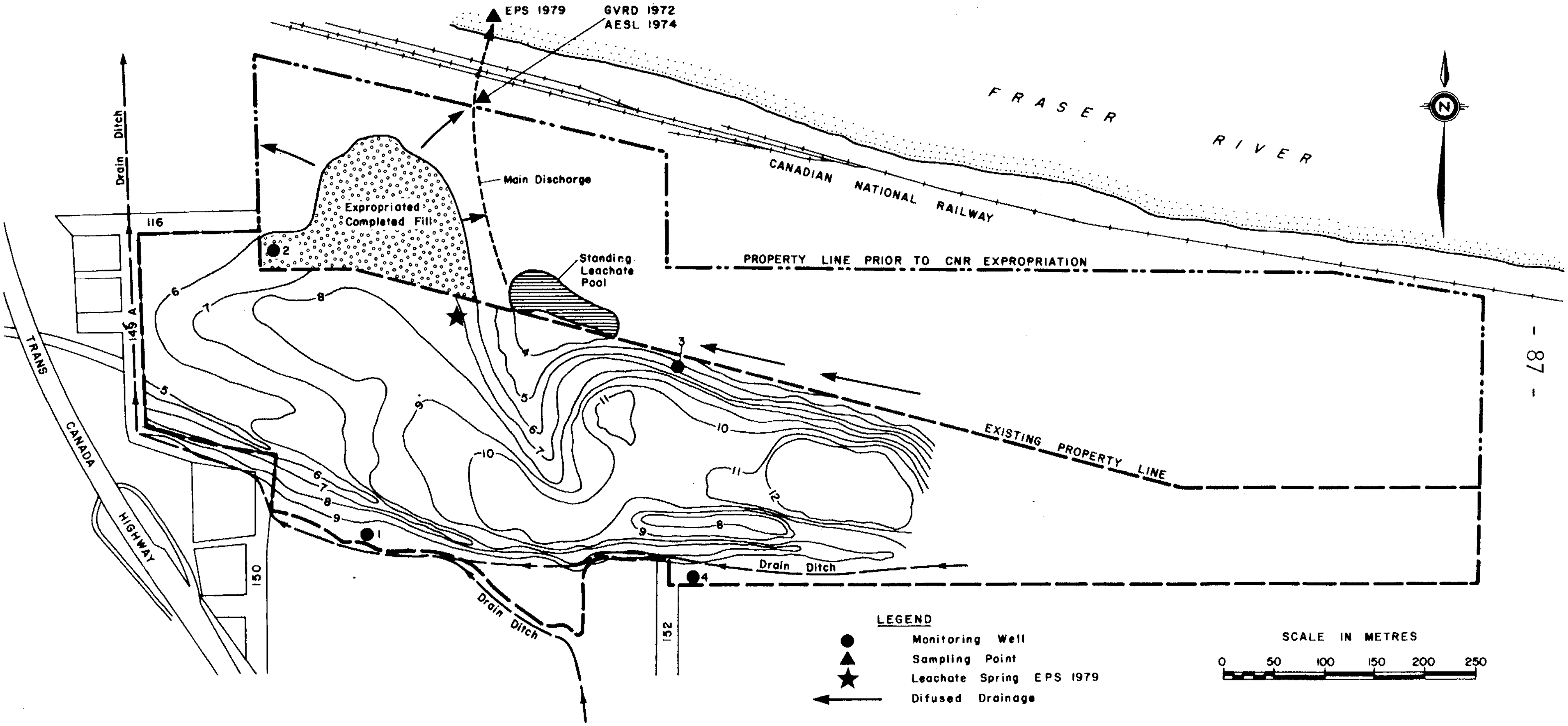


FIGURE 3.3.5 DRAINAGE LAYOUT AND MONITORING LOCATIONS - PORT MANN LANDFILL - DISTRICT OF SURREY

The main leachate discharge from the Port Mann Landfill has been monitored sporadically since 1972 by a number of groups (GVRD, 1972, 1973), (AESL, 1974), (PCB, 1974, 1975), (CNR, 1976), (EPS, 1976, 1979) and (FMS, 1976). Leachate data are also available for four monitoring wells, the locations of which are shown in Figure 3.3.5. These wells have been monitored on a continuing basis by the Waste Management Branch (EQUIS, 1978).

A summary of the well data is presented in Table 3.3.1. Wells 1 and 4 are considered to be control wells, however, the nitrogen, COD, phosphate, copper and zinc levels are somewhat elevated. It may be possible that they are being influenced by the landfill or by the upstream drainage which has some contamination. In the GVRD 1972/73 study, elevated levels of constituents were noted on occasion in the stream drainage. This may be due to the presence of the Johnston Road Landfill discussed in Section 6.2.3. The Well 2 results are representative of the leachate from the old site. For the most part the results have not changed significantly since 1975, however, the chloride and COD levels have tended to decrease while the calcium, ammonia and hardness values have increased. The results from Well 3, which is adjacent to surface leachate discharges, is likely typical of the leachate in the ditch before the effects of dilution.

A summary of the analytical flow data for the main surface leachate discharge is presented in Table 3.3.2. The GVRD measurements were taken prior to the diversion of the upslope surface waters and, while the concentration could be representative, the flows are not. The AESL data taken for the period April to July 1974 represent the drainage from about 6 ha plus some surface water from the east side, and was thought to be representative at that time so as to allow for extrapolation to the then filled 8 ha (AESL, 1974). No flow information is available for use with the 1974/1975 PCB or 1976 EPS, CNR or FMS data, although the data are useful for comparison of concentration.

The 1979 EPS data are the only recent data. These data are representative of the leachate flow for days in the month of March from some 17.8 ha of fill plus surface drainage entering the site from the

TABLE 3.3.1 MONITORING WELL ANALYSES - PORT MANN LANDFILL - DISTRICT OF SURREY

Analysis	Well #1				Well #2				Well #3				Well #4			
	No. of Samples	Max.	Min.	Avg.	No. of Samples	Max.	Min.	Avg.	No. of Samples	Max.	Min.	Avg.	No. of Samples	Max.	Min.	Avg.
Colour (relative units)	3	80	30	57	2	60	40	50	2	40	10	25	3	10	5	6.7
pH (relative units)	14	7	5.7	6.4	14	7.5	6.2	6.8	14	6.6	6.2	6.4	14	6.9	6.2	6.6
NFR	1	739	739	739	1	263	263	263	1	836	836	336	1	3567	5567	3567
Spec. Cond. (umhos/cm)	15	600	207	267	15	>4000	700	1885	14	1160	296	495	15	600	184	250
Temperature (°C)	1	8	8	8	1	11	11	11	1	11.5	11.5	11.5	1	11	11	11
DO	1	2.2	2.2	2.2	1	2	2	2	1	3.2	3.2	3.2	1	3.2	3.2	3.2
Chloride	11	15.8	3.2	7.7	11	157	44.4	85.3	10	57.8	8.5	19.5	10	15.5	4.7	7.7
Fluoride	11	0.12	<1.1	0.1	11	.13	<0.1	<0.1	11	<0.1	<0.1	<0.1	11	<0.1	<0.1	<0.1
Hardness	11	139	61.8	84.1	10	883	347	650	11	281	83.5	128.8	11	147	59	74.9
Ammonia	11	3.52	0.57	2.5	11	57.5	22.8	46.9	11	16.9	3.92	8.02	11	0.034	0.007	.021
NO /NO	6	0.04	0.02	0.02	6	0.03	<0.02	0.021	2	0.04	<0.02	.023	6	1.64	.81	1.32
Nitrate	11	0.1	<0.02	0.029	10	0.07	<0.02	0.025	11	0.04	<0.02	.022	11	2.1	0.03	1.32
Nitrite	6	<0.005	<0.005	<0.005	6	0.006	<0.005	0.005	7	<0.005	<0.005	<0.005	6	0.005	<0.005	<0.005
Nitrogen Org.	11	4.88	0.48	2.02	11	7.6	0.5	3.5	11	2.2	<0.01	0.98	11	4.98	.82	2.32
Nitrogen Kjeldahl	11	7	2	4.5	11	63	24	50	11	17	6	9	11	5	.85	2.34
COD	11	409	23	133	11	538	77.7	194.7	11	351	16.1	99	11	252	<10	114
Phosphate (T)	8	3.27	0.46	1.51	8	.859	.344	.494	8	1.34	0.46	0.83	8	2.89	1.27	2.46
Sulphate	3	<5.0	5.0	<5.0	3	58.7	7.6	25.6	3	<5.0	<5.0	<5.0	3	23.3	20	21.2
Tannin & Lignin	10	17.5	2.5	5.6	10	140	0.9	21.7	10	8	1.6	3.64	10	9	1.2	5.3
Sulphide	2	<0.5	<0.5	<0.1	2	<0.5	<0.5	<0.5	2	<0.5	<0.5	<0.5	2	<0.5	<0.5	<0.5
Arsenic (T)	11	0.01	<0.005	.006	11	0.019	<0.005	.008	11	0.02	<0.005	.006	11	0.017	<0.005	.007
Calcium (D)	11	49.7	20.3	27.9	10	303	113	214.6	11	86	24.2	38.5	11	44.5	14.9	19.8
Chromium (T)	11	0.074	<0.005	0.036	11	0.044	<0.005	.011	11	0.11	<0.005	.019	11	0.14	0.031	0.066
Copper (T)	11	0.58	0.01	0.20	11	0.09	0.01	0.031	11	0.49	0.005	.018	11	0.58	0.14	0.29
Iron (D)	9	11.0	1.6	3.4	10	5.5	0.1	13.13	9	45	0.1	16.6	9	0.1	<0.1	<0.1
Iron (T)	1	18.3	18.3	18.3	1	115	115	115	2	62.2	39.0	50.6	2	126	38.3	82.2
Lead (T)	11	0.1	<0.001	0.023	9	0.1	<0.001	.026	10	0.014	<0.001	0.005	11	0.2	<0.001	0.029
Magnesium (D)	11	4.8	2.7	3.5	11	35	15.7	28.1	11	16	5.6	7.9	11	8.6	5.3	6.1
Manganese	11	3.79	.26	1.57	11	6.11	1.34	2.59	11	4.78	0.46	1.16	11	14.2	.96	3.8
Zinc	11	0.43	0.02	0.18	11	0.13	0.005	.053	10	0.44	<0.005	0.087	11	.76	.11	0.31

Units in mg/l except where indicated.

TABLE 3.3.2 ANALYTICAL DATA LEACHATE DITCH - PORT MANN LANDFILL - DISTRICT OF SURREY

Analysis	GVRD 1972 (May - Dec.)				AESL 1974 (Apr. - July)				PCB 1974/1975 (Nov./74 - Feb./75)				EPS 1976, FMS 1976 (Nov./76 - May/76)				EPS 1979			
	No. of Samples	Max.	Min.	Avg.	No. of Samples	Max.	Min.	Avg.	No. of Samples	Max.	Min.	Avg.	No. of Samples	Max.	Min.	Avg.	No. of Samples	Max.	Min.	Avg.
Colour (relative units)	-	-	-	-	6	----------Block---------			-	-	-	-	-	-	-	-	-	-	-	-
pH (relative units)	17	7.0	6.5	6.7	6	7.6	7.1	7.4	4	7.5	6.85	7.2	1	6.7	6.7	6.7	5	7.6	7.3	7.4
NFR	17	973	4	133	6	289	49	153	-	-	-	-	-	-	-	-	5	120	66	92
Spec. Cond. (umhos/cm)	18	2441	550	1352	-	-	-	-	2	3000	2360	2680	-	-	-	-	5	1750	1365	1580
Temperature (°C)	17	24	5	11.5	-	-	-	-	2	14	9	11.5	-	-	-	-	-	-	-	-
DO	17	8.4	0	1.9	-	-	-	-	2	3.2	0.4	1.8	-	-	-	-	-	-	-	-
Chloride	17	139	13	61	6	230	112.5	170	2	142.5	86.6	114.5	-	-	-	-	5	106	70.5	86
Fluoride	-	-	-	-	-	-	-	-	2	0.11	< 0.1	< 0.1	-	-	-	-	-	-	-	-
Hardness	-	-	-	-	6	336.8	1257.3	702	1	1415	1415	1415	-	-	-	-	-	-	-	-
Ammonia	12	42	6.0	21.2	-	-	-	-	2	67.5	46.6	57.0	-	-	-	-	5	45.0	25.9	35
NO /NO	-	-	-	-	-	-	-	-	2	0.26	0.03	0.14	-	-	-	-	-	-	-	-
Nitrate	-	-	-	-	-	-	-	-	2	0.26	0.03	0.14	-	-	-	-	5	.309	.010	0.16
Nitrite	-	-	-	-	-	-	-	-	2	78	51	64.5	-	-	-	-	-	-	-	-
Nitrogen Org.	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Nitrogen Kjeldahl	-	-	-	-	6	132	60	103	2	78	51	64.5	-	-	-	-	-	-	-	-
COD	14	1939	88	732	6	679	277	460	2	509	400	454.5	1	430	430	430	5	500	95	220
Phosphate (T)	10	621	< .01	0.03	6	< .1	< .1	< .1	2	1.5	0.57	1.0	-	-	-	-	5	.455	.140	.28
Sulphate	8	290	12	162	6	63	24	41	-	-	-	-	-	-	-	-	5	12.4	4.26	9.36
Tannin & Lignin	-	-	-	-	-	-	-	-	2	96.0	17	56.5	-	-	-	-	-	-	-	-
Sulphide	-	-	-	-	6	2.9	7.9	4.9	-	-	-	-	1	.28	.28	.28	-	-	-	-
Arsenic (T)	1	< 0.1	< 0.1	< 0.1	-	-	-	-	1	0.012	0.012	0.012	-	-	-	-	5	< .15	< .15	< .15
Calcium (D)	17	380	30	179	6	418	32.4	225	1	493	493	493	-	-	-	-	5	197	158	174
Chromium (T)	17	--15 of 17 below detection--			3	0.042	0.030	0.037	1	0.019	0.019	0.019	2	.020	< .02	< .02	5	< .015	< .015	< .015
Copper (T)	17	0.30	0.01	0.07	3	0.048	0.042	0.041	1	0.050	0.050	0.050	2	.020	.010	.015	5	< .01	< .01	< .01
Iron (D)	-	-	-	-	6	7.95	0.05	2.0	2	2.6	1.0	1.8	-	-	-	-	-	-	-	-
Iron (T)	17	206	5.9	42.0	6	38	7.1	19	1	37.1	37.1	37.1	2	31.0	25.0	28.0	5	39.5	23.7	30.2
Lead (T)	17	--11 of 17 below detection--			3	< 0.005	< 0.005	< 0.005	1	0.01	0.01	0.01	2	.06	.02	0.04	5	< .08	< .08	< .08
Magnesium (D)	17	30.0	3.3	15.4	6	62.4	1.6	34.0	1	44.7	44.7	44.7	2	14	12	13	5	27.2	20.2	23
Manganese	15	3.58	0.26	1.86	-	-	-	-	1	3738	3.38	3.38	-	-	-	-	5	6.01	1.62	4.8
Zinc	17	1.62	0.07	0.33	3	0.026	0.012	.020	1	.18	.18	.18	-	-	-	-	5	.32	< .02	0.13
T.Alk.	17	1060	125	442	6	-	1470	2032	-	-	-	-	-	-	-	-	5	790	620	680
Total Residue	17	3645	447	1437	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
Phenols	3	< 1.0	< .01	0.62	-	-	-	-	-	-	-	-	-	-	-	-	2	.046	.039	.043
Sodium (D)	17	365	15	64	-	-	-	-	-	-	-	-	-	-	-	-	2	82.1	58.7	72.5
Potassium (D)	17	204	6.8	40.5	-	-	-	-	-	-	-	-	-	-	-	-	5	-	-	-
Cadmium (T)	17	--11 of 17 below detection--			3	< 0.005	< 0.005	< 0.005	-	-	-	-	2	0.05	< 0.01	< .030	5	< .01	< .01	< .01
Nickel (T)	17	--17 of 17 below detection--			3	0.032	< 0.005	0.22	-	-	-	-	2	< .05	< .05	< .05	5	< .08	< .08	< .08
Aluminum (T)	1	< 1	< 1	< 1	-	-	-	-	-	-	-	-	-	-	-	-	5	.71	< .09	.25
Coli. (T) MPN/100 ml	17	43000	230	7799	-	-	-	-	2	2200	< 2000	42100	-	-	-	-	-	-	-	-
Fecal MPN/100 ml	17	9300	< 30	867	-	-	-	-	2	2000	500	1250	-	-	-	-	-	-	-	-
Diss. Solids (T)	17	3589	413	1305	6	2024	1470	1634	-	-	-	-	-	-	-	-	-	-	-	-
BOD	17	973	4	133	6	475	200	310	-	-	-	-	1	160	160	160	-	-	-	-
Flow	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	5	.8 cfs.	.13 cfs.	.32

Units in mg/l except where indicated.

east. With the construction of the CNR yard in 1977 and 1978, it is felt that the drainage entering from the east is likely more significant than when AESL carried out their study. However, no estimate is available as to the amount.

3.3.6 Loading. There is a paucity of information upon which one could base the calculations of loadings to the Fraser River. The calculated loadings presented in Table 3.3.3 were based on the following assumptions: one, the March 9, 1979 EPS values represent the maximum daily loadings; and two, a ratio of maximum daily loading to average daily loading of 2.25:1, calculated using the 1972 GVRD data and the 1974 AESL data can be applied to the 1979 data. The flow on March 9, 1979 was 0.022 m^3/sec. The ratio calculated from the GVRD and AESL data is averaged from seven constituent ratios: COD, ammonia or total Kjeldahl nitrogen, calcium, total iron, magnesium, chloride and sulphate.

A summary of the GVRD and AESL loading data and calculated ratios is presented in Table 3.3.4. The calculations using the GVRD data are based on weir flow times concentration measured at the weir.

3.3.7 Impact on Water Quality

3.3.7.1 Point of egress. In this discussion of the impact on water quality only the main leachate discharge is considered. Most of the leachate is believed to discharge through the one culvert onto the Fraser foreshore, with the remainder possibly diffusing through the ground into the west diversion ditch. No monitoring data, chemical or biological, are available to suggest what the impact of the leachate is on the Fraser River at that point. It is very unlikely that there is any measurable change in the water chemistry beyond the immediate mixing zone. The discharge is obvious by the dark plume and foam that hugs the shore for the first 30 or so metres.

3.3.7.2 Impact on biota at point of egress. No biological monitoring of this disharge has been done. The substrate along that portion of the Fraser River is a medium clean sand. There were no obvious deposits on

TABLE 3.3.3 LEACHATE CONSTITUENT LOADINGS - PORT MANN LANDFILL DISTRICT OF SURREY

Parameter	Concentration (mg/l)	Maximum Loading (kg/day)	Average* Loading (kg/day)
COD	500	980	435
NFR	93	180	80
T.Alk. as $CaCO_3$	620	1210	540
SO_4	12.4	24	11
Chloride	92.7	180	80
Total Phosphate	0.455	.90	.40 ?
Ammonia	45	88	40
Aluminum	.11	.22	.10
Arsenic	<.15	NC	NC
Barium	.136	.27	.12
Calcium	158	310	138
Cadmium	<.01	NC	NC
Cobalt	<.015	NC	NC
Chromium	<.015	NC	NC
Copper	<.01	NC	NC
Iron	30.9	60	27
Mercury	<.1	NC	NC
Magnesium	21.0	41.0	18
Manganese	1.62	3	1.4
Molybdenum	.15	NC	NC
Sodium	73.4	145	63
Nickel	<.08	NC	NC
Lead	<.08	NC	NC
Antimony	<.08	NC	NC
Selenium	<.15	NC	NC
Tin	<.2	NC	NC
Strontium	1.04	203	90
Titanium	<.009	NC	NC
Vanadium	<.05	NC	NC
Zinc	.32	.63	.30
Silicon	6.48	13	5.60

* Average loadings calculated from maximum loadings, using a ratio of max:min = 2.25:1

NC = Not Calculated

TABLE 3.3.4 SUMMARY OF LOADING DATA - PORT MANN LANDFILL - DISTRICT OF SURREY

Analysis	GVRD 1972 (May - Dec.)					AESL 1974 (Apr. - July)					Avg.
	No. of Samples	Load			Ratio	No. of Samples	Load			Ratio	
		Min.	Max.	Avg.	Max./Avg.		Min.	Max.	Avg.	Max./Avg.	
COD (lbs/day)	10	317	3967	1341	2.96	6	58.7	537.1	276.1	1.95	2.45
Ammonia or Kjeldahl N (lbs/day)	9	13.7	65.1	30.8	2.11	6	17.65	98.1	58.4	1.68	1.90
Calcium (lbs/day)	10	25.0	740.0	327.5	2.26	6	14.0	260.9	146.8	1.78	2.02
Magnesium (lbs/day)	10	7.6	70.9	28.3	2.50	6	1.6	27.0	14.2	1.90	2.2
Chloride (lbs/day)	11	40.5	221.5	104.3	2.12	6	29.0	150.3	96.6	1.56	1.84
Iron (T) (lbs/day)	11	4.8	272	80.0	3.37	6	5.40	23.0	9.9	2.32	2.84
Sulphate (lbs/day)	5	81	726.5	267.3	2.72	6	7.2	64.1	27.2	2.35	2.54
Average					2.58					1.93	2.25

Source: GVRD, 1972 and AESL, 1974.

the substrate at the point of discharge. The toxicity (LC_{50}) to rainbow trout of the leachate as measured in May, 1976 (FMS, 1976), March 9, 1979 and March 23, 1979 (EPS, 1979) was 42%, 50% and 38%, respectively.

3.3.8 Sewer Connection. The Corporation of the District of Surrey anticipates the installation of leachate control works before the end of 1979, with discharge through a GVRD trunk sewer to the Annacis Island STP. It would be anticipated that the quality of the leachate from the Port Mann Landfill will change with a shift towards lower pH and higher metal levels. It is felt that this change will come about as a result of the greater depth of refuse and restricted infiltration of water envisioned for the future Port Mann Landfill operation.

3.3.9 Intended Use. While the future of the surveyed portion of the Port Mann Landfill has not been finalized it is anticipated that it will likely be recreational. The settlement and gas related problems due to the deep fill are not felt to be conducive to industrial development (Surrey, 1978).

The Canadian National Railway intends to place track over that portion of the old landfill that was expropriated. While the placement of trackage is not anticipated for a number of years, a 2 metres preload has been placed over the old fill (Jansen, 1978, personal comm.).

3.4 GVRD - Coquitlam Landfill (Braid Street)

3.4.1 General. The Coquitlam Landfill, owned by GVRD, is a 31.1 ha site, located in the Fraser Mills area of Coquitlam Municipality. The landfill operation is carried out by Ambassador Industries Limited, a private contractor, and was initiated in 1975 under a temporary Pollution Control Branch approval. The site is essentially a continuance of the Intertidal Industries (Terra Nova) landfill which ceased operation upon initiation of this site. A Provincial Pollution Control Permit PR-4385 was issued in November 1975, superseding the Pollution Control Branch Lower Mainland Region temporary approval.

3.4.2 Physical Description. The Coquitlam Landfill is bordered on the north by Highway No. 1 and the Canadian Pacific Railway line, on the east by Crown Zellerbach, on the west by the Brunette River and New Westminster, and on the south by Domtar Industries. The Domtar property is located between the site and the Fraser River. The site is generally flat with elevation variations (geodetic) between 30.5 metres and 32 metres. The Fraser River dyke elevation is 31.7 metres, and the 1948 recorded high water level at the foot of Braid Street was 31.25 metres (102.54 ft geodetic). Plate 3.4.1 is an aerial photograph of the site, and Figure 3.4.1 is a site plan.

The surficial geology is silty clay overlain by a swampy layer of humus. The unfilled portion of the site is covered by second-growth deciduous trees. Although a tree buffer has been left as a visual screen along the western portion of the site, the operation is clearly visible from Highway No. 1. Rainfall is approximately 150 cm/yr.

As with many of the landfill sites located in the Fraser River Estuary, the surficial deposits tend to preclude vertical migration of leachate. In this case the silty clay layer which is believed to be continuous throughout the site is, in turn, underlain by blue clay at a depth of 1.8 to 2.4 metres. Accordingly, little or no vertical leachate migration is anticipated.

The leachate interception system involving perimeter ditches and central site drains would be expected to both control the site watertable level and intercept the majority of the site water input. Section 3.4.4 details the leachate control works in place at this site. There are no water courses on the site. Two GVSS and DD sewer interceptors fall within the property, generally along the perimeter of the site (Figure 3.4.2).

3.4.3 Operation. Refuse discharged to the Coquitlam Landfill is from New Westminster, Coquitlam, Port Coquitlam, and Burnaby residential collection forces and commercial and industrial refuse haulers on a six-day per week basis. The Pollution Control Permit is for an average

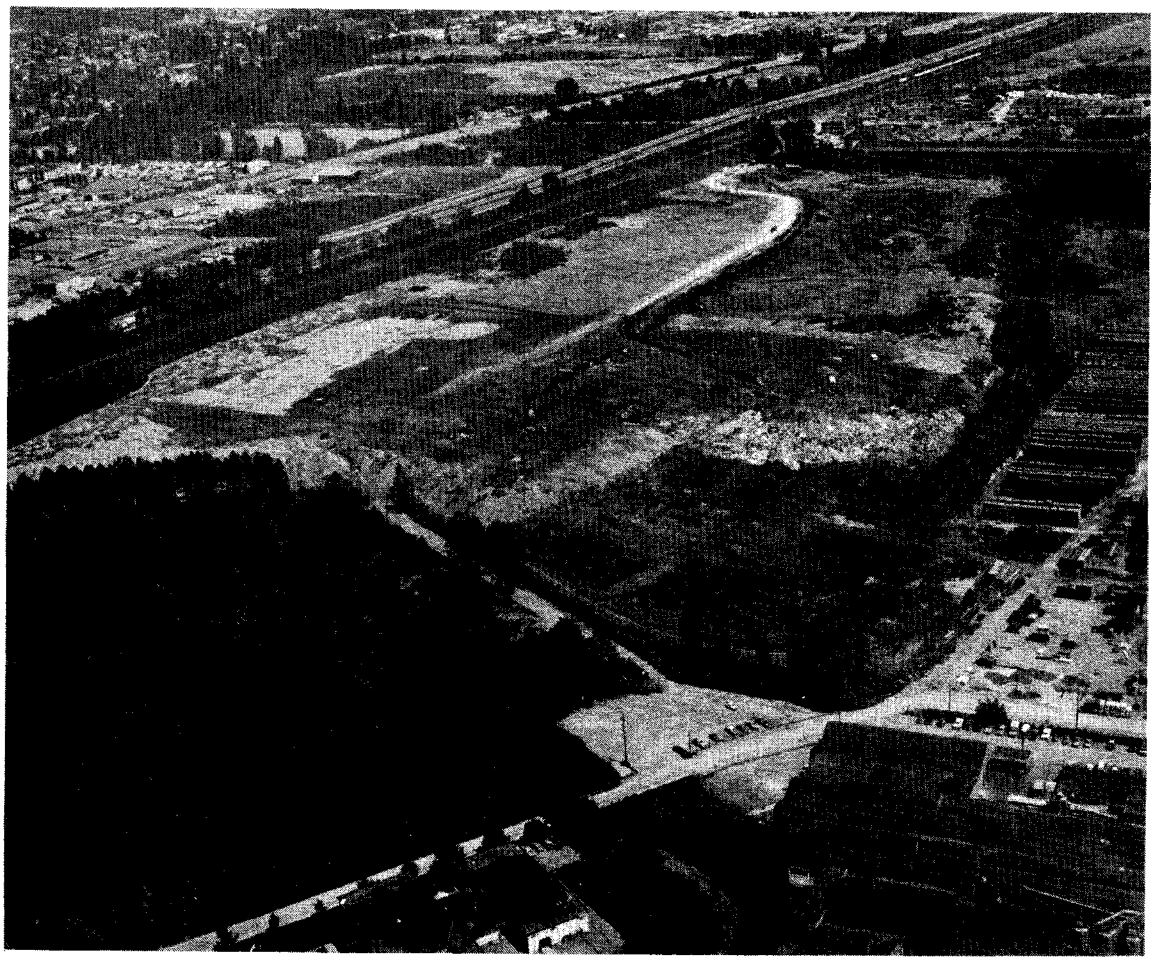


PLATE 3.4.1 GVRD - COQUITLAM LANDFILL SITE

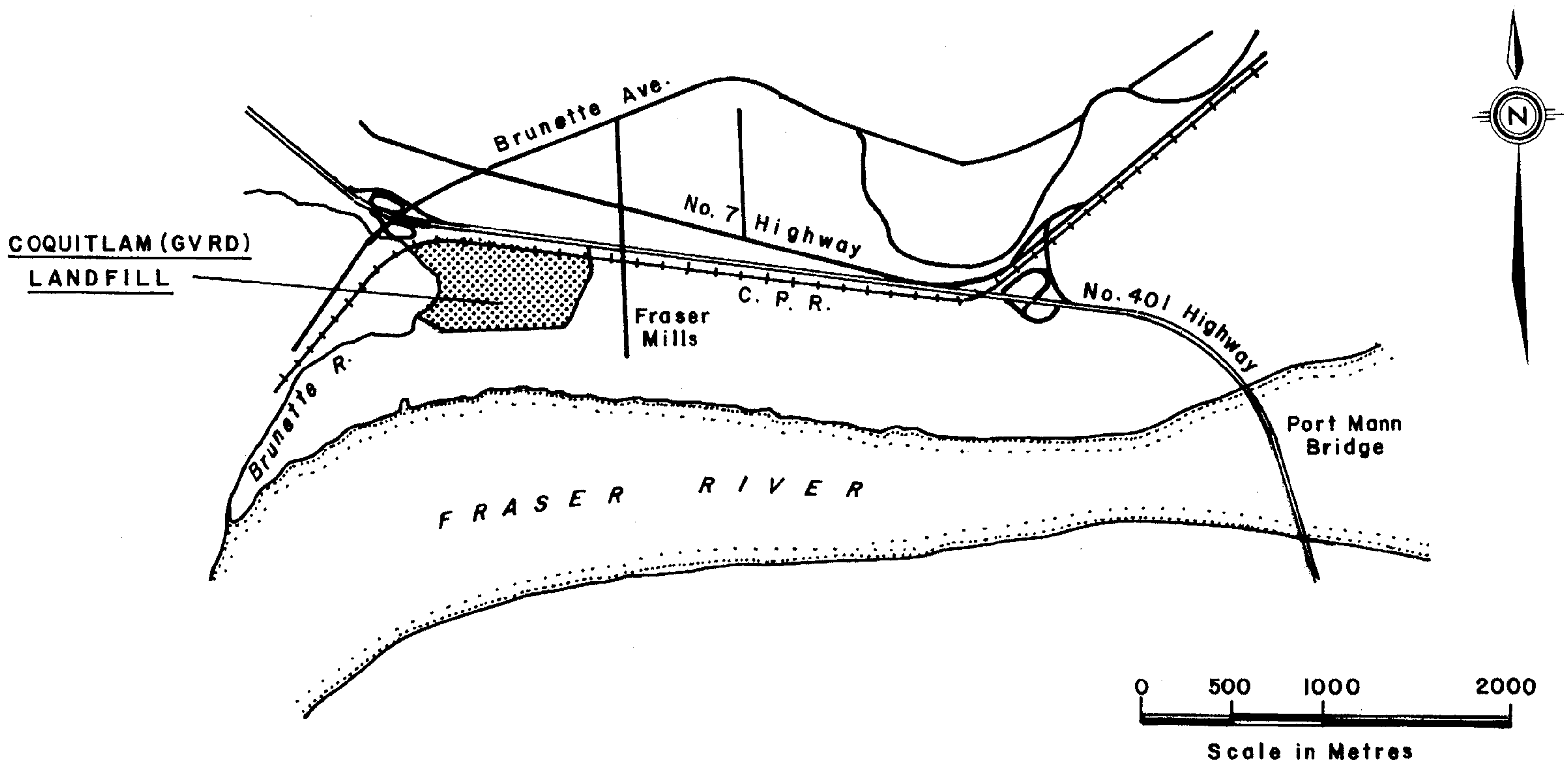


FIGURE 3.4.1 GVRD - COQUITLAM LANDFILL - LOCATION MAP

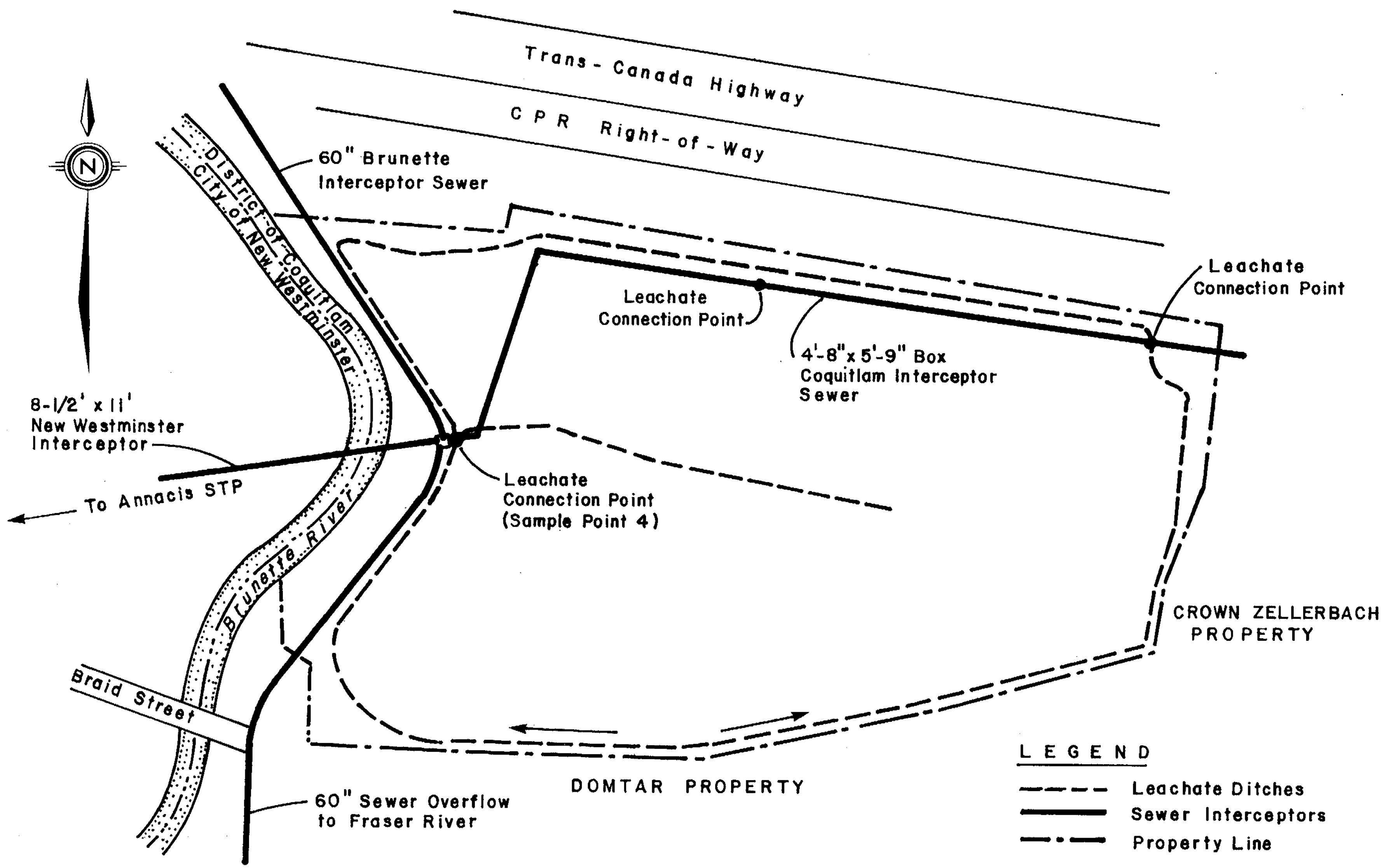


FIGURE 3.4:2 GVRD - COQUITLAM LANDFILL LEACHATE CONTROL WORKS

discharge of 408 tonnes/day, approximately 65% of which is from residential collection forces; the balance is commercial and industrial. Hazardous and toxic wastes are not accepted.

The filling operation is by means of cells in which the native soils are removed to form a perimeter dyke (22.8 to 24.4 metre base) around the individual cells. Figure 3.4.3 shows the layout of the total 10 cells planned. The total refuse depth is approximately 6.7 metres, consisting of two 3.35 metre lifts. The second lift is placed 18 months after the first. Intermediate daily cover is hog fuel, and the final cover is proposed as 0.6 metre of granular materials.

GVRD records for the cumulative fill volume are shown in Table 3.4.1. Based on these figures, the average in-place rate of refuse filling is in the order of 22 900 m^3/month or 267 600 to 305 800 m^3 per annum. The average in-place density is 565 kg/m^3. With the proposed 6.7 metre total lift height and the available acreage of approximately 28 ha, the remaining life of this site would be expected to be six to eight years (2.8 ha per year fill rate).

3.4.4 Leachate. Although initially there were leachate discharges to the Brunette River resulting from this landfill operation, a leachate containment and collection system has seemingly alleviated this problem. Following installation of the leachate control works there was a flapgate discharge to the Brunette River in the north west corner of the property. This discharge point was cut off in early 1978. The leachate control works involve a perimeter and central drainage ditch network. The central drainage ditch is connected to each cell with drainage pipe through the interior cell barrier. As the fill progresses, the central drainage ditch is filled with drainage rock and is covered. Leachate generated is therefore contained vertically by the natural surficial geology noted earlier and contained laterally by the drainage and interception works. The perimeter control ditch which receives the collected leachate discharges to the perimeter GVSS and DD sewer interceptors at three connection points as shown in Figure 3.4.2; that is, to the Coquitlam interceptor at the northeast corner of site, to the Coquitlam interceptor on the north boundary positioned approximately

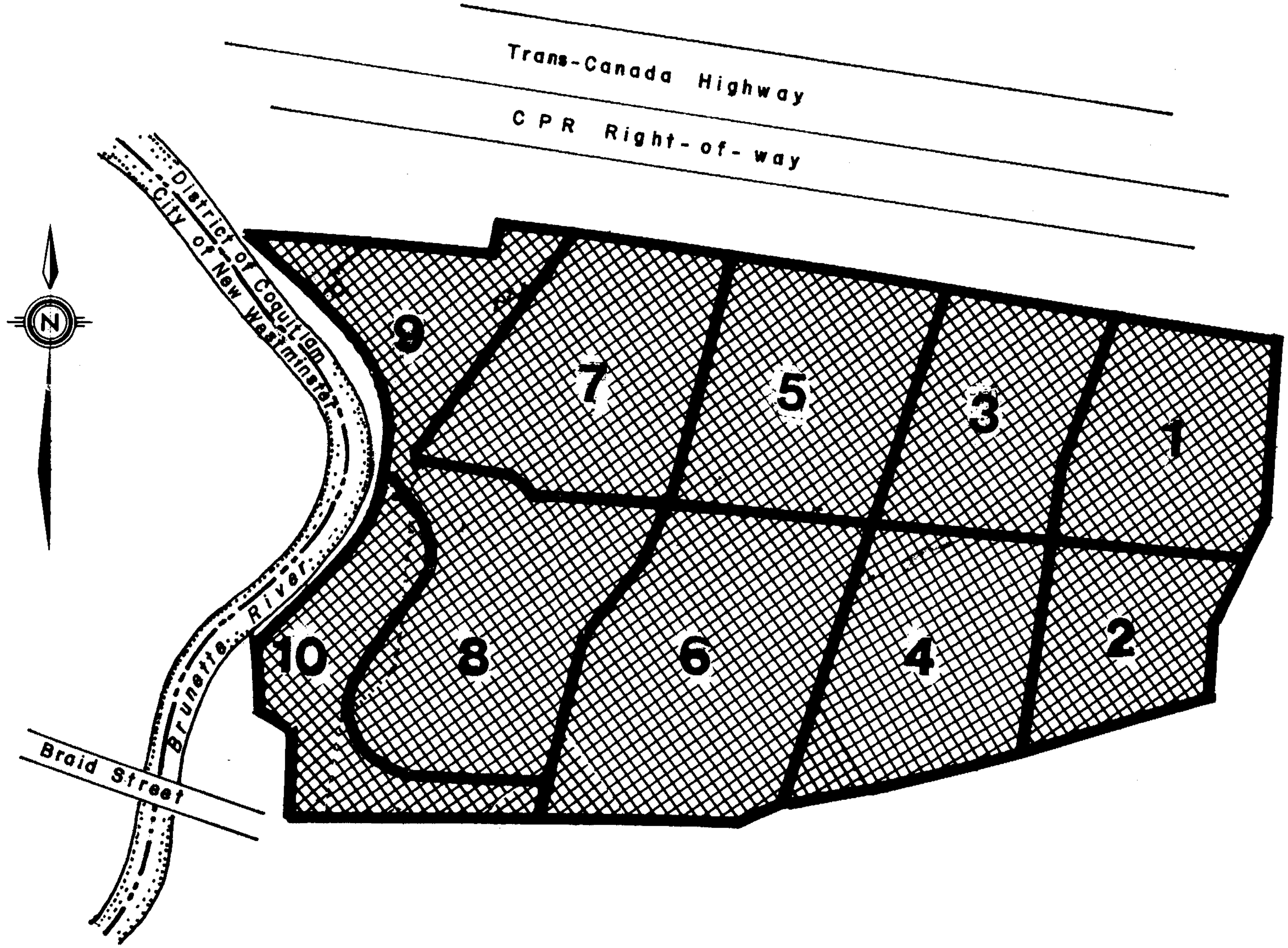


FIGURE 3.4.3 APPROXIMATE CELL LAYOUT-GVRD-COQUITLAM LANDFILL

TABLE 3.4.1 CUMULATIVE FILL VOLUMES - GVRD - COQUITLAM LANDFILL

Survey Date	Cumulative Volume of Fill (cu. yds)
January 2, 1976	75 000
February 2, 1976	114 000
March 2, 1976	150 000
March 29, 1976	166 000
June 2, 1976	262 200
August 9, 1976	321 200
September 7, 1976	345 200
October 1, 1976	365 200
November 4, 1976	391 500
December 8, 1976	418 500
March 18, 1977	500 170
August 18, 1977	675 000
April 4, 1978	890 900

Source: GVRD Survey Records.

365 metres west of the northeast corner, and to the New Westminster interceptor on the west side of the property. The New Westminster interceptor connection point is the common connecting point of the Coquitlam and Brunette interceptors. From this connection point, the New Westminster interceptor conveys the combined flows to the Annacis Island STP. The southern portion of the Brunette interceptor from this connection point serves as an emergency overflow to the Fraser River. Accordingly, during high flow periods, leachate together with other flows, could be bypassed to the Fraser River.

As the ditch levels are reported to be 1.5 to 1.8 metres below the surrounding lands, the resultant hydraulic gradients would be expected to provide lateral hydraulic isolation. The elevation difference would also be expected to drain surrounding land, such that significant drainage water would be discharged to the sewer interceptor. More precise control of this would require a leachate pump discharge which would allow control of water elevations as a function of neighbouring land water levels.

As with all landfill sites, a determination of accurate water balance flow estimate is difficult. A rough estimate based on the 150 cm of annual precipitaion falling over the entire site coupled with 15% - evaportranspiration losses, water uptake in the fill, and negligible additional water inputs (neighbouring lands, soils compression, refuse water component, etc.), yields an annual flow of approximately 0.012 m^3/sec. Initially, plans called for the pumping of leachate to the interceptor, which would have allowed a flow measurement estimate in addition to water level elevation control. The present gravity discharge scheme however precludes flow measurement.

The leachate is difficult to characterize in any specific terms due to the many site variables and dilution and aging factors. The mixed runoff drainage and leachate sampled from the perimeter interception ditch yields a generalized picture of the mixed drainage/leachate character.

GVRD initiated a sampling and monitoring program of this landfill site at the onset of filling, which has continued to the present on a monthly to quarterly basis. There are some ten surface ditch sample sites and five monitoring wells involved. Recorded results of the sampling program (GVRD, 1975 and 1978)* indicate the expected surface contamination due to leachate interception. The monitoring results of sampling site No. 4 are shown in Table 3.4.2. Sampling occurred between May 1976 and March 1978. This site located at the point of discharge to the sewer interceptor (Figure 3.4.2), is believed to provide a representative indication of the composite leachate discharge. In line with the previous landfill sites, the Coquitlam Landfill shows similar leachate characteristics; that is, high ammonia, oxygen demand, and total solids levels. Again, with the exception of iron, manganese and zinc, the heavy metal concentrations are low. In this regard, it is interesting to note that alkalinity has increased steadily from very low concentrations in the order of 30 mg/l at the onset of sampling (and landfilling), to concentrations in excess of 1000 mg/l during the more recent sampling. This trend, shown in Figure 3.4.4, with its attendant buffering capacity and elevated pH, may provide some further insight into the apparent immobility of the metals within the fill as evidenced by their seemingly low concentrations in the leachate.

In an attempt to quantify the mass loading contribution of the landfill leachate, the average contaminant concentration at sampling site No. 4 over the two-year sampling period has been extended and annualized, using an estimated average annual leachate outflow of 0.012 m^3/sec. The daily contaminant loading estimates are shown in Table 3.4.3. Again, these figures must be viewed as approximate only due to the minimal data base, variation of results, and approximate nature of the water balance estimated outflow.

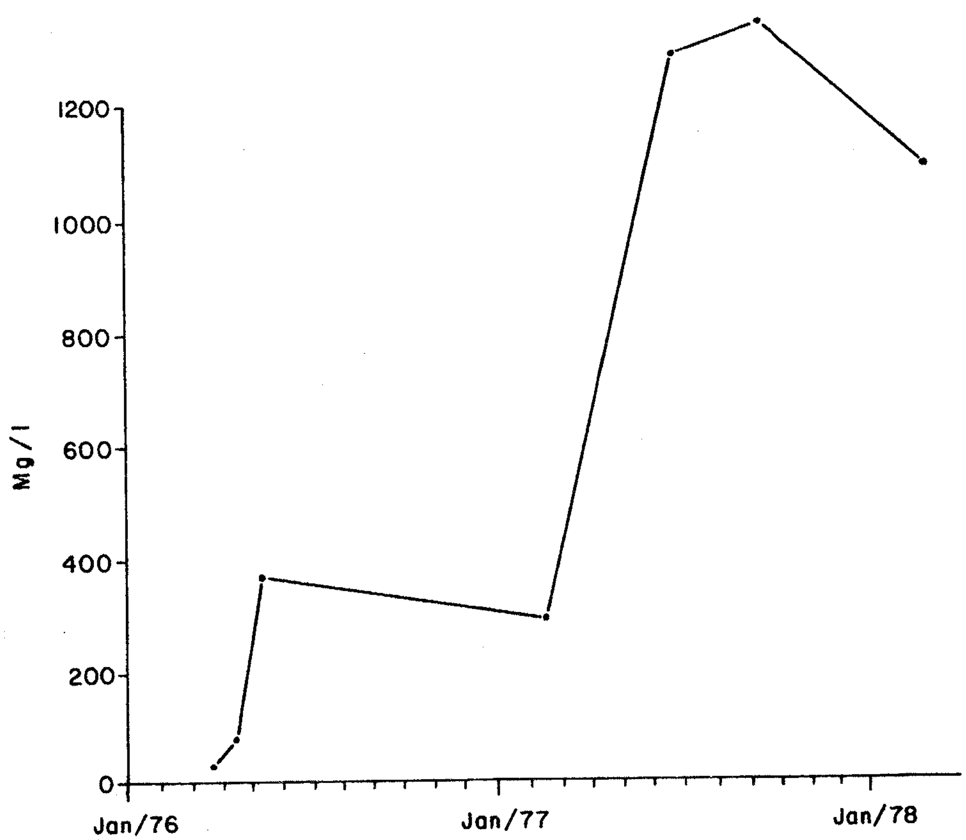
---

*Monitoring program was not directed by the Pollution Control Branch, as the ultimate discharge was to a sewer interceptor.

TABLE 3.4.2 MONITORING DATA FROM SAMPLING POINT 4 - COQUITLAM LANDFILL

Parameter (mg/l)*	Sample Point 4 No. of Samples	Range Min. - Max.	Avg.	Control Wells Well B 07/12/77	Well H 07/12/77	Well H 06/03/78	Well B 06/03/78
pH (units)	7	5.7 - 7.6	6.4	7.1	7.2	6.2	5.9
Ammonia (N)	7	.91 - 80	28.9	.14	.03	0.05	0.04
Spec. Cond. (μmho/cm)	7	410 - 3700	1915	223	107	100	155
Non-filterable Residue	7	101 - 450	211	205	104	280	272
Total Residue	7	487 - 3495	1660	397	203	481	673
COD	7	19.5 - 1572	842	74	254	56	72
T. Alk ($CaCO_3$)	7	26 - 1223	399	71.3	38.7	35	38
Chloride	7	30 - 370	173	13.3	5.3	6	8
Sulphate	7	2 - 56	33.3	39.6	6.5	.8	15
Sodium	6	24 - 230	108	7.2	2.2	2.0	5.6
Calcium	6	12 - 280	119	32.0	16.0	12	14
Magnesium	6	5.5 - 53	27.3	9.7	5.1	5.4	7.5
Copper	6	< .04 - 0.11	< 0.07	0.03	0.02	< 0.04	< 0.04
Lead	5	.009 - 0.2	0.052	0.02	< 0.02	< 0.005	< 0.014
Iron	7	11.9 - 102	42.7	30.0	39.0	11.6	15.6
Zinc	7	0.25 - 1.17	0.59	0.06	0.03	0.03	0.03
Cadmium	7	<.0005 - .02	< .0038	< 0.005	< 0.005	< 0.0005	< 0.0005
Chromium	7	.002 - .025	< .037	0.02	0.02	< 0.05	< 0.05
Nickel	7	.06 - .64	< .13	0.02	0.02	< 0.07	< 0.07
Arsenic	2	< .02 - .195	< .01	0.02	< 0.02	< 0.02	< 0.02
Manganese	7	.25 - 7.9	3.07	-	-	0.40	0.27

*Except as noted.



DATE	T. ALK. ($CaCO_3$) Mg/l
March 25/76	26
April 15/76	92
May 5/76	370
Feb. 14/77	243
June 17/77	1210
Sept. 7/77	1223
March 6/78	1055

FIGURE 3.4.4 ALKALINITY CONCENTRATION INCREASE WITH FILL AGE - COQUITLAM LANDFILL

TABLE 3.4.3 ESTIMATED DAILY CONTAMINANT MASS LOADINGS (Based on average outflow contaminant concentrations and estimated annual leachate outflow of 0.012 m^3/sec) - COQUITLAM LANDFILL

Parameter (mg/l)	Daily Loading (kg/day)
Ammonia (N)	29
Total Residue	1640
Non-filterable Residue	210
COD	846
Chloride	173
Sulphate	34
Sodium	110
Calcium	123
Magnesium	28
Copper	0.065
Lead	0.050
Iron	43
Zinc	0.6
Cadmium	< 0.004
Chromium	0.03
Nickel	<0.13
Arsenic	< 0.008
Manganese	3.3

3.4.5 Impact on Water Quality

3.4.5.1 Point of egress and defined receiving environment. The primary mode of leachate egress at this site would appear to be to the sewerage works and ultimately to the Fraser River at the Annacis Island STP outfall. The degree of biological and chemical alteration of the leachate in the interceptor conveyance works and at the primary treatment plant would be as discussed for Burns Bog in Section 3.1.5.3. The Fraser River is considered to be the receiving environment.

Other discharge possibilities will depend on the effectiveness of the leachate interception and collection system and the use of the interceptor overflow via the old Brunette interceptor (Figure 3.4.2).

Since the leachate collection works connect to the sewer by gravity, direct control of a negative hydraulic gradient is precluded, thereby possibly allowing some sub-surface lateral leachate migration to neighbouring lands. The control well data shown in Table 3.4.2 relating to monitoring Wells B and H located on the north and south sides of the property respectively, would appear to indicate that hydraulic isolation is provided. The indicator of specific conductance, chloride and others appear to be typical of expected background concentrations. An anomaly to this general conclusion is with the iron concentrations which, in the case of the control wells, appears to be high.

It is understood that the Brunette interceptor overflow is to be utilized from time to time during peak flow periods, and during that period, leachate along with other sewage could be allowed to bypass directly to the Fraser River.

3.4.6 Intended Use. The future use of the Coquitlam Landfill site has not been specifically defined at this time. It would be reasonable to assume, however, that because the location is adjacent to a highly industrialized area and road and rail corridors, some form of structural development is highly probable.

The structural engineering requirements and constraints are therefore of possible future concern with this landfill site, as with several others outlined in this report.

3.5 Leeder Landfill

3.5.1 General. The Leeder Landfill is located in the Fraser Mills area, west of the Port Mann Bridge and at the foot of Leeder Avenue. The site presently comprises approximately 27 ha, is privately owned and operated, and has been in operation since 1965. A Provincial Pollution Control Permit, PR 1350, was issued in 1972, following a fire at the site in 1970 which lasted a month. The filling operation is being carried out in order to raise the elevation of the land for future industrial development. Plate 3.5.1 is an aerial photograph of the site.

3.5.2 Physical Description. The site shown in Figures 3.5.1 and 3.5.2 is bounded on the north by the Canadian Pacific Railway tracks and No. 1 Highway, and to the south by the Fraser River. The eastern lot, 9.3 ha of the original five-lot land parcel, was sold following issuance of the PCB Permit in 1972. The geology of the site, as determined by a soils study in 1963 (Cook, 1963), is illustrated by means of a geologic fence diagram (Figure 3.5.3). Generally, the geologic stratigraphy is comprised of a 1.5 to 2.4 metre silt and clay unit overlying sand and silt. The main sand unit is located 2.1 to 3.6 metres below the surface. A thin surface peat layer (0.09 to 0.9 metres), overlies the silty clay unit on the northern portion of the site. The site geology is therefore typical of deltaic deposition, presumably the result of the Coquitlam River outwash deposits. The important differences with this site are the apparent lack of any substantial peat or organic surficial materials and the proximity of the sands to the surface. According to the site soils report, the lack of peat on this site is apparently anomalous to the area in that lands both to the east and west have more substantial peat units.

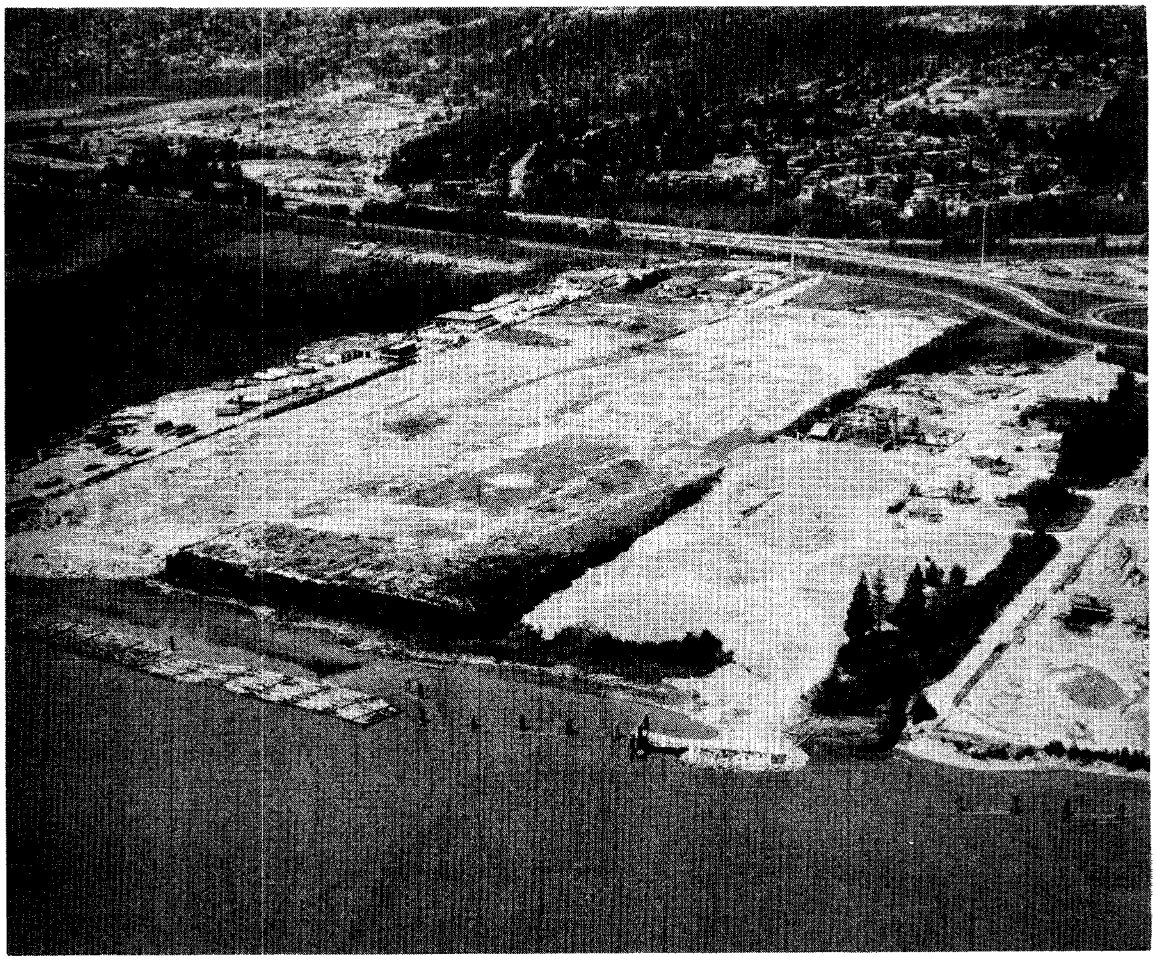


PLATE 3.5.1 GENERAL VIEW OF THE LEEDER LANDFILL SITE

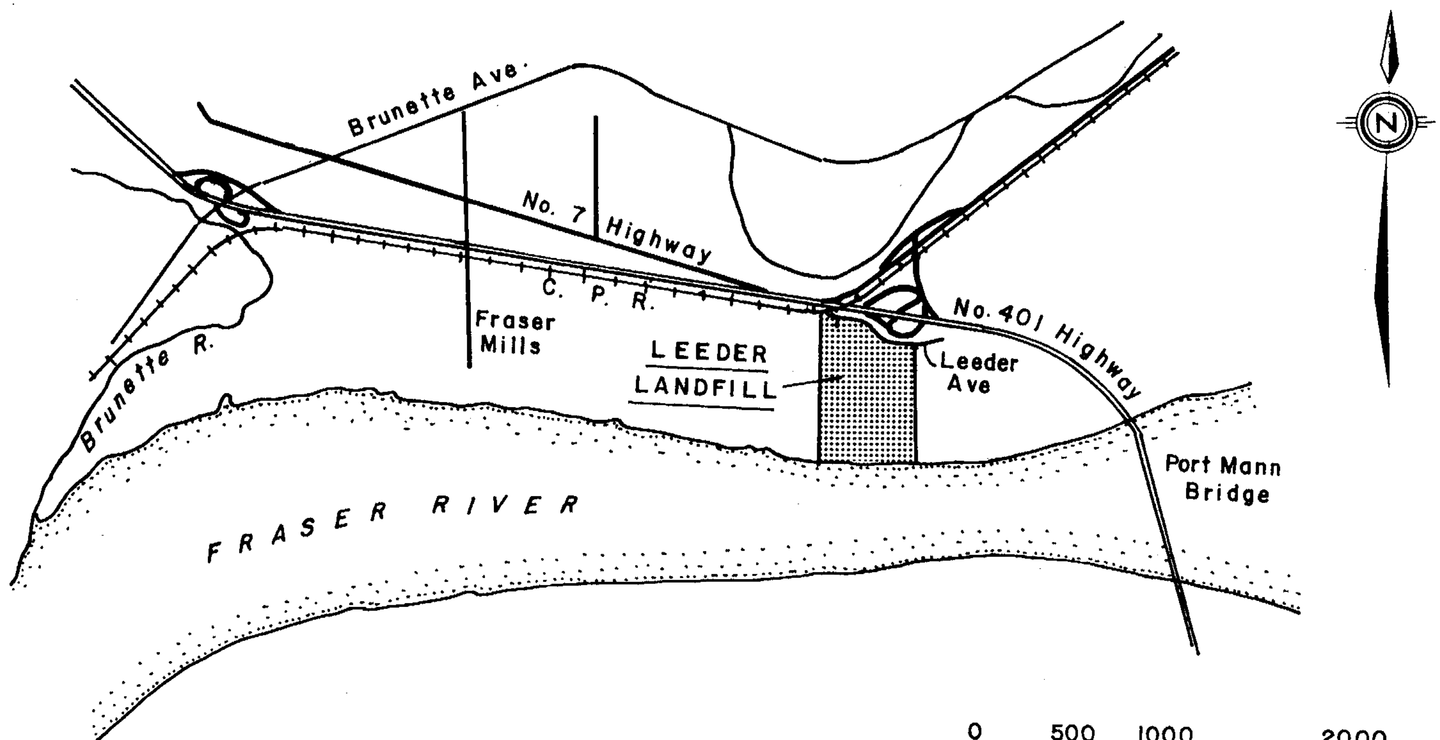


FIGURE 3.5.1 LEEDER LANDFILL LOCATION MAP

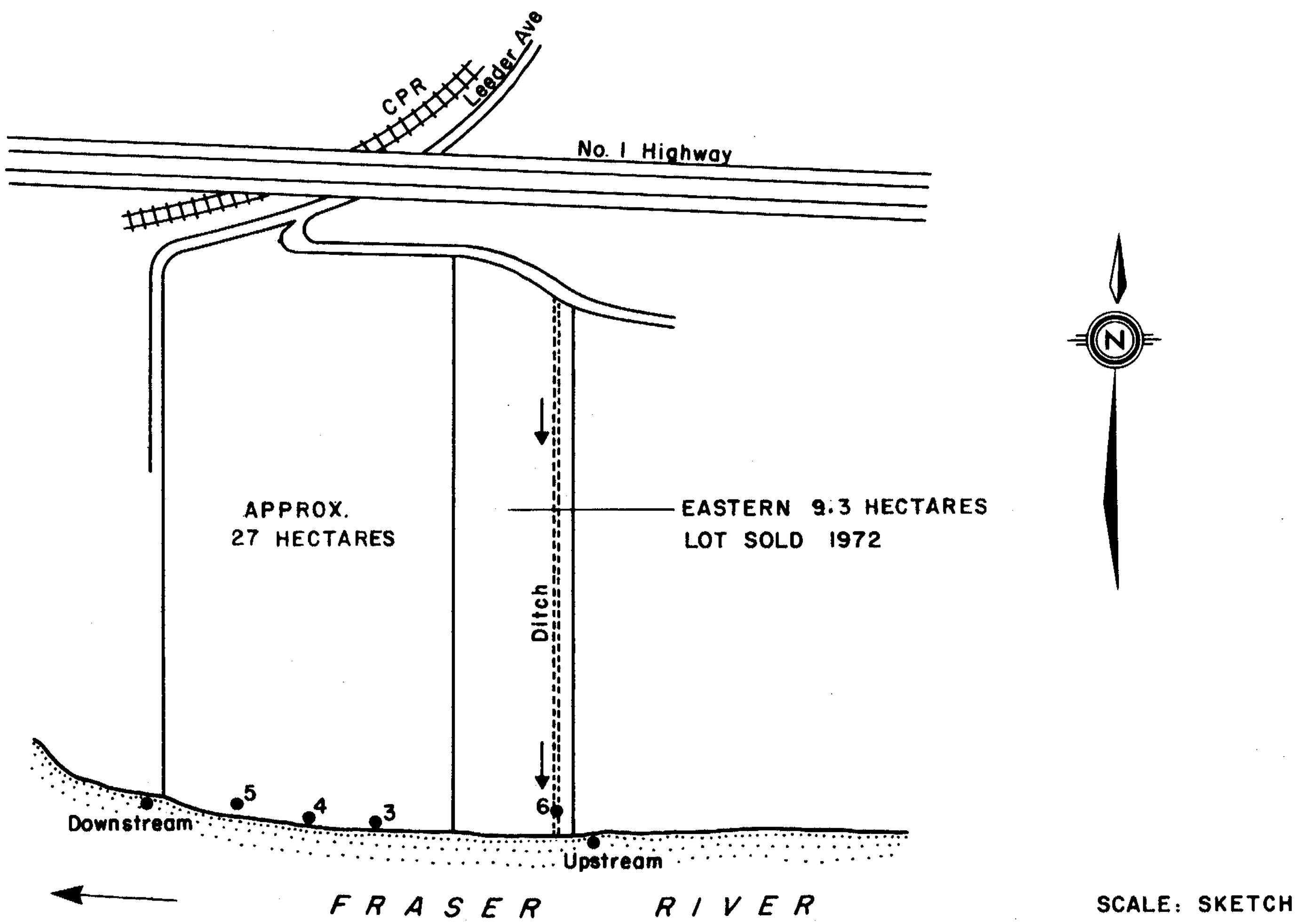


FIGURE 3.5.2 LEEDER LANDFILL SITE PLAN AND MONITORING SITES

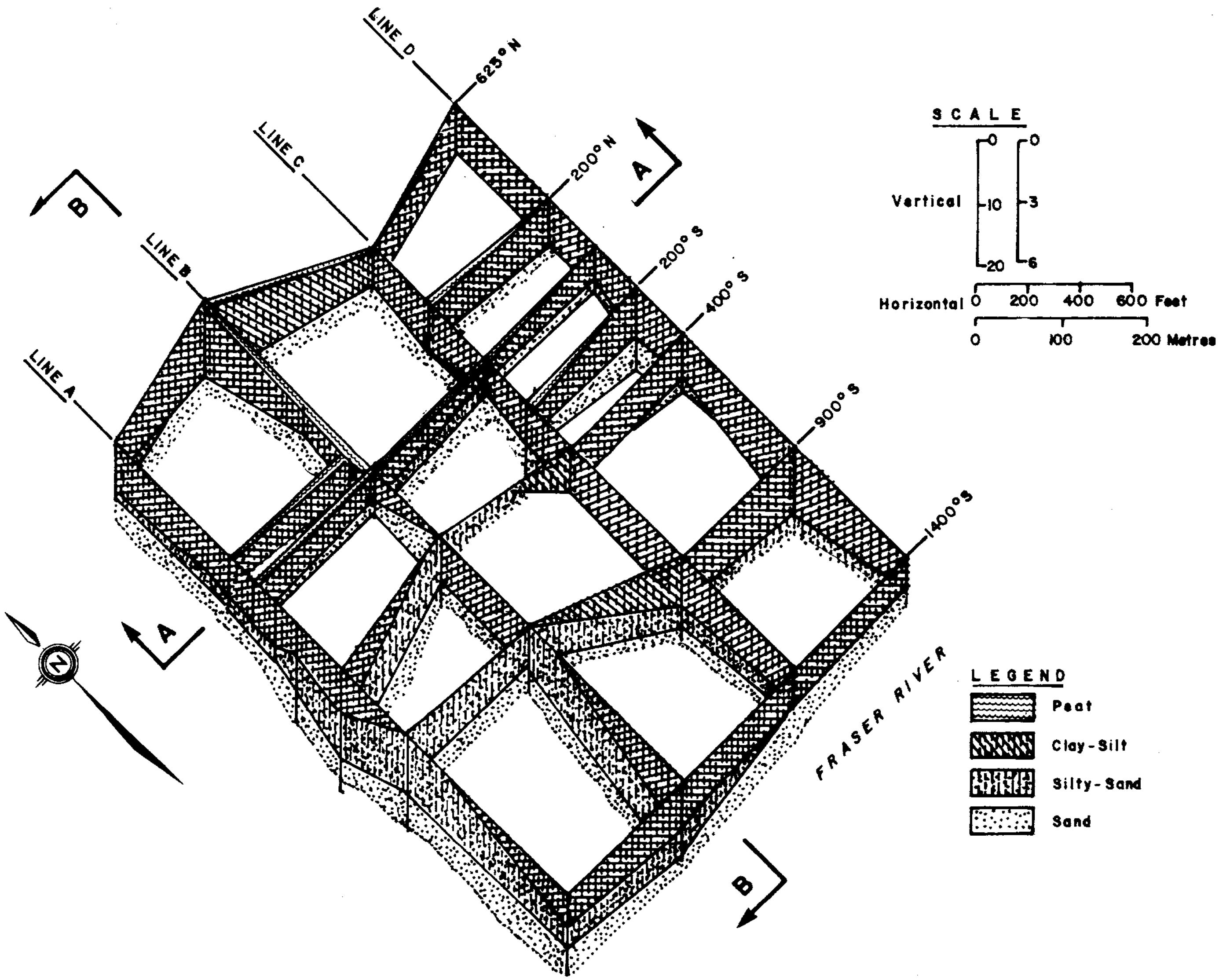


FIGURE 3.5.3 GEOLOGICAL FENCE DIAGRAM, LEEDER LANDFILL

Prior to filling this site, the natural surface drainage was towards the river as a result of the natural slope. Some ponding occurred from runoff in the uplands to the north. The principal surface drainage water course is a surface ditch which flows to the Fraser River along the east side of the original property.

Annual precipitation to this site is estimated to be 150 cm per year. It is postulated that there may be two distinct groundwater units beneath the site: the sand unit waters, and the refuse unit waters. The sand unit waters would be expected to be connected with the Fraser River and the refuse unit waters would be expected to be perched over at least part of the site by virtue of the restricted silt clay permeability. As some of the bore-hole geology indicated a low clay content, there is a high probability of vertical connection between the waters of the upper refuse unit and the lower sand unit. As will be discussed later in this section, the method of site operation may also influence the potential for vertical migration. In terms of horizontal migration, the water movement would be expected to be generally towards the perimeter interception ditches at a rate dependent upon the hydraulic gradient resulting from the mounded water table and the permeability of the underlying soil.

Any firm conclusions in this regard would require a hydrogeologic assessment. However, as will be outlined in Section 3.5.3., sub-surface site waters would appear to be directly connected to the Fraser River by geologic continuity and apparent flow gradients. The result of this postulated flow system would be a diffuse sub-surface leachate discharge to the Fraser River.

3.5.3 Operation. The landfill operation has been carried out since 1965, with predominantly commercial and industrial refuse materials on a five-day per week basis. There is no municipal refuse discharge from municipal collection forces. Any putrescible wastes discharged are therefore associated with the commercial and industrial refuse. International waste has also been discharged to this site under the direction and supervision of Agriculture Canada. Discharge of demolition materials ceased approximately two years ago.

Between 1965 and 1970, the method of operation was straight end-on dumping. Following the 1970 fire, the operation was converted to an excavated cell method. Cells are constructed by excavating the native soils to depths of 1.8 to 2.4 metres. Two additional celled lifts to a total height of 3.6 metres are constructed above the first trenched cell. Excavated material from the trenched cell is used for cell dyke construction, cell preload and cell cover. Cell areas have ranged from approximately 60 m x 120 m in the past, to the present 30 m x 30 m. Refuse is discharged to the cell in layers and is compacted with crawler tractor equipment. Water via fire monitors is sprayed on the fill to prevent fire. The addition of water would also be expected to enhance settlement.

The Pollution Control Branch Permit authorizes the daily discharge of 3060 m^3 of refuse. The actual rate of discharge is, however, in the order of 920 m^3 per day. The estimated daily discharge weight (no weigh scales), based on the above volume figure, is approximately 272 tonnes. Assuming an in-place density of approximately 600 kg/m^3 (2:1 compaction), the areal fill rate based on a total refuse depth of 5.5 m is estimated to be 2 ha per year. A requirement that no refuse be filled within 304.8 m of the Fraser River was placed on the operation; however, there is no indication that this requirement has been followed. It is expected that the fill will be completed by the end of 1979.

3.5.4 Leachate. As with all Lower Mainland landfill sites, the high water inputs due to precipitation necessarily mean that the field capacity will be rapidly exceeded and leachate produced. The nature and movement of the leachate will be dependent upon site-specific operational and hydrogeologic factors. In this regard, the Leeder Landfill site is unique, as there is little or no natural geologic protection against vertical sub-surface migration of leachate, due to the absence of a substantial clay unit.

Although refuse load compressed peat could provide a further line of defence against vertical leachate migration, at this site there are only trace amounts of peats. The method of cell excavation could also enhance the vertical loss of leachate, due to removal of clays or silty clays and/or exposure to permeable sands. An east-west geologic cross-section, shown in Figure 3.5.4, illustrates the peat-clay-silt-sand stratigraphy with a theoretical cell excavation base superimposed. It is seen that it is theoretically possible to intercept the more permeable silt-sand unit, thereby exposing the underlying aquifer to leachate migration. Again, as the hydrogeologic factors at this site are complex, it is difficult to make firm judgements on the fate of site waters and leachate without a thorough hydrogeologic assessment.

An estimate of the site water balance, based on 150 cm precipitation over the 27 ha, yields an annual average flow of 0.011 m^3/sec. This figure does not consider Fraser River ingress or the addition of fire prevention water to the fill, which could be significant factors. As the site is not considered impermeable to vertical flow, the proportion of surface water which is intercepted and infiltrates is a further important but unknown factor. In this regard, the site operation in terms of cell dyke construction, ditch location, and invert elevations and water ingress variations due to Fraser River level, would bear significantly on the flow system.

Figure 3.5.5 shows a north-south geologic cross-section which illustrates areas in which there is no clay unit and where the sands unit is exposed at or close to the surface. The theoretical cell excavation base line shows, in this case, the interception of the sands unit.

3.5.5 Impact on Water Quality

3.5.5.1 Point of egress. An accurate description of the leachate flow system is complicated by the site hydrogeology. It would appear that leachate egress to both surface waters and groundwaters is probable due to the proximity of the permeable sands and the absence of a substantial clay or peat layer at the soil surface.

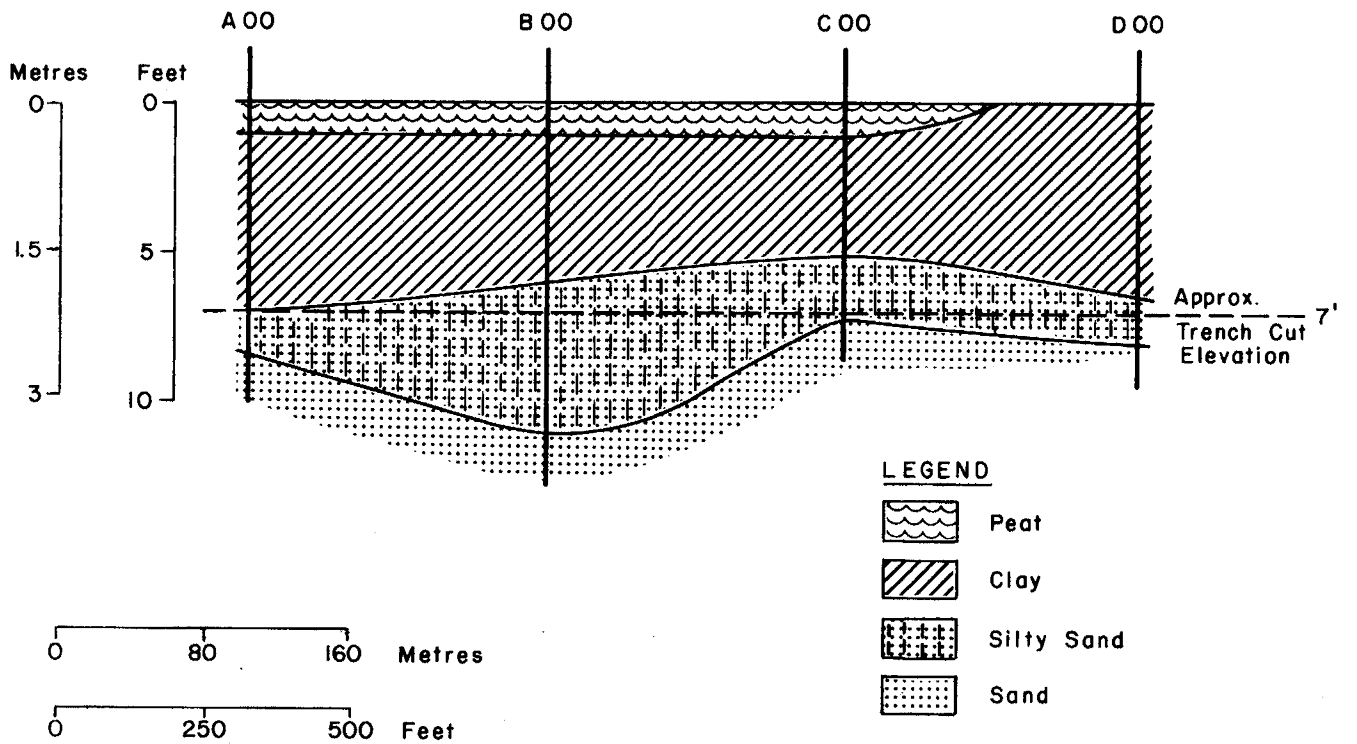


FIGURE 3.5.4 GEOLOGICAL CROSS-SECTION LINE A - A (EAST-WEST) LEEDER LANDFILL

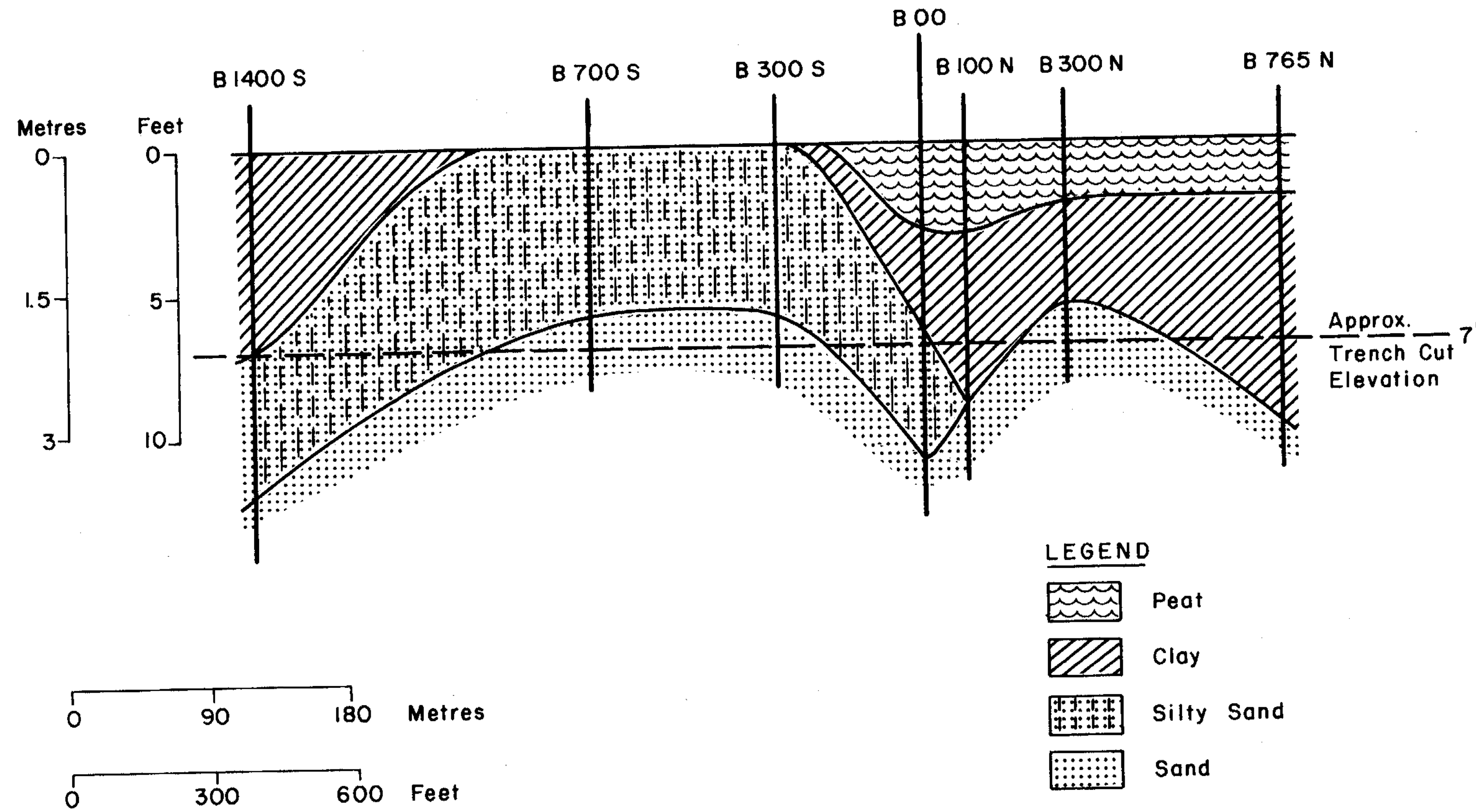


FIGURE 3.5.5 GEOLOGICAL CROSS-SECTION LINE B-B (NORTH-SOUTH)
LEEDER LANDFILL

As the sand unit would be directly connected to the Fraser River, it would be expected that the leachate will enter the Fraser River via a diffuse discharge through the connection of sands unit waters. Some variations in flow of the sands unit waters would be expected due to Fraser River level variation and the resultant alteration of hydraulic gradient. However, the general flow path would be expected to be towards the river, due to the uplands area to the north. A hydrogeologic investigation involving water quality and piezometric surface elevation monitoring could confirm the nature and proportion of this leachate egress mechanism.

Leachate will be produced and will be discharged to the Fraser River whether by groundwater or surface water means. An important point to note, however, is that the silt and clay fractions in the surficial geology would be expected to provide at least some of the leachate renovation as it migrates. The leachate monitoring program established for this site under Pollution Control Permit PR-1350 involves surface stations and wells as shown in Figure 3.5.2.

Table 3.5.1 shows the monitoring results for the surface ditch and Fraser River sampling points. Table 3.5.2 shows the monitoring results for the wells.

The monitoring well results are difficult to characterize in that there is no upgradient control well for comparison. As it is postulated that the silty-sand and sand units are directly connected to the Fraser River with some localized recharge and discharge mechanism, the Fraser River water character could be used for comparative purposes. Significant increases in chloride, sulphate, total iron, organic carbon, specific conductance, dissolved solids, alkalinity and ammonia would tend to confirm the existence of some leachate movement in the sub-surface waters. It should be noted however that the values would not be considered high.

By way of comparison, the contaminant concentrations for the sub-surface silty-sands leachate egress at the Richmond Landfill tend to be considerably higher, especially in the indicator parameters of iron and chloride.

TABLE 3.5.1 MONITORING DATA, SURFACE WATERS, 1972-1978 - LEEDER LANDFILL

Parameter	Surface Ditch Station #6			Fraser River					
				Upstream			Downstream		
	No. of Samples	Range Min. - Max.	Avg.	No. of Samples	Range Min. - Max.	Avg.	No. of Samples	Range Min. - Max.	Avg.
pH	10	6.5 - 7.9	7.07	11	6.9 - 8	7.4	10	7.1 - 8	7.6
Dissolved Solids	3	130 - 386	254	4	40 - 74	63.5	3	60 - 74	69.3
Spec. Cond. (μmho/cm)	11	80 - 410	227	11	45 - 220	100	11	65 - 195	116
Total Alkalinity	5	33 - 117	58.8	5	24.1 - 43	37.7	5	32.7 - 45	40.7
Organic Carbon	5	9 - 109	40.6	6	< 1 - 9	4.3	5	< 1 - 10	5.2
Chloride	4	16.7 - 50.4	36	5	.68 - 3.6	2.3	92	1.3 - 5	5
Ammonia	4	.17 - .336	0.26	5	.01 - .048	.033	5	.033 - .198	.081
Nitrate	1	.49	.49	1	.25	.25	1	.22	.22
Kjeldahl N	-	-	-	-	-	-	-	-	-
Sulphate	6	< 5 - 10.7	7.9	6	< 5 - 8	6.1	6	< 5 - 8.8	< 6.6
Tannin & Lignin	5	1 - 100	35.8	5	.5 - 1.4	.94	5	.5 - 1	0.84
Dissolved Oxygen	3	5 - 8.4	7.2	3	9.4 - 11.7	10.6	10	9 - 14.3	11.9
Iron (D)	1	.5	.5	1	.1	.1	1	.1	.1
Iron (T)	4	2.5 - 7.2	4.9	4	.6 - 14	5.1	4	1 - 19.5	7.2
Magnesium (D)	5	2.7 - 8.2	4.9	5	1.6 - 3.1	2.5	5	2 - 3.4	2.72
Manganese (T)	4	.26 - .82	.56	4	.03 - .3	.13	4	.04 - .4	.17
Sodium (D)	5	11 - 35.2	22.9	5	1.9 - 3.9	2.9	5	2.6 - 4.2	3.5

Units mg/l, except as noted.
D = Dissolved
T = Total

TABLE 3.5.2 MONITORING DATA, WELLS, 1972-1978 - LEEDER LANDFILL

	MONITORING WELLS								
	Station #3			Station #4			Station #5		
Parameter	No. of Samples	Range Min. - Max.	Avg.	No. of Samples	Range Min. - Max.	Avg.	No. of Samples	Range Min. - Max.	Avg.
pH	9	6.3 - 7.0	6.6	9	6.7 - 7.3	7.0	15	5.2 - 7.8	7.1
Dissolved Solids	4	106 - 366	237	4	154 - 570	404	6	1048 - 1866	1446
Spec. Cond. (μmho/cm)	9	148 - 710	440	9	221 - 917	618	15	1200 - 2100	1651
Total Alkalinity	7	41 - 305	163	6	104 - 459	274	9	168 - 520	330
Organic Carbon	6	20 - 34	27	5	16 - 55	27	10	21 - 142	54.5
Chloride	6	2.5 - 38.1	19.8	6	3.2 - 30.7	15.8	10	4.8 - 24	10.2
Ammonia	6	.236 - 1.39	.61	7	.096 - .93	.44	8	.074 - 8	1.8
Nitrate	-	-	-	-	-	-	3	.05 - .55	-
Kjeldahl N	-	-	-	-	-	-	3	3.0 - 7.0	-
Sulphate	8	< 5 - 91.5	37.3	8	<5 - 39.2	19.3	11	305 - 974	679
Tannin & Lignin	7	1.2 - 5	3.2	7	1 - 10	3.9	11	2 - 10	6
Dissolved Oxygen	-	-	-	-	-	-	-	-	-
Iron (D)	1	.1	.1	1	28.5	28.5	4	<0.1 - 2.3	< .68
Iron (T)	5	31 - 75	51	5	23 - 92.5	57.9	6	12 - 167	61
Magnesium (D)	2	.17 - .35	.26	6	5.4 - 38.6	21.6	-	-	-
Manganese (T)	5	1.1 - 2.4	1.7	5	.81 - 4.92	2.8	6	.13 - 2.09	.89
Sodium (D)	7	3.6 - 25.4	14.0	6	6.9 - 21.2	13.8	11	32.3 - 51.9	38.9

Units in mg/l, except where noted.
D = Dissolved
T = Total

The apparent attenuation at the Leeder Landfill is attributed to dispersion and dilution due to Fraser River ingress and other groundwater sources, and contaminant renovation by the silt fractions and combinations thereof. In this regard, it is seen that relatively low concentrations of the conservative chloride ion indicate that dispersion and dilution may be the primary attenuation mechanism. Of the three wells, Well No. 5, which is in line with the more permeable silty-sands unit, tends to show higher contaminant levels for most parameters. One interesting contaminant anomaly is the sulphate concentration for Well No. 5, which has shown consistently high levels. A possible explanation may be the leaching of gypsum wallboard materials.

The monitoring results for the south end of the easterly ditch at Station 6 again cannot be comparatively evaluated due to the absence of upstream control. Generally, it is seen that there are indications of minor levels of contamination. By comparison with other landfill perimeter ditch systems, the values are significantly lower, in most cases by an order of magnitude. The unknown dilution and upstream factors will preclude any firm judgements on the cause-effect results of the landfill leachate. One interesting combined parameter is tannins and lignins which shows some high levels. These levels may be attributable to wood waste in the fill and possibly the upstream peats.

In terms of the Fraser River monitoring results, many of the contaminant parameters show slight increases downstream. It is noted, however, that the increases in all cases are very slight and are not necessarily significant. Certainly, the large dilution capability of the Fraser River would be expected to considerably mask the leachate discharges, especially in view of their diffuse character.

Estimates of annual contaminant loading and flow estimates cannot reasonably be carried out for this site. As a large portion of the leachate discharge is assumed to be of a sub-surface diffuse nature, any loading extension would necessarily require quantification of the aquifer flow system among other data. Hence, suffice it to say, that the contaminant loading would be expected to be more or less in line with other sites discussed in this report. Important variables at this site which require restatement in respect to their possible influence on the resultant leachate are as follows:

- the silt sand unit would be expected to provide some contaminant attenuation,
- the refuse materials involve a minor portion of municipal refuse,
- the diffuse rather than point source discharge.

3.5.6 Intended Use. The intended use of this site is for industrial and/or commercial development. The location adjacent to the Fraser River, road and rail corridors, and industrial establishment makes this site ideal for industrial development. The structural factors of any future development will depend upon the type of development (bearing loads, etc.) and the density character of the in-place refuse fill.

4 LARGE MUNICIPAL LANDFILLS (CLOSED)

Within the study area there are three large landfills which were operated for the disposal of municipal, commercial, and industrial wastes, but which are now inactive. They are: the Kerr Road Landfill in the City of Vancouver; the Terra Nova Landfill which is privately owned and located in the District of Coquitlam; and the Stride Avenue Landfill in the Municipality of Burnaby.

4.1 Kerr Road Landfill

4.1.1 General. The now closed Kerr Road Landfill was utilized by the City of Vancouver for refuse disposal between 1952 and 1966. This site is located in the extreme southeast corner of the City immediately above the North Arm of the Fraser River. Plate 4.1.1 is an aerial photograph of the site.

4.1.2 Physical Description. The fill site, located on the south slope of Vancouver, was originally a gully much like those that can be seen above Marine Drive to the east. Figure 4.1.1 shows two cross-sections (North-South, and East-West) depicting original and present ground surfaces. The slope generally consists of Pleistocene deposits of sands and gravels with some silts and clays normally blanketed with a marine till. A cross-section drawn through the site showing till underlain by sands supports this generalized geology (Figure 4.1.2). Only drill hole DH4 is located in the landfill site, drill holes DH1 and DH2 are located 213 metres and 425 metres, respectively, to the east of the landfill property. The log of DH4 shows 0.9 metres of very dense sand and cobbles underlain by medium fine dense sand with some silt to the bottom of the hole at 9.1 metres. It is reasonable to assume that the landfill site is underlain throughout by this sand; however, it may not be reasonable to assume an absence of the till mantle, which is present in the other three holes in the area. The filled site covers approximately 38 ha and is generally graded to the south at about a 10% slope. Figure 4.1.3 is a site plan overlaid with the present and past topography and fill limits.

PLATE 4.1.1 GENERAL VIEW OF THE KERR ROAD LANDFILL SITE

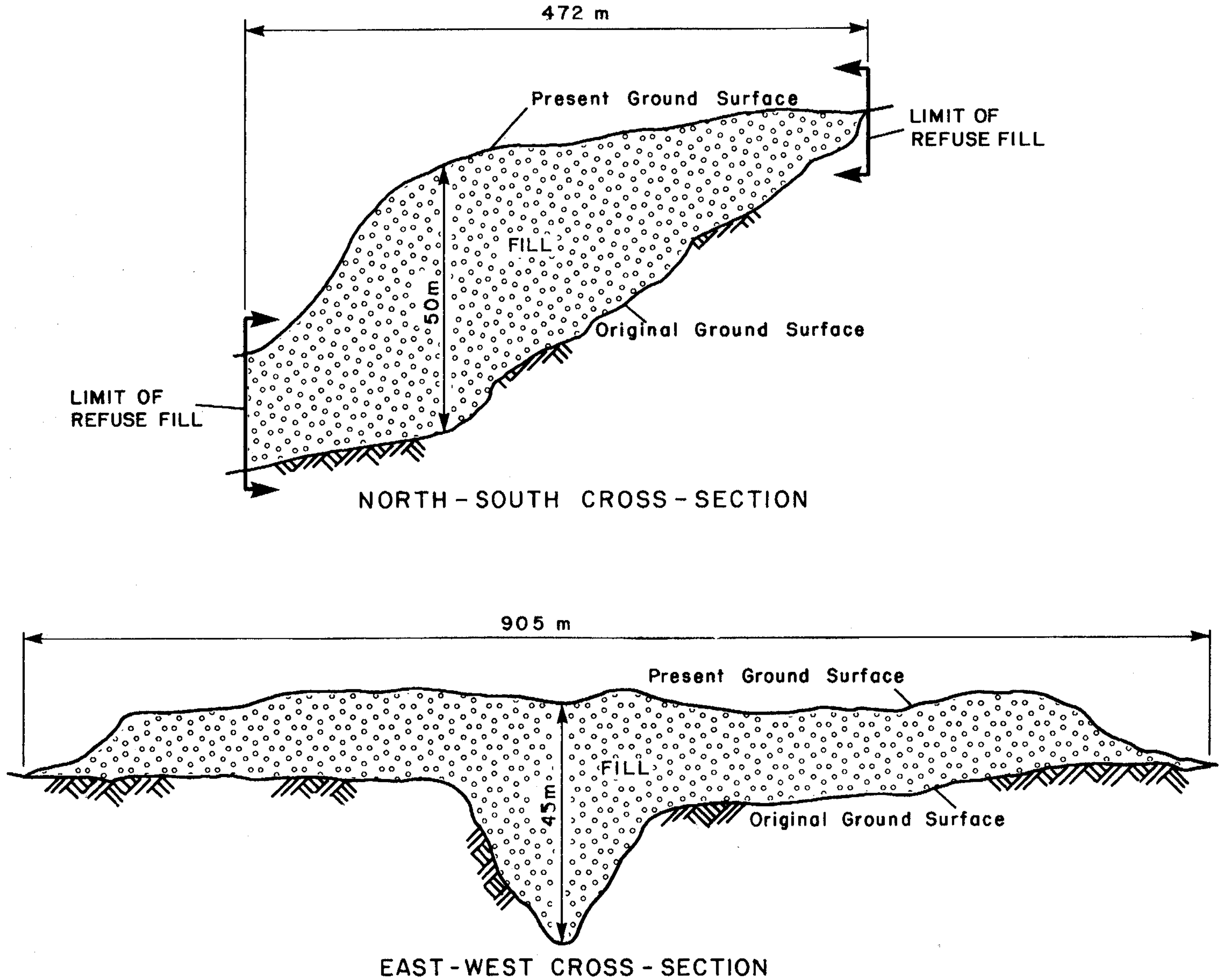


FIGURE 4.1.1 CROSS - SECTIONS - KERR ROAD LANDFILL - CITY OF VANCOUVER

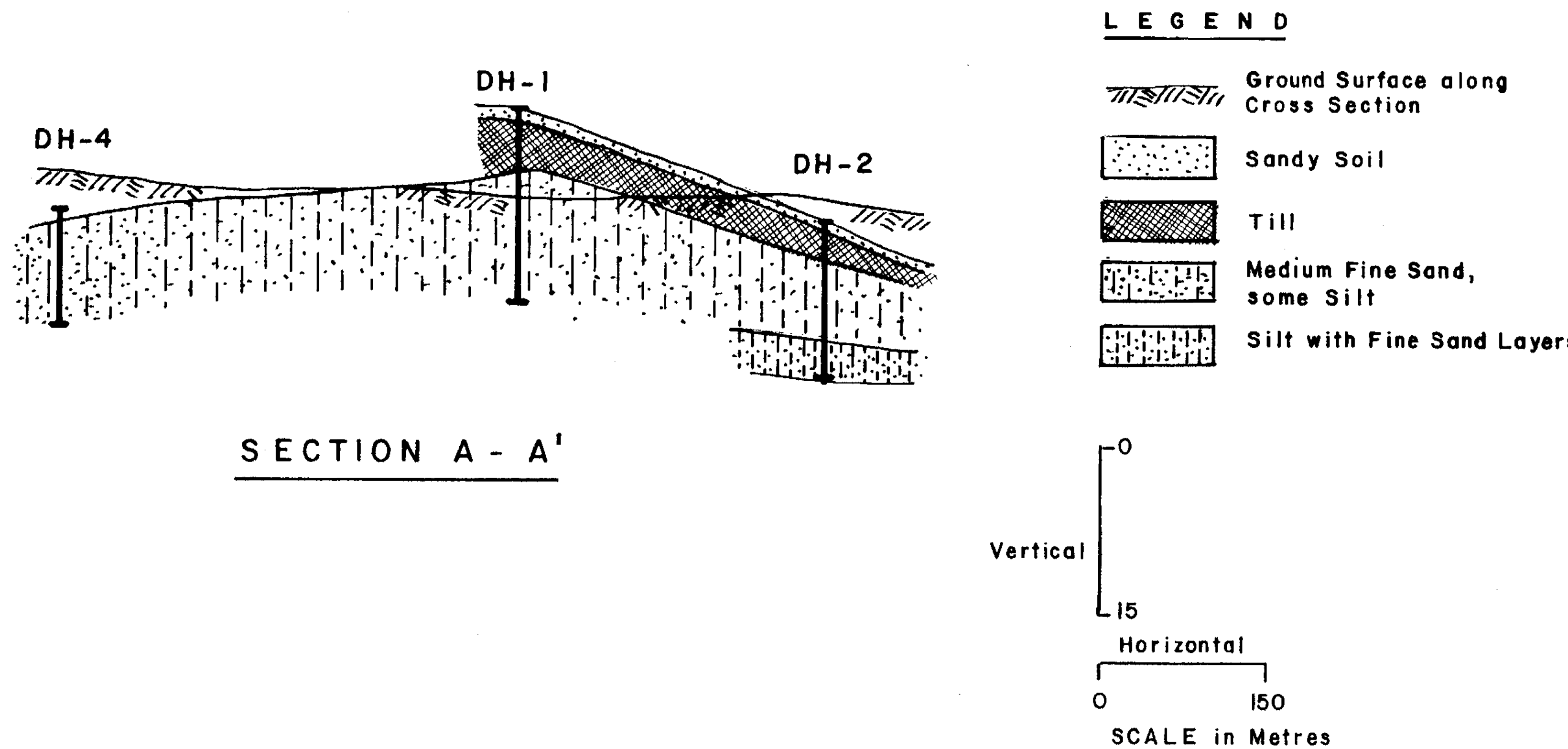


FIGURE 4·1·2 GEOLOGICAL CROSS-SECTION - KERR ROAD LANDFILL
CITY OF VANCOUVER
(Source: Taken from Drill Logs prepared by Paul M. Cook, P. Eng.)

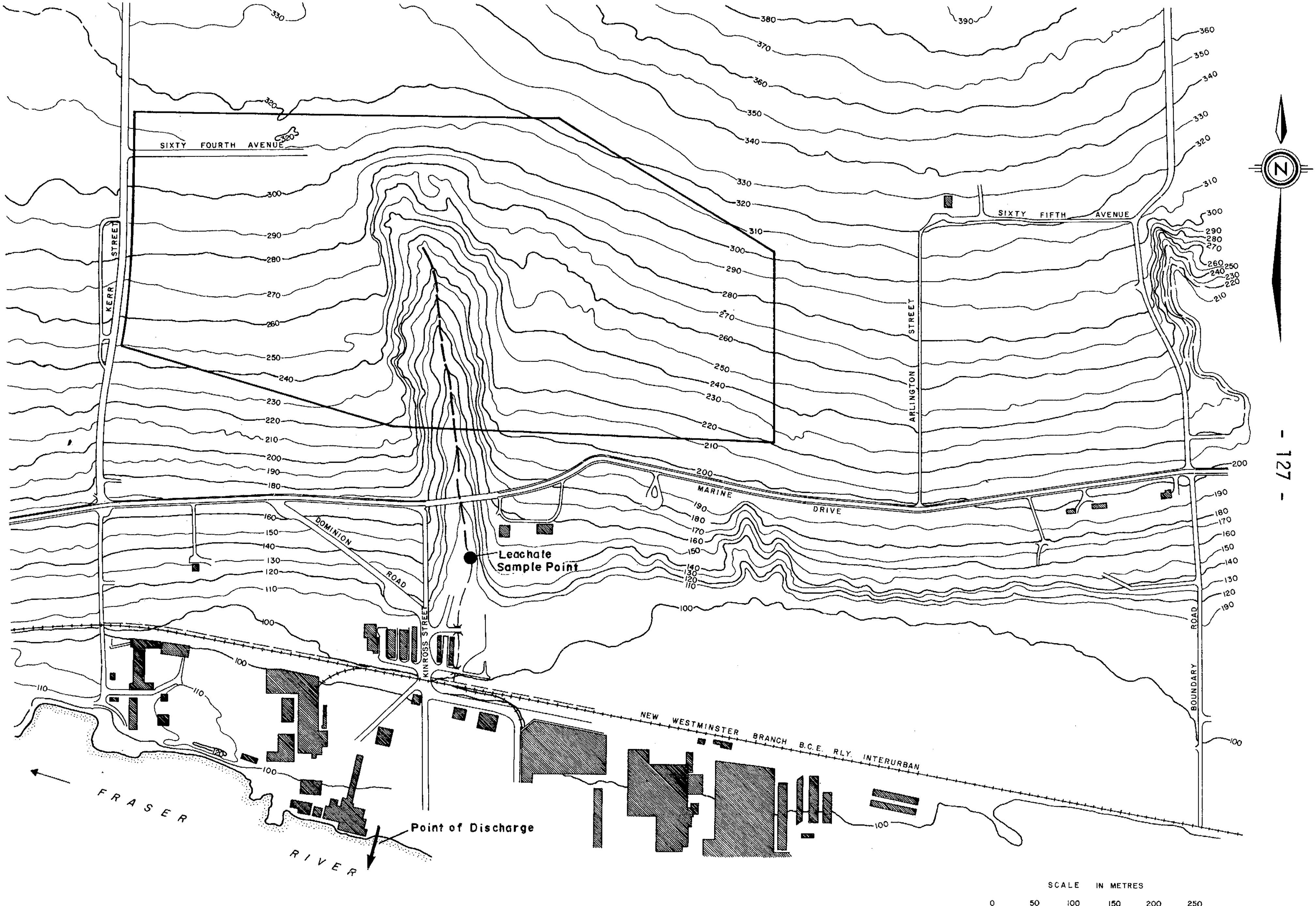


FIGURE 4.1-3 ORIGINAL AND PRESENT CONTOURS - KERR ROAD LANDFILL - CITY OF VANCOUVER

4.1.3 Operations. The Kerr Road site was used by the City of Vancouver for the disposal of all types of refuse including septic tank pumpings and clean mineral soils for some 15 to 17 years. The site was used after 1966 for two or three years for the disposal of clean fill. During the 1960's, concurrent with the filling operation, was sand excavation. An estimated 3 830 000 m^3 of refuse was placed on the site. This estimate is based on calculations using the Equal Depth Contour Method and contour maps of the site before and after filling. A considerable percentage of the fill was excavated soils from elsewhere in the City, usually heavy clay type soils (P. Herring, 1978, personal communication). Burning was not carried out at the site, although fires did occur particularly during labour disputes. There was no construction of cells during the filling operation, nor was there any intentional covering of the working face. During the filling of the site the gully was culverted. This culvert was extended under Marine Drive and presently discharges through a flood gate to the North Arm of the Fraser River just to the west of Kinross Street. The depth of fill in the site varies from zero along the edges to over 49 metres along the centre line of the gully. Over half the site has 12 metres or less of fill. The site was completed with the placement of some 1.5 metres of soil cover over the entire area.

4.1.4 Leachate. Some 12 years after the closing of the Kerr Road Landfill leachate, although of very low strength, can be measured discharging from the site (EPS, 1978, 1979). Analyses of the leachate in the culvert have been undertaken somewhat continuously since 1972: by the GVRD in 1972, by the City of Vancouver in 1974, 1975 and 1976, and by EPS in 1978 and 1979. Since 1972, the concentrations of most of the constituents have fallen off. Concentrations against time are plotted in Figures 4.1.4 and 4.1.5 for seven constituents COD, specific conductance, NH3, Cl, Cu, Zn and Pb. The concentrations of COD and NH3 are seen to fall off rapidly during 1973 with little change thereafter. The other constituents are seen to decrease in concentration up to about late 1976 with little change seen during the 1978, 1979 samplings.

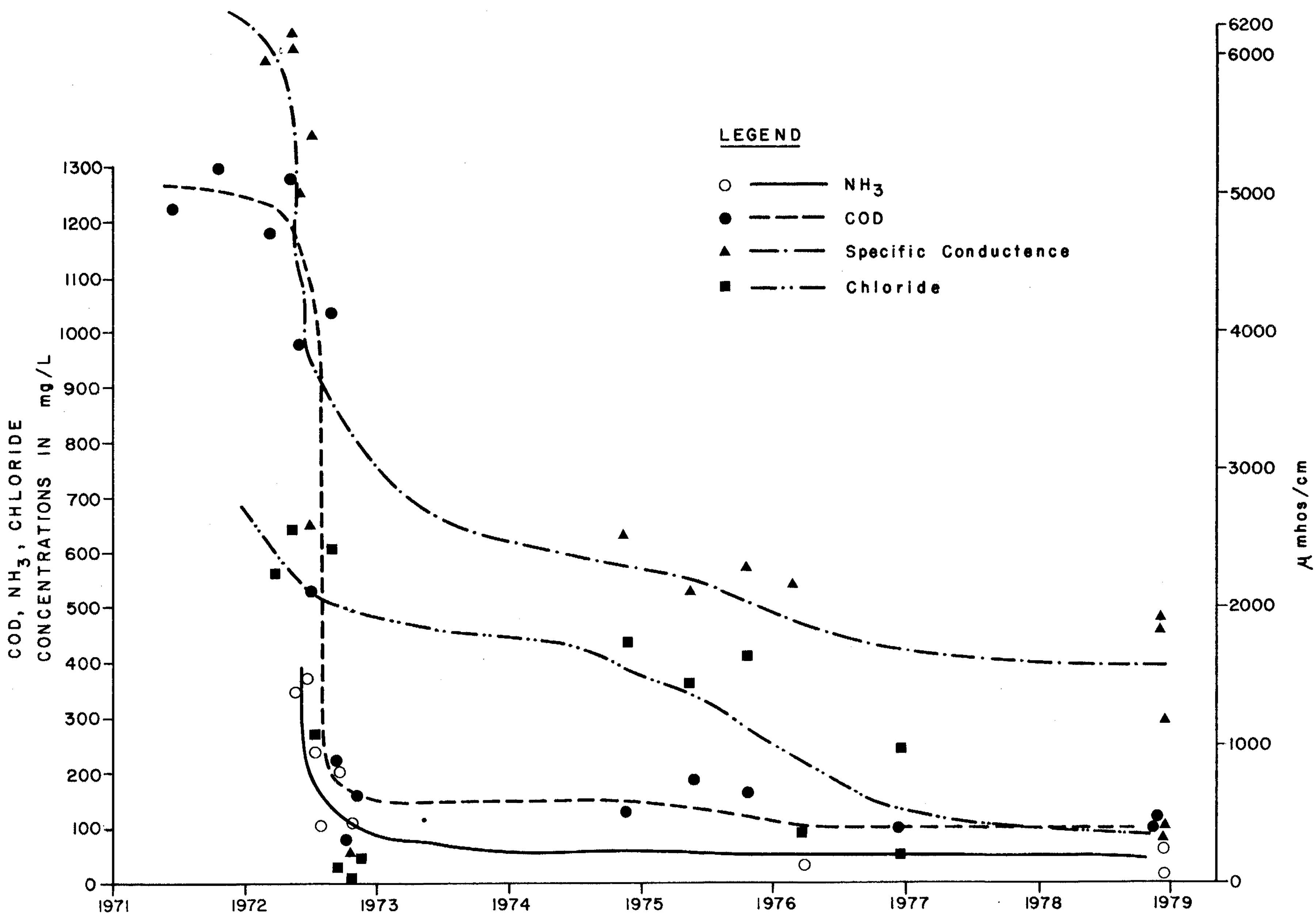


FIGURE 4.1.4 LEACHATE CONSTITUENT CONCENTRATIONS - KERR ROAD LANDFILL - CITY OF VANCOUVER

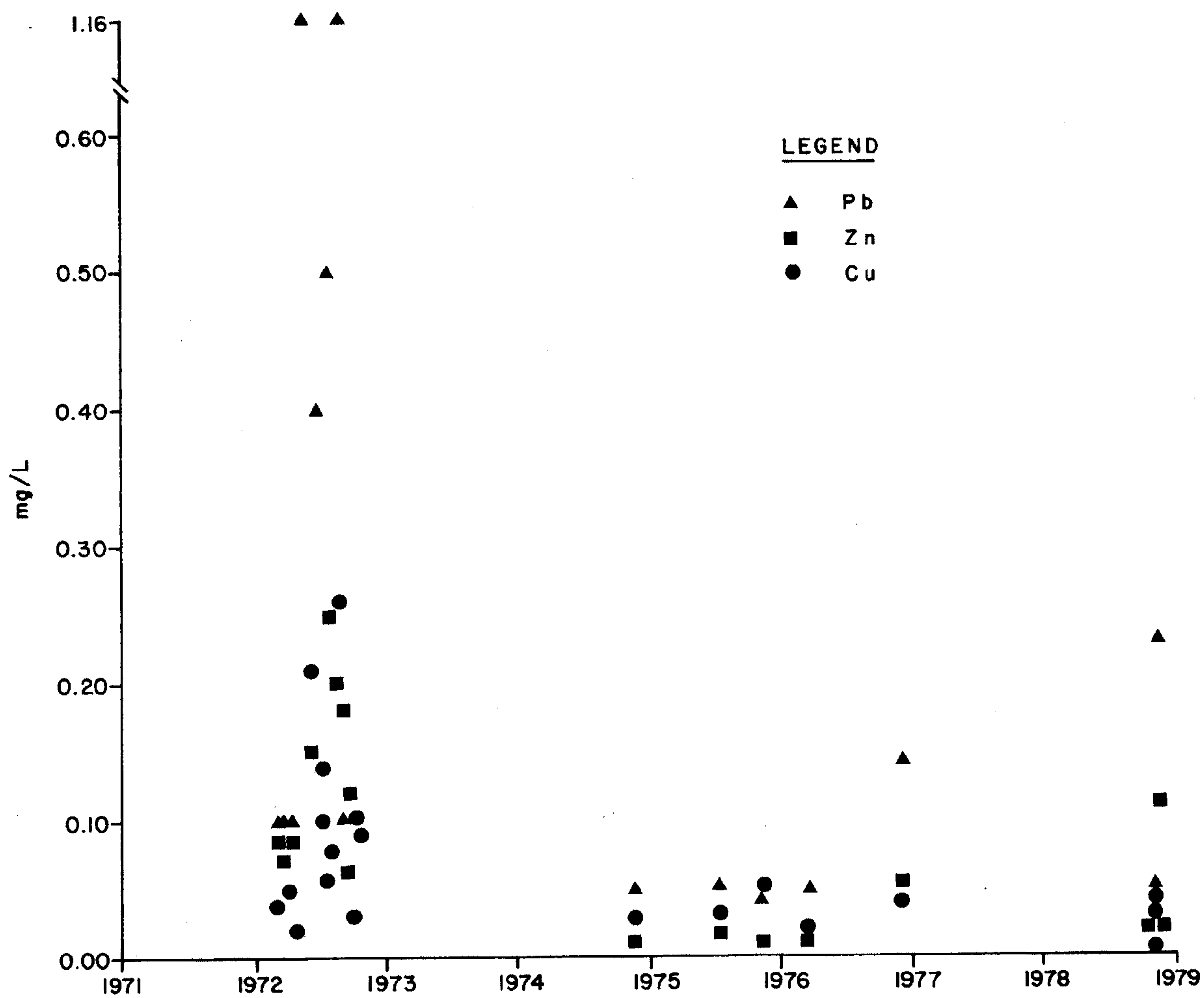


FIGURE 4.1.5 LEACHATE METAL CONCENTRATIONS - KERR ROAD LANDFILL - CITY OF VANCOUVER

Table 4.1.1 summarizes the results of six samples taken from November 1978 to April 1979. These samples are considered to be representative of the leachate presently coming from the culvert under the landfill (EPS, 1978, 1979).

The flow in the culvert coming from the landfill was estimated to be 0.0015 m^3/sec under dry weather flow and up to 0.0076 m^3/sec after two days of rain. It is likely that this value will be higher after prolonged rainfall and lower during the summer. For the purpose of loading calculations of constituents emanating from the culvert, an average yearly flow of 0.0038 m^3/sec is used.

The 0.0038 m^3/sec value cannot represent the total leachate discharge from the Kerr Road site and, may in fact, include some groundwater from above the site. Theoretically, the leachate flow from the site on a yearly averaged base should be about 0.0053-0.0068 m^3/sec. That range of values is based on 127 cm of precipitation, a runoff coefficient of 0.25 to 0.35, and an evapotranspiration of 38 cm of the amount that infiltrates. Monitoring of wells in the Lower Fraser Valley has shown that between the end of March and the beginning of November there is virtually no groundwater response to precipitation (H. Liebscher, 1978, pers. comm.). That particular observation is not applicable to bog settings such as the Burns Bog or Richmond Landfills. It is assumed that the leachate which does not enter the culvert infiltrates through the sands and silts and eventually seeps into the North Arm of the Fraser River. The concentrations of the leachate constituents at the point of the groundwater entering the Fraser River is unknown. They would be expected to be considerably lower than those measured in the culvert because of attenuation afforded by movement through the sands and silts.

Using the 1978 and 1979 values in Table 4.1.1 and a flow rate of 0.0038 m^3/sec, the constituent loadings to the Fraser River from the direct leachate discharge were calculated and are given in Table 4.1.2. All the reservations that were placed on such calculations for the active sites must also be placed on these loadings. Based on the limited

TABLE 4.1.1 LEACHATE MONITORING DATA, 1978-1979 - KERR ROAD LANDFILL - CITY OF VANCOUVER

Paramter	No. of Samples	Maximum	Minimum	Average
pH (units)	6	8.0	7.7	79
NFR	6	90	17	37
COD	6	169	100	116
Spec. Cond. (μmhos/cm)	6	2390	1160	1960
Sulphate	6	10.0	4.26	7.5
Chloride	6	135	77.5	105
Total Phosphate	6	0.4	0.25	0.33
Total Alkalinity as $CaCO_3$	5	1050	760	870
Aluminum	6	0.18	< .09	0.08
Arsenic	6	< .15	< .15	< .15
Barium	6	0.84	0.36	0.57
Calcium	6	94	37	57
Cadmium	6	< .015	< .015	< .015
Cobalt	6	< .02	< .02	< .02
Chromium	6	< .02	< .02	< .02
Copper	6	.034	< .01	.01
Iron	6	13.0	7.3	9.5
Mercury	6	< .1	.1	< .1
Magnesium	6	53	26	41
Manganese	6	.97	.07	.30
Molybdenum	6	.15	< .15	< .15
Sodium	6	256	99	173
Nickel	6	< .08	< .08	< .08
Lead	6	.23	< .09	.08
Antimony	6	< .08	< .08	< .08
Selenium	6	< .15	< .15	< .15
Tin	6	< .2	< .2	< .2
Strontium	6	1.46	.566	1.00
Titanium	6	< .009	< .009	< .009
Vanadium	6	< .09	< .09	< .09
Zinc	6	.112	< .045	.04
Silicon	6	15.3	13.4	14.6
Potassium	4	85	38	63

mg/l except as noted.

TABLE 4.1.2 LEACHATE CONSTITUENT LOADINGS TO THE FRASER RIVER - KERR ROAD LANDFILL - CITY OF VANCOUVER

Parameter	Avg. Conc. (mg/l)	Avg. Daily Loading (kg/day)
COD	116	38
Sulphate	7.5	2.5
Chloride	105	35
Total Phosphate	.33	0.11
Total Alkalinity as $CaCO_3$	870	285
Total Ammonia	58	19
Aluminum	0.08	0.03
Barium	0.57	0.19
Calcium	57	18.6
Copper	.01	.003
Iron	9.5	3.0
Magnesium	41	13
Manganese	.30	0.1
Sodium	173	56
Lead	.08	0.03
Strontium	1.00	.33
Zinc	.04	0.01
Silicon	14.6	4.8
Potassium	63	20

data samples taken in 1976, 1978, and 1979, there is no apparent trend of decreasing concentrations; therefore, one must assume for the next few years that the concentrations will persist and consequently the loadings to the river will also persist.

4.1.5 Impact on Water Quality. The discharge to the Fraser River is generally submerged and when exposed there is no apparent plume. No analysis of the receiving water at that point has been done. Two bio-assays using rainbow trout were conducted on the leachate: one, on January 2, 1979; the other, on February 16, 1979. The January 2 sample had an LT_{50} at 100% concentration of 34 minutes, while the February 16 sample had a 96-hour LC_{50} of 24%.

4.1.6 Intended Use. The landfill site is now the property of the Vancouver Parks Board. The site will be used to extend the adjacent Fraserview Golf Course.

4.2 Terra Nova Landfill

4.2.1 General. Terra Nova Landfill, also known as Inter-Tidal, is located on the property of the Crown Zellerbach's Fraser Mills Division and is administered through their development arm, Ven Dev Enterprises Ltd. The site was operated as a refuse landfill from 1965 to September 1975, and is presently utilized for the storage of hog fuel, dredged sand and manufactured soil. Plate 4.2.1 is an aerial photograph of the site.

4.2.2 Physical Description. The Terra Nova Landfill is situated on the north bank of the Fraser River in the District of Coquitlam. The site is generally bounded on the north by the CPR right of way and the Trans-Canada Highway, to the west by the Crown Zellerbach Mill, and to the east by the Leeder Landfill.

The site is underlain by recent Fraser Delta deposits (Hoos *et al*, 1974) of peat, silt and peat, loose-silt and sands, and dense sands and with deep Pleistocene deposits of a marine clay silt and dense glacial till. Figure 4.2.1 depicts a cross-section north to south

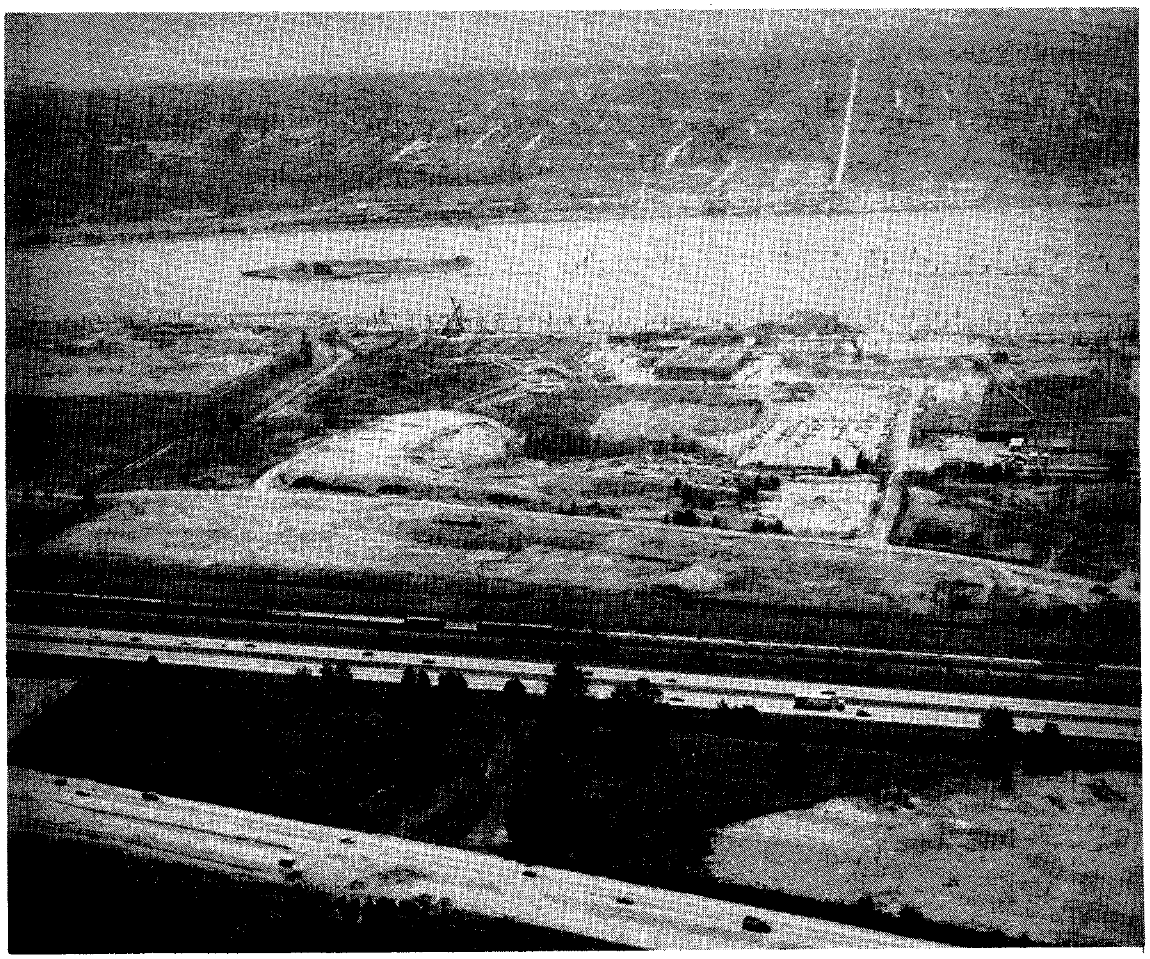


PLATE 4.2.1 GENERAL VIEW OF THE TERRA NOVA LANDFILL SITE

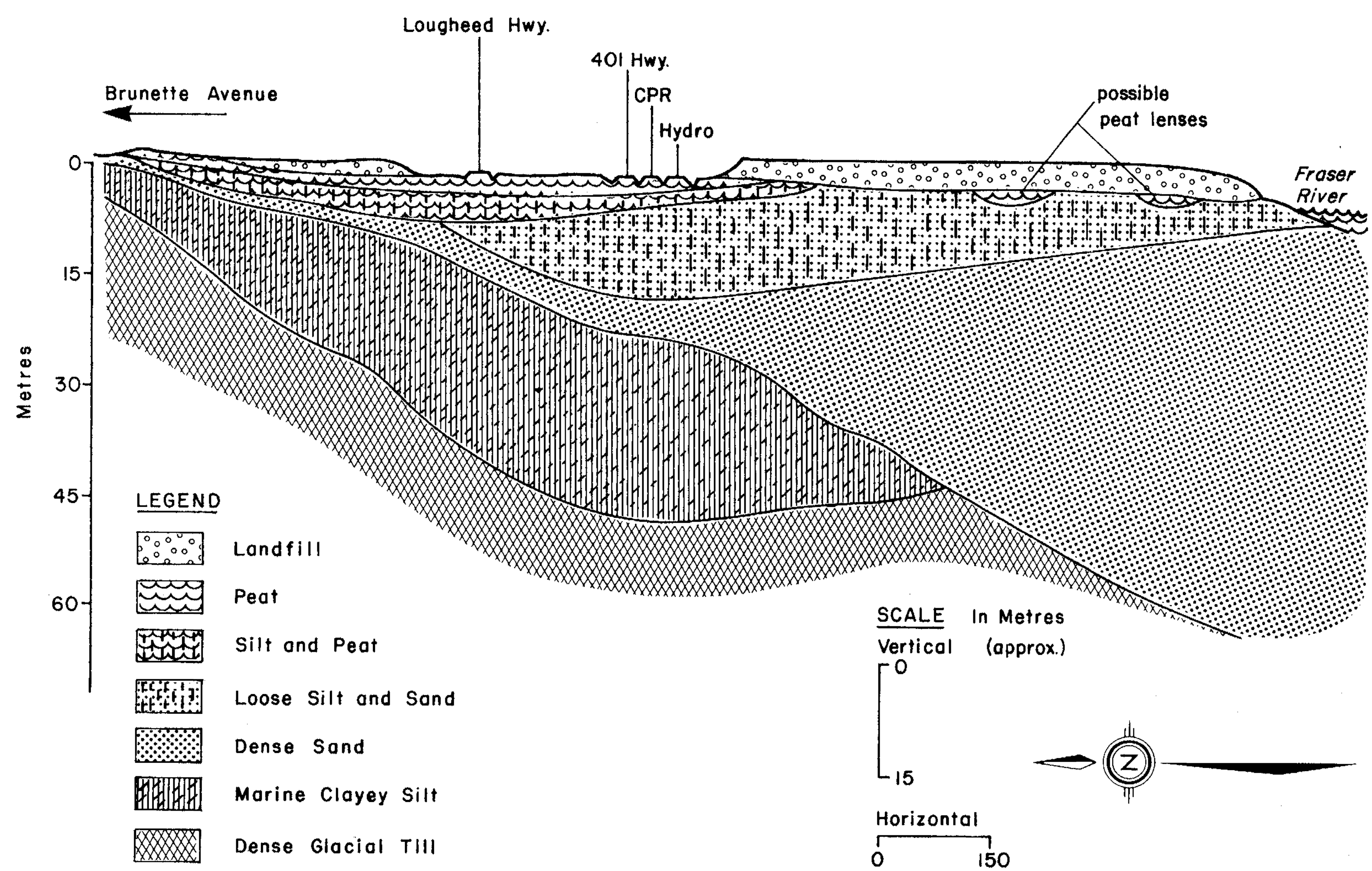


FIGURE 4.2·1 GEOLOGICAL CROSS SECTION – TERRA NOVA LANDFILL

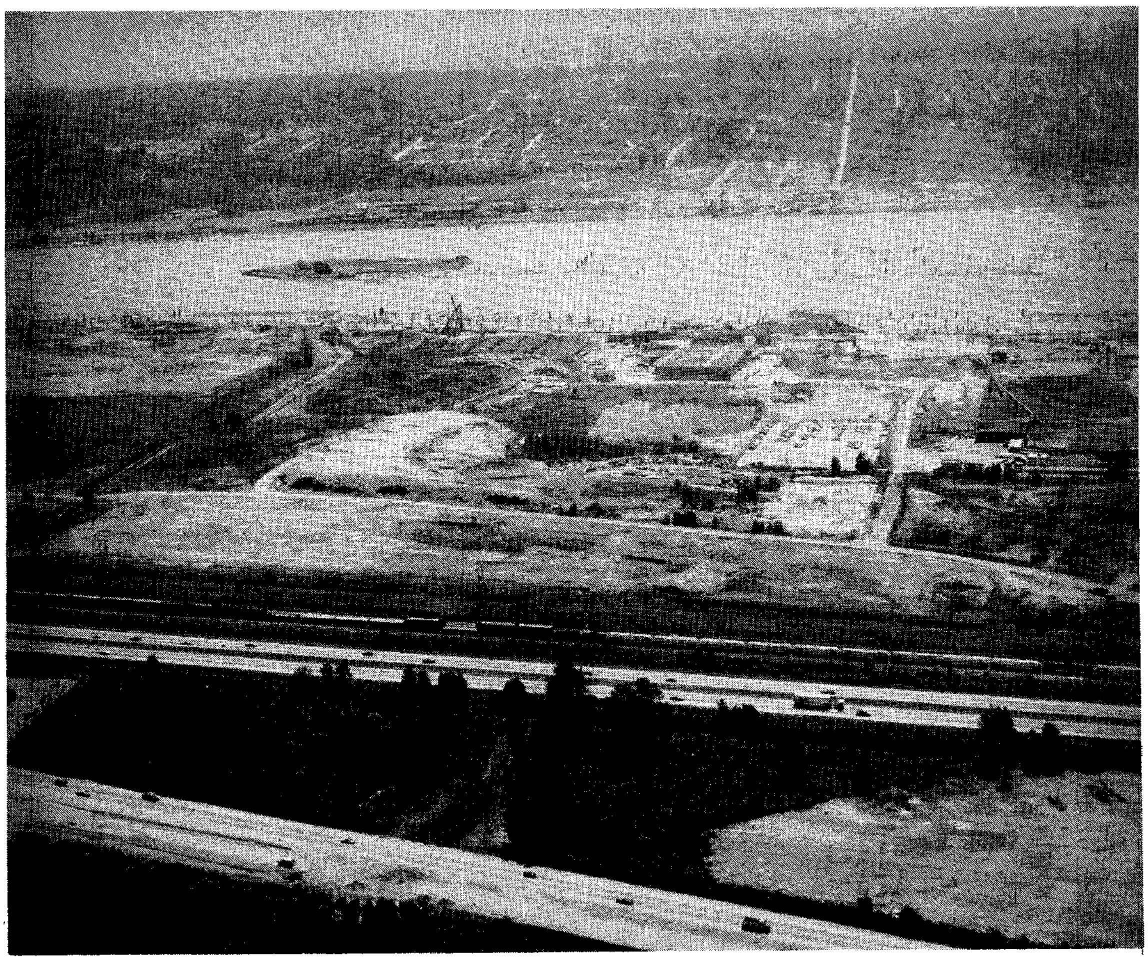


PLATE 4.2.1 GENERAL VIEW OF THE TERRA NOVA LANDFILL SITE

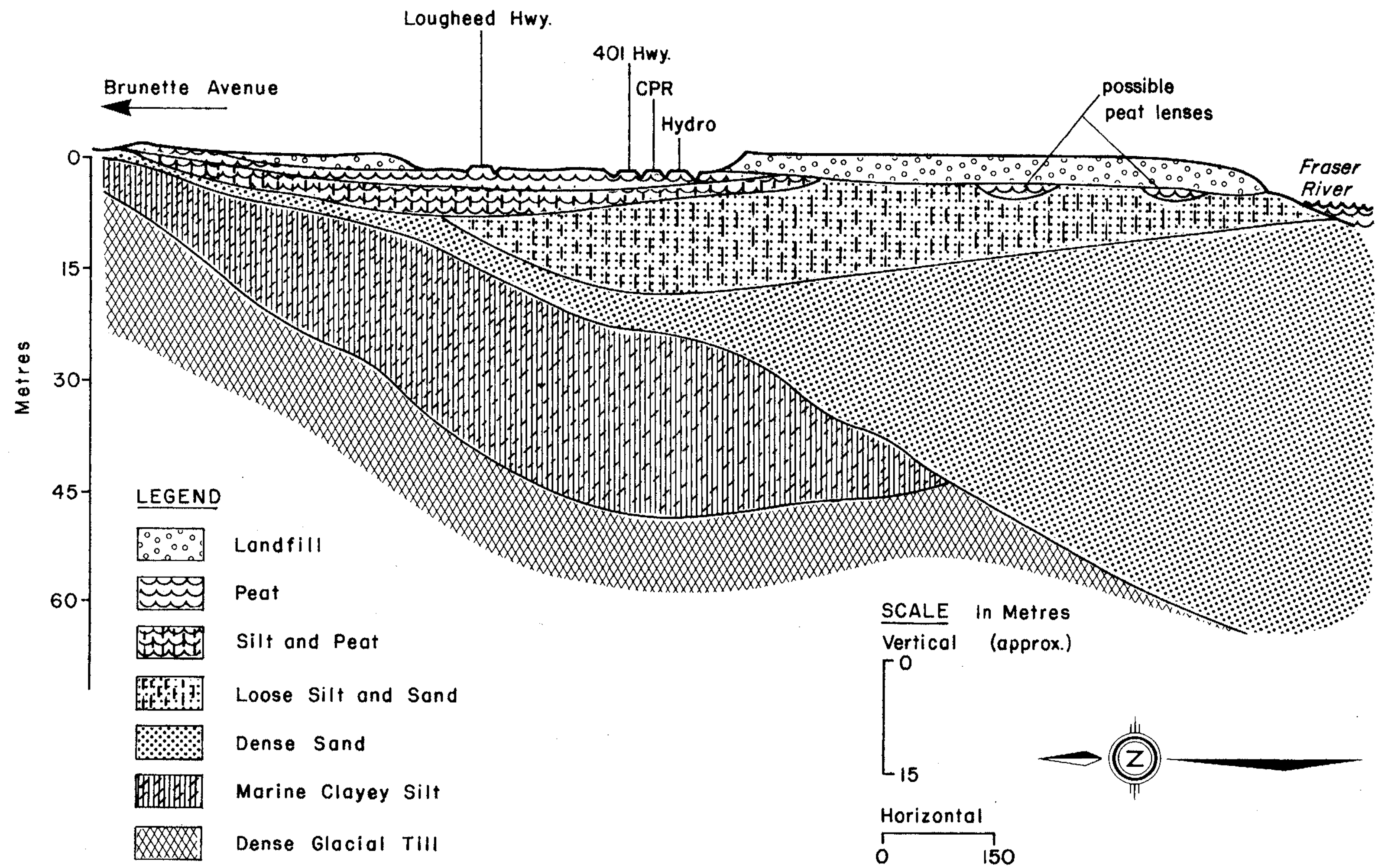


FIGURE 4.2·1 GEOLOGICAL CROSS SECTION – TERRA NOVA LANDFILL

through the fill area and is generally typical for the area lying between King Edward Avenue and Schoolhouse Street. To the east the dense sand comes closer to the surface and, to the west, the glacial till approaches the surface (Dr. Fran-Whipple, 1978, personal communication). Figure 4.2.2 shows the site plan for the Terra Nova Landfill.

From the cross-section, it can be expected that the fill was placed on peat, and peat and silt in the northern portions, and loose silts and sands and possibly some peat as the fill progressed towards the river. Water leaves the site by two obvious means: Mill Creek, which borders the west side of the fill; and through Popeye Creek, which flows more or less through the middle of the site and starts in a swampy area to the north of the fill. Both of these creeks can be reversed at their mouths by high stages on the Fraser River. Popeye Creek in places is 3.6-4.3 metres below the top of the fill. It is intended that Mill Creek will be diverted to the east along the north side of the fill into Popeye Creek. Drainage from the site can also occur directly into the Fraser River from the fill along the south boundary and also to the low created by the Trans-Mountain Gas Pipe Line and the unfilled section to south and east of the fill. Neither of the latter two are readily apparent. The drainage is detailed on the site plan, Figure 4.2.2.

Mill Creek is made up of the flow from Laurentian Creek, Schoolhouse Creek, and a swampy area north of the Freeway. Water quality in Mill Creek at the Terra Nova site is influenced to a great extent by upstream activities.

4.2.3 Operation. The filling of the Terra Nova site began in early 1965 and continued through to September 1975, covering a total area of some 60 ha. In the early years, a large percentage of the fill was hog fuel and represented 30% of the total tonnage, whereas, in later years, it was less than 10%. Table 4.2.1 summarizes the refuse placed at the Terra Nova Landfill. Refuse was initially accepted from commercial haulers, New Westminster, Coquitlam and Port Coquitlam, and starting in 1969, was also accepted from Burnaby. The hog fuel came predominantly from Crown Zellerbach. Increased tonnages between 1968 and 1969 resulted from Burnaby's shift from the Stride Avenue Landfill to the Terra Nova site.

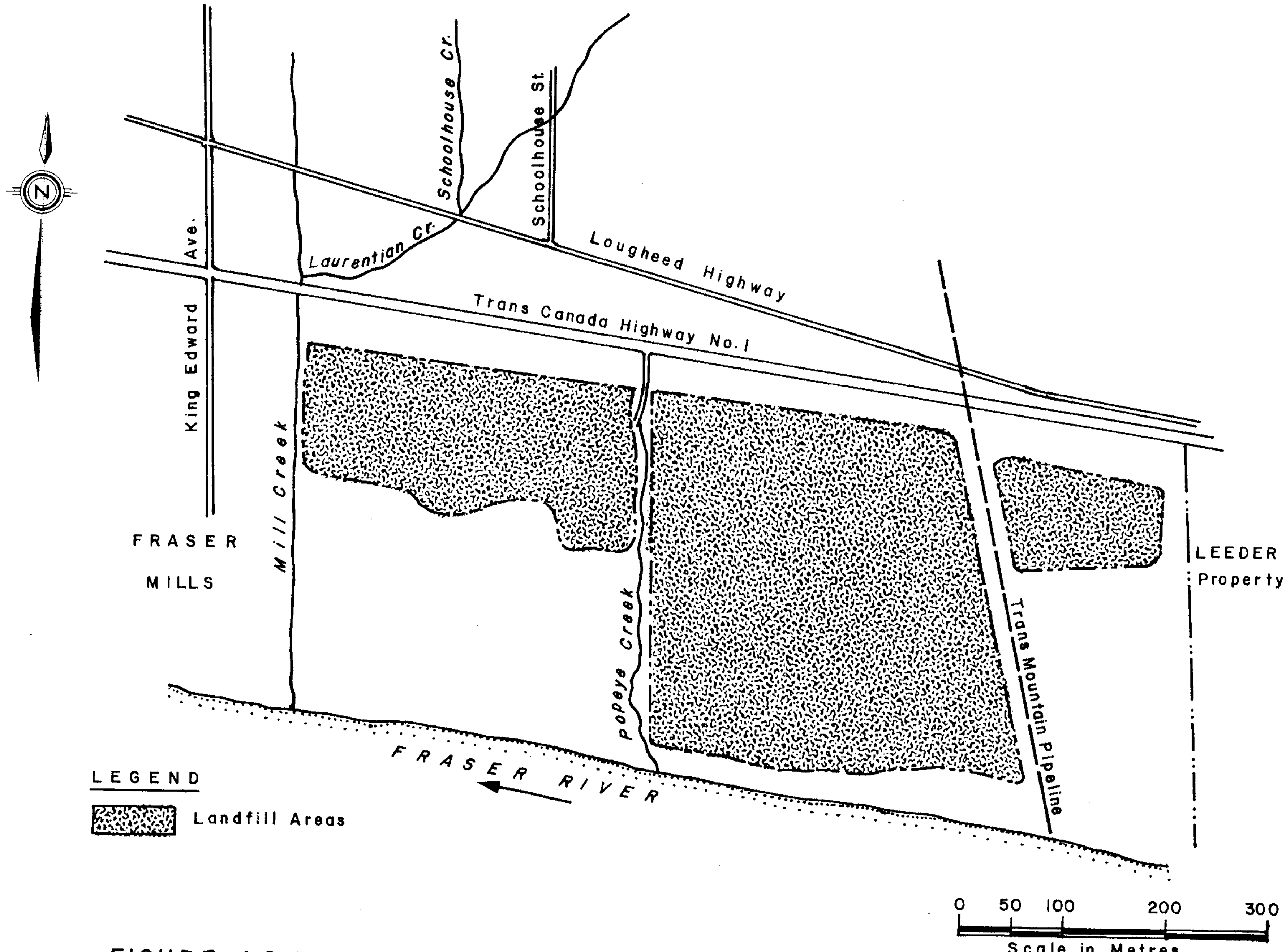


FIGURE 4.2.2 TERRA NOVA LANDFILL - SITE PLAN

TABLE 4.2.1 ANNUAL WASTE QUANTITIES - TERRA NOVA LANDFILL

Year	Refuse (tons)	Wood Waste (tons)	Total (tons)
1965	25 000	12 500	37 500
1966	40 000	20 000	60 000
1967	44 000	13 000	57 000
1968	45 000	12 000	57 000
1969	67 000	10 000	77 000
1970	84 000	8 400	92 400
1971	86 000	8 600	94 600
1972	96 000	9 600	105 600
1973	107 000	8 500	115 500
1974	110 000	8 800	118 800
1975 (Jan.-Sept.)	77 000	6 000	83 000

All fill covered with 0.45 m of soil.

The filling operation progressed generally from west to east and north to south, with one 5.8 ha site in the north-east corner being filled last. Figure 4.2.3 shows the fill positions for a number of dates. The refuse was normally placed in a 5.5 metre lift and capped overall with 0.45 metres of soil.

Although refuse filling stopped in 1975, considerable quantities of hog fuel, in places up to 6 metres high, are presently stored on parts of the landfill.

4.2.4 Leachate. Leachate continues to emanate from the Terra Nova Landfill. The decay rate of the leachate is not readily determinable, although a decrease in constituent concentration in Popeye Creek, as shown on Table 4.2.2, has occurred between 1972 and 1976/77. Analytical data on leachate and waters flowing through the site are available for 1971 (B.H. Levelton and Associates, 1971), 1972 and early 1973 (GVRD, 1973), and from 1975 through 1978 from the Pollution Control Branch (EQUIS, 1978).

Leachate can be observed to leave the site through Mill Creek and Popeye Creek and, likely, to a lesser extent, through seeps directly into the Fraser River along the south side boundary of the fill and into the groundwater. The presence of the peats and silts will tend to minimize the latter flow. Present day inputs to the site are from precipitation (average of 150 cm), plus dredge water and infiltration from the Fraser River during higher river stages.

The volume of leachate used for the calculations of loadings are of necessity somewhat open to debate, although it is felt that they are low. Two flows are used, one for Popeye Creek, the other for Mill Creek.

In 1972 and early 1973, the GVRD placed weirs in Popeye Creek and in two ditches draining into it. Presented in Figure 4.2.4 are plots of the weir readings. The flow of Weir 2 includes that passing over Weir 1. An average flow for Popeye Creek of 0.0091 m^3/sec was calculated from the areas under the curves of Weirs 2 and 3. The flow accounts for the drainage from an area of about 34 ha of refuse fill, plus a hog fuel

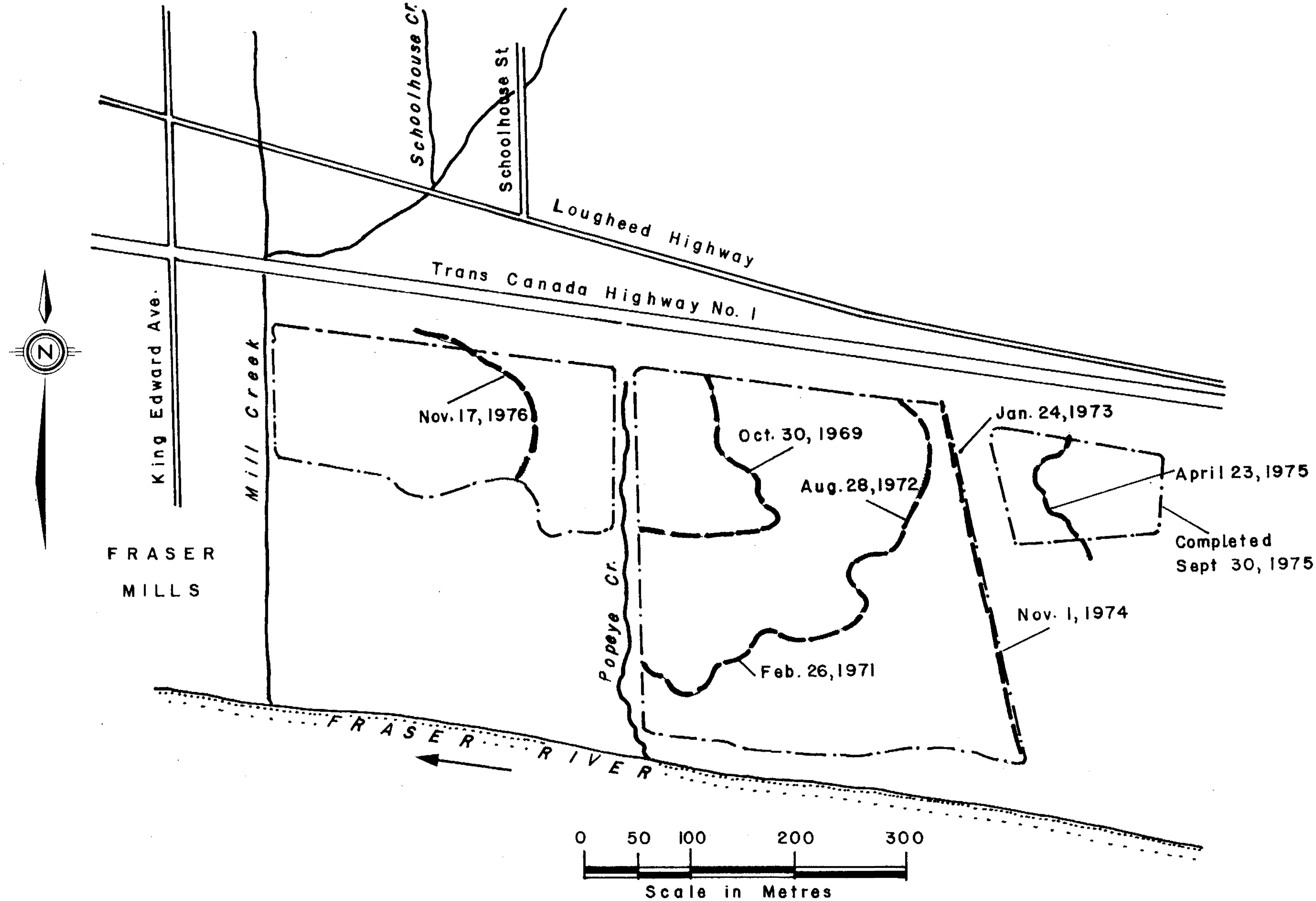


FIGURE 4.2.3 FILL POSITIONS - TERRA NOVA LANDFILL

TABLE 4.2.2 COMPARISON OF LEACHATE CONSTITUENTS - TERRA NOVA LANDFILL

Analysis	Weir #2 (Popeye Creek) GVRD 1972/73 (mean values)	Popeye Creek PCB (17/6/76)(18/1/77)
pH	6.8	7.0
Specific Conductance	1433 μ mhos/cm	334 μ mhos/cm
NH_4	30 mg/l	3.3 mg/l
COD	322 mg/l	49 mg/l
Chloride	106 mg/l	11 mg/l
Iron	80 mg/l	N/A
Manganese	2.06 mg/l	.57 mg/l
Zinc	0.36 mg/l	.025 mg/l
Fecal Coliform	2400 MPN/100 ml	90 MPN/100 ml
Total Coliform	14 600 MPN/100 ml	1100 MPN/100 ml
Chromium	below detection	below detection
Copper	below detection to - 0.12 mg/l	.004 mg/l*
Lead	below detection	.003 mg/l*

* Pb detection 1972/73 = 0.2 mg/l.
Cu detection 1972/73 = 0.01 mg/l.

N/A = not available. May 8, 1975 reading = 92.8 mg/l.

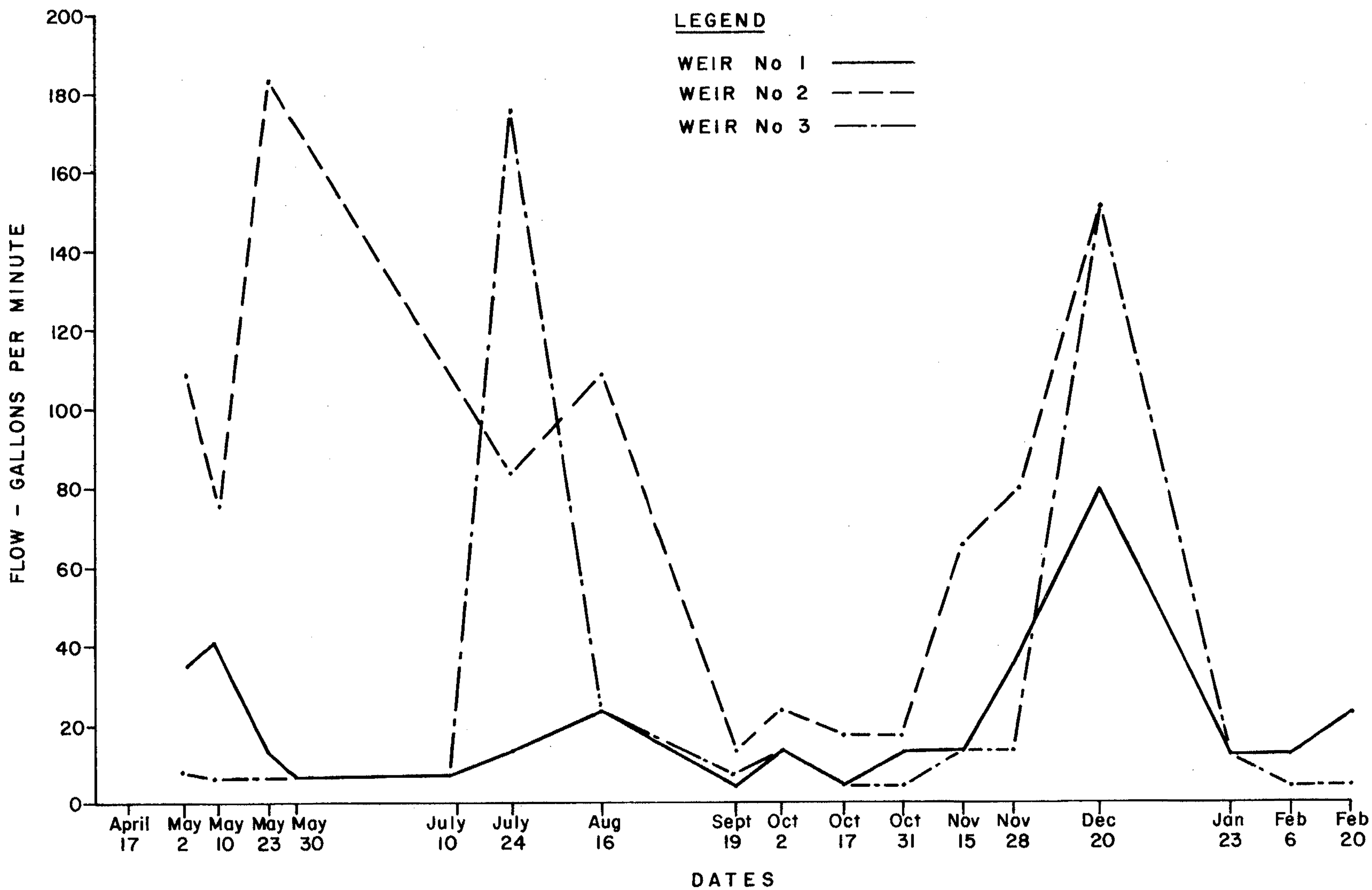


FIGURE 4.2.4 PLOTS OF WEIR FLOWS AGAINST TIME — TERRA NOVA LANDFILL

site of unknown size. A theoretical calculation of flow from precipitation assuming 30 cm of evapotranspiration yields a figure of 0.013 m^3/sec for 34 ha, or about 0.023 m^3/sec for the full site.

A field observation of the flow in Popeye Creek on January 31, 1979 after subtracting the upstream contribution, gave an estimate of 0.014 m^3/sec (EPS, 1979). That figure would include some snow melt and possibly some residual dredge water flow. Leachate resulting from expansion of the fill to the east subsequent to the GVRD readings, probably finds its way to the river through the underlying materials or by diffuse overland flow. For the purpose of loading calculations, the flow in Popeye Creek is taken to be an average of 0.009 m^3/sec. Table 4.2.3 shows the calculated leachate constituent loads to the Fraser River coming out through Popeye Creek. The constituent concentrations used are an average of two sets - June 1976 and January 1977. While it could be argued that there is a bias in using values two years old, it should be noted that the concentrations in the 1977 set were higher in almost every case than those in 1976.

Mill Creek, the second principal avenue of leachate loss from the site, skirts the west boundary of the oldest part of the fill. Flow measurements for Mill Creek are very skimpy, although one set of measurements (Levelton, 1971) indicated what was felt to be high and low flow figures of 0.513 m^3/sec and 0.088 m^3/sec, respectively, on the north side of the fill. These measurements were taken on April 8 and 30, 1971. The same source (Levelton, 1971) also reported an increase in the Mill Creek flow from a high of 0.513 m^3/sec to 0.88 m^3/sec in its passage to the south end of the fill. Levelton (1971) cautioned the use of this figure as it appeared to be too high. A field observation of the flow on January 31, 1979 gave an estimate as the creek entered the site, of 0.13 m^3/sec, and at the south side of the fill of 0.19 m^3/sec, (EPS, 1979). The increased flow cannot be attributed solely to the landfill, as drainage does enter from the mill site to the west of the creek. The theoretical average flow based on a fill area of 10.9 ha, 150 cm+ of precipitation and 30 cm of evapotranspiration, is 0.004 m^3/sec, which

TABLE 4.2.3 CONSTITUENT LOADINGS TO THE FRASER RIVER FROM POPEYE CREEK - TERRA NOVA LANDFILL

Constituent	Concentration*	Daily Load (kg/day)	Annual Load (kg/year)
pH (units)	7		
Chloride	11.3	8.9	3230
NH_4	3.25	2.5	930
NO_3	0.15	0.12	43
Organic N	1.25	1	360
Total Phosphate	0.11	0.09	31
COD	49	38.0	14 000
Tannin & Lignin	20	16	5720
Iron	NA	NA	NA
Spec. Conductance (mho/cm)	334		
Manganese	0.57	0.45	160
Copper	.0045	0.003	1.3
Lead	.003	0.002	0.9
Zinc	.025	0.02	7.15
Fecal Coliform	900 MPN/100 ml	70×10^9 MPN	25×10^{11}
Total Coliform	1100 MPN/100 ml	86×10^9 MPN	31.4×10^{11}
Arsenic	.0045	0.003	1.3
Flow	120 gpm		

*All mg/l, except as noted.

NA - not available

is considerably below inferred flows. The flow from the landfill to Mill Creek is, in almost all instances, a diffuse flow spread over the length of the fill.

Four sets of analyzed data are available between July 1977 and November 1978 for Mill Creek upstream (PCB site 18) and downstream (PCB site 9) of the site. These values are presented for comparison in Table 4.2.4. No general trends are discernible, some constituent concentrations to increase across the site, while some decrease and others are higher in one set and lower in the next.

In considering the concentrations of constituents of Mill Creek upstream of the Terra Nova site, it should be noted that tributaries to Mill Creek flow through active wood waste landfill sites. In comparing the concentrations of constituents in the streams that make up Mill Creek upstream of the active filling (PCB sample points 14 and 15) with those in Mill Creek immediately upstream of the Terra Nova site (Table 4.2.5), an increase of at least six to sevenfold can be seen for a number of constituents including ammonia, iron, zinc, manganese and phosphorus.

In calculating the loadings coming from the Terra Nova site into Mill Creek, the theoretical flow of 0.004 m^3/sec is assumed at the average constituent concentration present in Mill Creek for the two sets of 1978 data. In addition, metal loadings and ammonia loadings to the Fraser River are calculated for Mill Creek using an average flow of 0.137 m^3/sec, which takes into account the loads from the active sites and possibly some road wash. These loadings are presented in Table 4.2.6 along with a total from the Terra Nova Landfill, i.e., Popeye Creek, plus the 0.004 m^3/sec flow into Mill Creek.

4.2.5 Impact on Water Quality. No studies have been undertaken to ascertain whether or not there is an alteration of the Fraser River from the Mill Creek and Popeye Creek flows. The discharges at certain river stages can be seen as dark plumes hugging the shore. Historical records on Popeye and Mill creeks were not available, but it is reasonable to assume that the water quality has deteriorated to some extent as a result of leachate addition.

TABLE 4.2.4 COMPARISON OF CONSTITUENT CONCENTRATIONS IN MILL CREEK UPSTREAM (Site 18) AND DOWNSTREAM (Site 9) - TERRA NOVA LANDFILL

Constituent	July 28, 1977		October 24, 1977		January 30, 1978		November 21, 1978	
	#18	#9	#18	#9	#18	#9	#18	#9
pH (units)	7.1	6.9	6.9	6.9	7.4	7.4	7.1	7.3
Chloride	17.6	25.9	16.1	15.8	27.3	58.8	21.8	21.2
NH_3	.324	1.24	.348	.378	.336	1.48	.476	.595
NO_3	.28	.02	.26	.25	.45	.46	.59	.56
Organic N	.5	2.52	.43	.48	.48	.76	1.0	1.0
Total Phosphate	.055	.185	.102	.112	.075	.262	.086	.092
Spec. Cond. (μmhos/cm)	240	551	223	216	237	355	276	275
COD	16.1	124	31.5	39.4	27.1	46.5	40	32
Iron	2.7*	3.59*	3.3	3.8	2.3	NA	3.5	4.8
Manganese	NA	2.37	.38	.39	.39	.38	.43	.41
Chromium	< .005	< .005	< .005	< .005	< .005	< .005	< .005	< .005
Arsenic	.006	.006	< .005	< .005	.007	.009	.008	< .005
Copper	.003	.002	.007	.005	.003	.007	.003	.002
Lead	.005	.002	.007	< .001	.003	.004	.003	.007
Zinc	.19	< .005	.04	.04	.03	.04	.07	.08
Tannin & Lignin	2.0	1.5	4.0	2.1	7.5	3.6	5.5	5.0

All values as mg/l except for pH and specific conductance.
N/A = not available.

* Reported as 27.0 and 35.9; decimal shifted by J.W. Atwater

TABLE 4.2.5 COMPARISON OF CONCENTRATIONS OF CONSTITUENTS IN MILL CREEK UPSTREAM OF TERRA NOVA LANDFILL WITH THOSE IN TRIBUTARIES

Constituent	#14 Upstream from Schoolhouse Creek	#15 Upstream from Mill Creek	#18 Mill Creek upstream from Terra Nova downstream #14, 15
pH	7.6	7.1	7.1
Chloride	12	26.3	21.8
NH	.02	.02	.476
NO	1.04	.68	.59
Organic N	.12	.19	1.0
Total Phosphate	.012	.008	.086
Spec. Cond. (mho/cm)	170	214	276
COD	< 10	52	40
Iron	.6	.5	3.5
Manganese	.06	.07	.43
Chromium	< .005	< .005	< .005
Arsenic	< .005	< .005	.008
Copper	< .001	< .001	.003
Lead	< .001	< .001	.003
Zinc	< .005	.014	.07
Tannin & Lignin	.2	.4	5.5

All values are in mg/l except for pH and specific conductance.

TABLE 4.2.6 CONSTITUENT LOADINGS FROM TERRA NOVA LANDFILL AND MILL CREEK

Constituent	Concentration Mill Creek*	Daily Loading to Mill Creek from Terra Nova (kg/day)	Daily Loading Popeye Creek (kg/day)	Daily Loading Mill Creek to Fraser River (kg/day)	Daily Load to Fraser River from Terra Nova #1 & 2 (kg/day)
pH (units)	7.4	-	-	-	-
Chloride	40	14.4	8.9	470	23.3
NH_3	1.04	0.4	2.5	12.2	2.9
NO_3	.51	0.18	0.12	6.0	0.30
Organic N	.88	0.3	1.0	10.3	1.3
Total Phosphate	.177	0.06	0.09	2.1	0.15
Spec. Cond. (μmhos/cm)	315	-	-	-	-
COD	39	14	38.0	460	52
Iron	3.5	1.3	-	41	$<$ 1.3
Manganese	.40	0.14	0.45	4.7	0.60
Chromium	$<$.005	-	-	-	-
Arsenic	.005	.001	0.003	0.06	0.004
Copper	.0045	.001	0.003	0.05	0.004
Lead	.0055	.002	0.002	0.06	0.004
Zinc	.06	0.02	0.2	0.71	0.08
Fecal Coliforms	N/A	-	$70x10^7$ per 100 ml	-	$>$ $70x10^7$ per 100 ml
Total Coliforms	N/A	-	$86x10^8$ per 100 ml	-	$>$ $86x10^8$ per 100 ml

*mg/l except as noted.

4.2.6 Intended Use. The area all around the site is industrial or is currently being filled for industrial use. It can be anticipated that the Terra Nova site will have light industrial use.

4.3 Stride Avenue Landfill

4.3.1 General. The Stride Avenue Landfill located on Burnaby's south slope has been used as a refuse dump since 1910 (as far back as people can recollect). The site which has been closed since late 1969 now remains vacant. Plate 4.3.1 is an aerial photograph of the site.

4.3.2 Physical Description. The Stride Avenue Landfill is located on the south slope above Marine Drive, just to the west of the Burnaby/New Westminster boundary (Figure 4.3.1). The site is separated from the North Arm of the Fraser River by the Burnaby Flats and is located at a distance of some 1200 metres by the shortest route from the river.

The original site was a large gully more than half of which has now been filled. The existing contours and fill outline are shown in Figure 4.3.2.

The site is generally underlain by post glacial sands with some gravel. Along the gully floor there is some evidence of silt in the sandy gravel. The depth of the sands is not known, particularly at the centre of the gully, but has been reported to extend to at least 15 metres along the flanks (Burnaby Staff, 1978, personal communication). Sands were encountered during excavation of a gravity sewer and they extended over the entire length from the site to the Marine Drive.

A heavy, blue, soft clay strip some 4.6 - 6 metres wide and of unknown depth extends parallel to the gully along the east side of the fill. The clay was avoided during the fill operation but is known to have a lateral extent of some 60 metres. Adjacent to the fill, groundwater was encountered at a depth of about 15 metres. There is very little surface water apparent anywhere on the site, although runoff from above the site is flumed alongside the fill and is discharged into the gully at the toe of the fill. Seepage coming from the toe of the fill is evident in the gully.

PLATE 4.3.1 GENERAL VIEW OF THE STRIDE AVENUE LANDFILL SITE

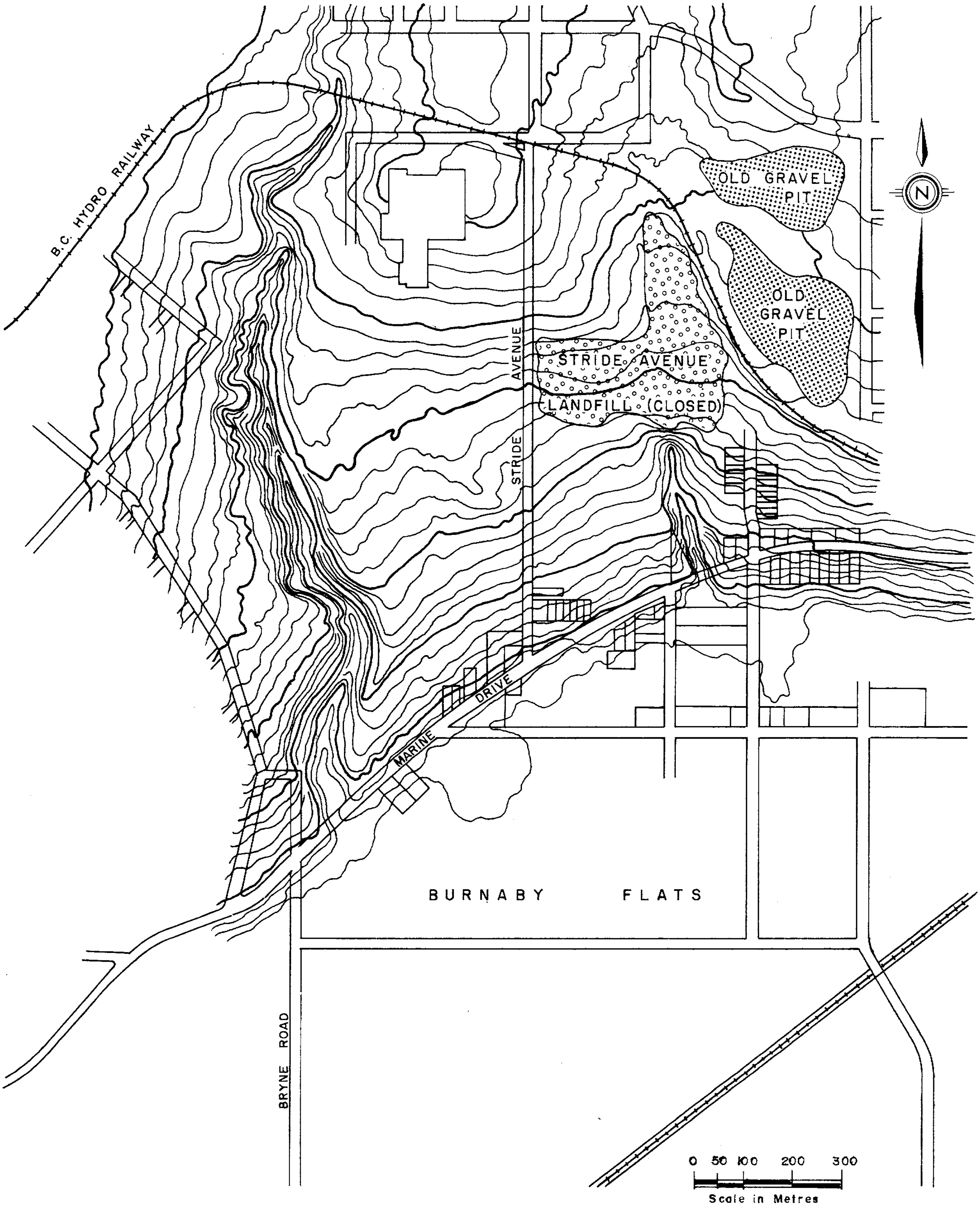


FIGURE 4.3.1 LOCATION MAP - STRIDE AVENUE LANDFILL

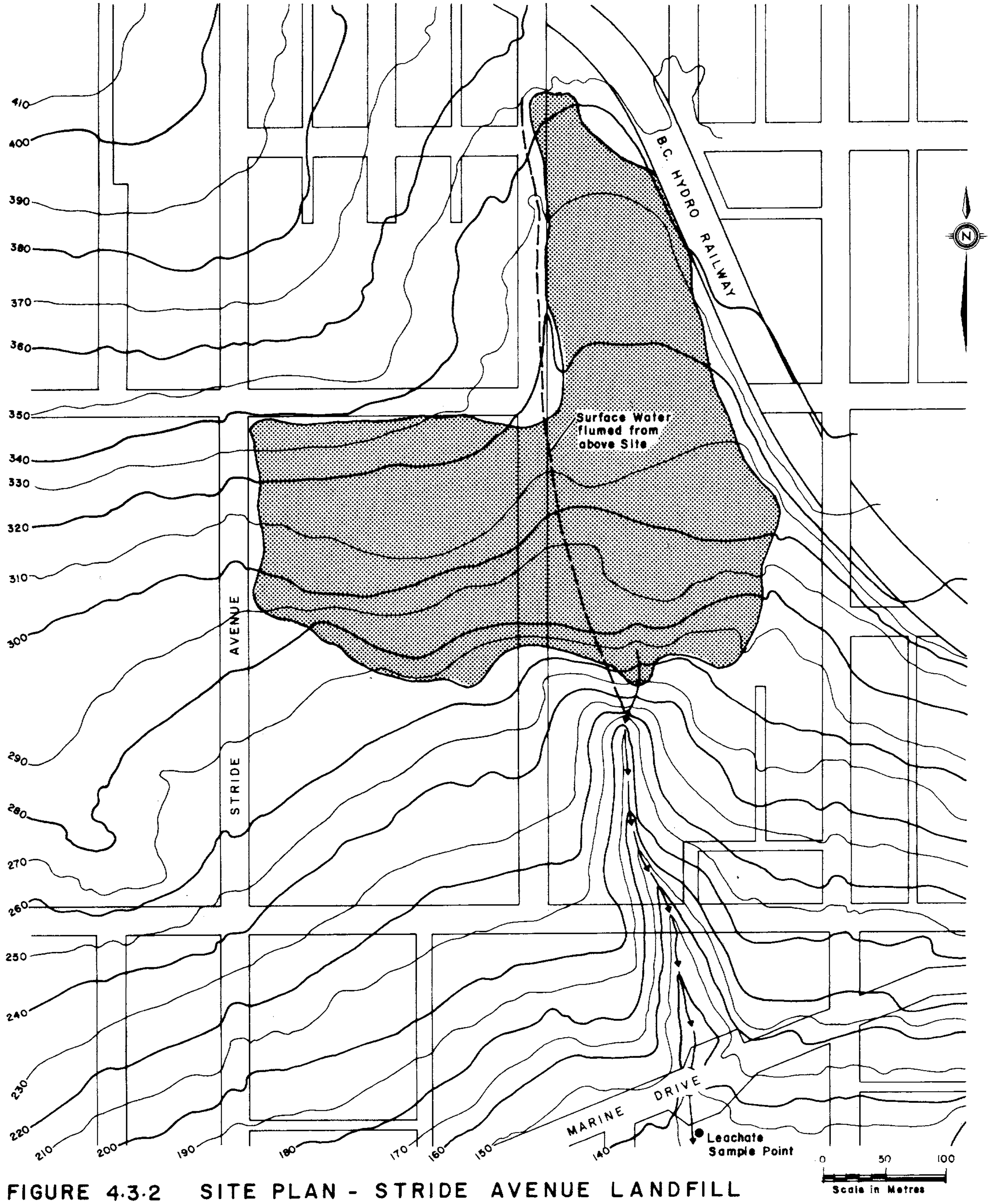


FIGURE 4.3.2 SITE PLAN - STRIDE AVENUE LANDFILL

4.3.3 Operation. The Stride Avenue Landfill has been used for refuse disposal since at least 1910, but it was not until about 1964 that all of Burnaby's refuse came to the landfill site.

Prior to 1964, the refuse from North Burnaby went first to the old Barnett Highway site then from about 1956 to 1964 to the Sperling Avenue site. The filling operation ceased in 1969 with Burnaby's refuse going first to the Terra Nova Landfill and now to the Braid Street (Coquitlam) Landfill. A sand and gravel pit immediately to the east of the refuse site is presently referred to as the Stride Avenue Landfill and is utilized only for excavated material and garden cuttings.

The filling operation proceeded from the north end of the site adjacent to the B.C. Hydro railway tracks, Figure 4.3.2, to the south. The operation consisted of a straight gully fill with some filling of 6-9 metres deep sand excavations on the western flanks. The fill depth is believed to average 12-14 metres and is up to 27 metres in depth. Fire breaks were placed about every 60 metres. The total fill covers an area of some 8.08 ha.

Most types of material were accepted at the site although demolition material was generally directed elsewhere, usually to the Kerr Avenue or Terra Nova landfills. Debris from municipal cleaning operations was usually segregated and burnt.

A crude estimate of fill volume may be made by taking the estimated average depth x the area of the fill; therefore, 12.2 m of depth x 8.08 ha of area x 10 000 m^2/ha = 987 000 m^3. Based on J.J. Kaller's values (Kaller, 1970), of in-place density of 534 kg/m^3 (900 lbs/yd^3) and including the weight of cover, the weight of placed fill is estimated to be 527 000 tonnes. This figure is not out of line with the average reported tonnage of 36 050 tonnes for the years 1967 through 1969 and the probable tonnages placed over the long life of this fill.

4.3.4 Leachate. Leachate contamination of the gully water remains apparent some 10 years after the closing of the Stride Avenue Landfill. The rate of decrease of the leachate constituent concentration is unknown although a comparison can be made with GVRD data from 1972/73 and EPS data collected early in 1979. These data are presented in Table 4.3.1. With the exception of NH4, the concentrations of all the constituents reported in 1979 are within the range of the concentrations found by the GVRD in 1972/73. Comparing mean values of constituents which can be indicative of leachate, Cl-, COD, NH4, iron, zinc, and specific conductance for the two periods, would suggest that the leachate strength has decreased with time. However, it should be recognized that at the time of the EPS sampling a considerable quantity of surface water was entering the gully. For the loading calculations the average of the 1979 EPS concentrations were used with GVRD measured flows. GVRD measured the gully flow between May 30, 1972 and February 20, 1973. These flow measurements were plotted against time and are shown in Figure 4.3.3. Averaging the flow under the curve, yields a flow of 0.004 m^3/sec. Site conditions have not changed appreciably and it is assumed that flow conditions have not changed. During early 1979, flows were estimated to be between 0.002 m^3/sec and 0.015 m^3/sec (EPS, 1979).

Presented in Table 4.3.2 are the daily and annual loadings for the leachate emanating from the Stride Avenue Landfill at Marine Drive.

The theoretical flow of leachate from the 8.08 ha site using 127 cm of precipitation, a runoff coefficient of 0.15 and evaporation of 16 cm is 0.002 m^3/sec, which is half the average measured flow. However, the catchment area of the gully is considerably in excess of the fill area, and water is bypassed to gully. Offsetting these factors is the probable loss of some of the precipitation directly to groundwater without the latter's appearance in the gulley.

4.3.5 Impact on Water Quality. The leachate discharge from the Stride Avenue Landfill winds its way past a number of truck farms on the Burnaby Flats in making its way through the drainage system out to the Fraser River at Byrne Road. It is understood that the drainage water is used for irrigation and, in that regard, the concentrations of various

TABLE 4.3.1 CONSTITUENT CONCENTRATIONS, 1972/73 AND 1979 - STRIDE AVENUE LANDFILL

Parameter (mg/l)*	E.P.S. (1979) No. of Samples	Maximum	Minimum	Average	GVRD (1972/73) No. of Samples	Maximum	Minimum	Average
pH (units)	6	8.1	7.6	7.8	20	7.9	6.6	7.5
T. Alk. $CaCO_3$	6	437	110	323	18	1012	119	545
Sulphate	6	13.7	7.6	9.8	8	12	< 2	8.0
Chloride	6	53	25.5	43	18	170	11.5	100.3
Total Phosphate	6	0.325	0.02	0.09	10	.13	< .01	8/10 b/d
Nitrate	6	1.91	0.144	1.55	N/A			
Ammonia	6	21.5	4.35	14.8	13	57	19	42
Spec. Cond. (μmhos/cm)	6	1010	337	810	20	2400	455	1312
NFR	6	210	< 5	40	18	639	9	104
COD	6	65	40	48	15	665	275	52
Aluminum	5	.12	< .09	.08		< .1	< .1	< .1
Arsenic	6	< .15	< .15	< .15	N/A			
Barium	6	2.74	.145	0.2	N/A			
Calcium	6	93.9	32.5	75	18	110	10.2	83
Cadmium	6	< .01	< .01	< .01	18	.04	< .01	14/18 b/d
Cobalt	6	< .015	< .015	< .015	N/A			
Chromium	6	< .015	< .015	< .015	18	.15	< .04	15/18 b/d
Copper	6	.037	< .01	.01	18	.31	< .01	7/18 b/d
Iron	6	12.9	1.53	4.5	18	105	0.7	39.5
Mercury	6	< .1	< .1	< .1	N/A			
Magnesium	6	27.6	9.37	22.2	18	35	7.5	26
Manganese	6	2.06	.608	1.60	14	1.70	.27	1.05
Molybdenum	6	< .15	< .15	< .15	N/A			
Sodium	6	47	17.7	37.7	18	115	15	82
Nickel	6	.18	< .08	5/6 b/d	18	.3	< 0.1	14/18 b/d
Lead	6	< .09	< .09	< .09	18	.25	< .02	13/18 b/d
Antimony	6	< .08	< .08	< .08	N/A			
Selenium	6	< .15	< .15	< .15	N/A			
Tin	6	< .2	< .2	< .2	N/A			
Strontium	6	.603	.246	.49	N/A			
Titanium	6	< .009	< .009	< .009	N/A			
Vanadium	6	< .09	< .09	< .09	N/A			
Zinc	6	.3	< .02	.08	18	.64	.04	.23
Silicon	5	11.6	8.2	10.6	N/A			

*mg/l except as noted
N/A = not analyzed
b/d = below detection limit

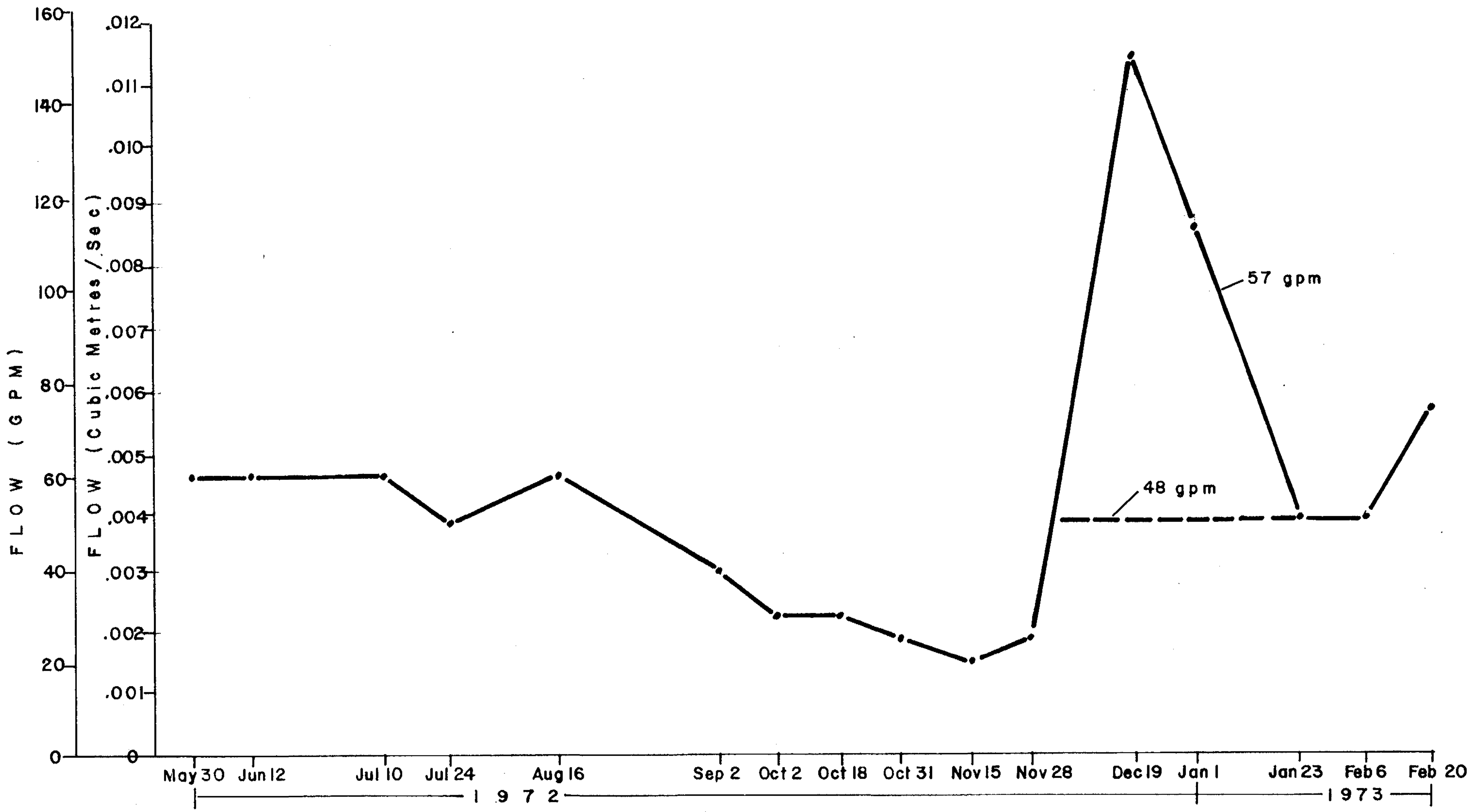


FIGURE 4.3.3 PLOT OF WEIR FLOW AGAINST TIME - STRIDE AVENUE LANDFILL -

TABLE 4.3.2 DAILY AND ANNUAL LEACHATE LOADINGS - STRIDE AVENUE LANDFILL (0.004 m^3/sec)

Parameter	Concentration (mg/l)	Daily Load (kg/day)	Annual Load (kg/day)
COD	48	17.5	6400
Ammonia	14.8	5.4	1975
Chloride	43	15.7	5740
Sulphate	9.8	3.6	1310
Total Phosphate	0.09	0.03	12
NFR	40	14.6	5340
TAlk. as $CaCO_3$	323	118	43 100
Nitrate	1.55	0.57	207
Aluminum	0.08	0.03	11
Barium	0.2	0.07	27
Calcium	75	27.4	10 000
Copper	0.01	.004	1.3
Iron	4.5	1.65	600
Magnesium	22.2	8.1	2960
Manganese	1.60	0.59	213
Sodium	37.7	13.8	5030
Strontium	0.49	0.18	65
Zinc	0.08	0.03	11
Silicon	10.6	3.88	1415

ions in that water should be taken into consideration if long-term irrigation usage is intended. For instance, the recommended maximum ...ntration of manganese in irrigation water for continued use is 0.20

red in the drainage

s example it should be

ated, depending on soil

s. Further study is

this leachate stream.

cant, however, it is

s currently considering

Note to word processor operator.

- Shadowing for headings

- letters run together

- Space in > 3 digit numbers too large

Major Changes

1) Leslie - technique comments & Tables

2) Guidelines & Recommended values

3) Iona - distances.

higher than; lower than

sentences ending in "there"

Ref Clark Drinnan & Wallar

Headings: titles & numbering

Order of Metals in Tables - Alphabetical

5 WOOD WASTE LANDFILL

5.1 Introduction

A report prepared for the British Columbia Wood Waste Energy Coordinating Committee estimated that in 1977, 2 721 000 m^3 (480 500 Gravity Pack Units - one GPU = 200 ft^3) of surplus hog, chips, and mill pieces required disposal in the Lower Fraser Valley (Appleby, 1978). Of this material, 264 000 m^3 (46 700 GPU) was incinerated and 2 458 000 m^3 (433 800 GPU) or about 427 000 tonnes was landfilled (Appleby, 1978).

An aerial survey during September 1977 revealed 35 wood waste sites covering some 130 ha in the study area (EPS, 1977). The 1977 survey would have missed an unknown number of fills built on or covered with sand or soil. A similar aerial survey undertaken today would miss a number of fills covered since 1977 and would reveal new ones that have since started.

The use of wood waste as fill for reclaiming or raising land is ubiquitous throughout the Lower Fraser Valley. Wood waste from a structural viewpoint is a desirable fill material for boggy land. At the same time, there are large volumes of wood waste that require disposal. In both these aspects, the landfilling of wood waste results in the production of leachate.

5.2 Wood Waste Disposal

The Reid Collins Report (Appleby, 1978) indicated that for 480 500 GPU (2 721 000 m^3) of wood waste disposed of in the study area in 1977, 2 458 000 m^3 were landfilled. It is likely that comparable volumes have been landfilled annually for the past number of years. Wood waste volumes are dependent upon both wood production and wood scrap utilization, i.e., sawmill production and utilization in the pulp and paper industry. It can be anticipated that as the presently committed hog fuel boiler expansion comes on-line, about 20% of the wood waste will be diverted to these boilers. There is also a suggestion that sufficient wood waste is available in the study area to support a power boiler in the Lower Mainland (Bob Evans, B.C. Energy Commission, pers. comm.). The present species mix of the wood waste is cedar 58%, hemlock, balsam, fir 40%, and cottonwood 2%. With the completion of presently planned expansion of hog fuel burners, the mix is expected to be cedar 75%, hemlock, balsam and fir 23%, and cottonwood 2%.

5.3 Wood Waste Disposal Sites

The 1977 aerial survey identified 35 wood waste sites, 89% of which were on the foreshore of the Fraser River. Over 90% of the total were active. From the aerial photographs it was estimated that the sites covered some 130 ha and contained an estimated 4 350 000 m^3 of wood waste. This is less than one half of what has probably been landfilled in the last five years, but is reasonable considering the usually short life of many fills. Details of the aerial survey, site locations and site volumes are presented in Appendix D.

5.4 Wood Waste Leachates

The characteristics of wood waste leachate were discussed at some length in the introduction to this report and are reviewed further in this section where field values for a number of wood waste sites are presented. From a review of the available field data it would appear to be impossible to calculate leachate loadings as there is not one item of loading information available for any site. Notwithstanding that, two estimates using two different sets of parameters and assumptions are made which, hopefully, bracket the loadings. Data from two lysimeter studies of wood waste, one by Dr. R.D. Cameron (Cameron, 1975a) predominantly using hemlock, and the other by Econotech (Thomas, 1977) using western red cedar, can be used to arrive at a percentage value for leachate solids.

Cameron found that about 1.9% of the dry wood weight leached after four years, or about 2.9% measured as COD using 229 cm of infiltration. From Thomas' 1977 data, after one year (at 254 cm of infiltration), 1.23% had leached from the red cedar, which was comparable to 1.16% for hemlock after one year using Cameron's data. At the same time, Cameron found that only 0.5% total solids leached after four years under the influence of 38 cm of infiltration. At the net infiltration rate found in the study area, 95 cm (75% of average rainfall), it could be anticipated that the percent solids leachable from the fill sites would lie between the two values. It should be noted that many of the sites are inundated while the lysimeters were free draining. For the purpose of the loading calculations, it is assumed that the percent leached after four years is 1% for total solids and 1.5% as COD for both hemlock and cedar.

The second assumption is that roughly equal quantities of wood waste have been landfilled in the last four years, therefore, the percent leachable x annual quantity landfilled will yield the annual loading.

The third assumption is that the characteristics of cottonwood can be considered comparable to cedar, and fir and balsam comparable to hemlock.

Using data from the Reid Collins report (Appleby, 1978), 433 800 GPU (427 000 tonnes) was landfilled of which 60% was cedar and cottonwood, and 40% was hemlock, balsam and fir.

From conversion tables there are 71 to 73 ft^3 of solid wood per unit of hog, and the oven dry weights of red cedar and hemlock are 23 lb/ft^3 and 29 lb/ft^3, respectively (IFP, 1957).

Therefore the total solids leached:

$$= [433\ 800\ (0.6 \times 72\ ft^3/unit \times 23\ lb/ft^3) + (0.4 \times 72\ ft^3/unit \times 29\ lb/ft^3)] \times 0.01$$

$$= 4\ 338\ (993.6 + 835.2)$$

$$= 7\ 933\ 334\ lb/yr$$

$$= 3\ 597\ 884\ kg/yr$$

$$9\ 857\ kg/day$$

and the COD leached:

$$= 1.5 \times 9\ 857\ kg/day$$

$$= 14\ 786\ kg/day$$

From the calculations it is estimated that an average of 9 857 kg of total solids and 14 786 kg as COD equivalent of the total solids enter the Lower Fraser and tributaries each day.

The second estimate was developed from the multiplication of average total solids and COD concentrations for 12 wood waste sites of varying ages (EPS, 1979; Thomas, 1977) with an assumed leachate volume. COD was available for five sites and total solids for seven sites. The values were transposed, COD to total solids and vice versa at the previously used ratio of 1.5 to 1. It was assumed that the total area covered with wood waste over the last five years was equal to twice that presently observed, i.e., 260 ha, and that the average net infiltration of 95 cm was 75% of the average precipitation for the study area.

The average COD value is 740 mg/l, while the values averaged varied from 120 mg/l to 2370 mg/l. Maximum and minimum values of 7600 mg/l and 90 mg/l were rejected. The average total solids value is 490 mg/l with values, when averaged, varying from 80 mg/l to 1580 mg/l. Maximum and minimum values rejected were 5075 mg/l and 60 mg/l, respectively. Estimated COD loadings were:

$$0.740 \text{ gm/l} \times 260 \text{ ha} \times 0.95 \text{ m} \times 10\,000 \text{ m}^2\text{/ha}$$
$$\times 1000 \text{ l/m}^3 \times 1 \text{ kg/l}$$
$$= 1\,827\,800 \text{ kg/yr}$$
$$\text{or } 5\,000 \text{ kg/day}$$

Estimated total solids loadings:

$$5\,000 \text{ kg/day} \times 1/1.5 = 3\,340 \text{ kg/day}$$

Similar estimates can be made for a number of other parameters, specifically total nitrogen and phosphorus, copper, iron and manganese. Table 5.1 summarizes the loading estimates.

A number of points that will affect the absolute loadings but were not considered in the estimates, include:

1) A quantity of wood waste finds its way onto the refuse landfills for road building and intermediate cover.

TABLE 5.1 SUMMARY OF CONSTITUENT DAILY LOADINGS FROM WOOD WASTE LEACHATES

Parameter	Estimate #1			Estimate #2				
	% Leachable	Weight (tonnes)	Loadings (kg/day)	No. of Sites	Maximum	Minimum	Average	Loading (kg/day)
COD (chemical oxygen demand)	1.5	3597	14 786	12	2370	120	740	5000
Total Solids	1	3597	9 857	12	1580	80	490	3340
Total Nitrogen	-	-	-	6	6.8	0.5	3.5	24
Total Phosphorus	-	-	-	6	1.8	0.16	0.53	4
Copper	-	-	-	8	.07	.04	0.054	0.4
Iron	-	-	-	8	33	2	13.5	91
Manganese	-	-	-	7	5	.2	1.6	11

2) It is very likely that the leachates sampled included some dilution waters; therefore, flows of two or three times greater than those used could possibly be considered.
3) Attenuation of some leachate constituents is certain to occur as leachate moves towards the surface waters.

Wood waste leachate has not to date been considered a source of heavy metals. The iron and manganese loadings from wood waste leachates are less than one-half of those from refuse leachates. Metal analyses of fresh leachate springs at Port Mann and United Auto Wrecking are presented for comparison in Table 5.2. Comparison of this one wood waste leachate sample shows most of the constituent concentrations to be lower than in the refuse leachate, except notably for calcium, sodium and magnesium. Salt water storage may account for some of this. It could be expected that the mobility of metals from wood waste will be greater than from refuse because of the general acidic pH's of wood waste leachate, even though total contained metals may be lower. It is suggested that further investigation of wood waste leachate as a source of heavy metals is required.

5.5 Impact on Water Quality

5.5.1 Fraser River. The estimated COD loadings of 5000 - 14 800 kg/day are comparable to those from the large refuse landfills and amount to about 7-13% of the COD loading to the Fraser Estuary excluding the Iona Island Treatment Plant. Wood waste leachate also contributes daily some 91 kg of iron, 11 kg of manganese, and 24 kg of ammonia nitrogen to the Lower Fraser River system. The principal difference between the wood waste leachate discharges and the other discharges such as the STP, industrial discharges or urban storm sewers, is that the leachate discharges are often to small tributary streams rather than directly to the Fraser River. The result is that the water quality of the tributary streams is often degraded. (This point will be discussed in some detail in Section 7.)

TABLE 5.2 COMPARISONS OF METAL CONCENTRATIONS FROM LEACHATE SPRINGS AT A REFUSE LANDFILL AND A WOODWASTE DISPOSAL AREA

Parameter	Port Mann Leachate Spring (mg/l)	United Auto Wreckers Wood Fill Leachate Spring (mg/l)
Aluminum	33.45	0.71
Arsenic	.2 & < .15	< .15
Barium	.89	2.98
Calcium	1310	6400
Cadmium	.24	0.020
Cobalt	< .015	< .015
Chromium	.69	.015
Copper	0.06	0.22
Iron	280	79.4
Mercury	< .1	< .1
Magnesium	196	58.3
Manganese	31.3	9.3
Molybdenum	< .15	< .15
Sodium	781	2040
Nickel	.59	< .08
Lead	.39	< .08
Antimony	< .08	< .08
Selenium	< .15	< .15
Strontium	3.98	2.05
Titanium	1.17	0.04
Vanadium	0.11	< .05
Zinc	38.6	9.33
Silicon	38.2	18.5

6 SMALL MUNICIPAL LANDFILLS AND MISCELLANEOUS REFUSE DUMPS

6.1 Introduction

In considering present-day landfills or earlier dumps, more thought is normally given to the Burns Bog and Stride Avenue scale of landfill and not, for example, the small tiny burials of the old town of Barnett, or the refrigerators that were dumped over the dyke last week. This chapter contains an inventory of dumping occurrences, and presents information available on the industrial and the small, old and new, municipal landfills. Present problems of random dumping are discussed and a cursory review of the impact of a number of the small municipal landfills on local tributaries to the Fraser River is made.

6.2 Small Municipal Landfills

6.2.1 Introduction. This section covers one active municipal landfill (Maple Ridge) and seven closed landfills. Details on the sites have been obtained by reference to Kelly (1971), from information supplied by various municipal authorities, and from on-site inspections conducted during April 1979. Some chemical data have been supplied by the Pollution Control Branch for the Maple Ridge Landfill (EQUIS, 1978). The location of the eight sites are shown in Figure 6.2.1. Location maps and site plans for the seven closed sites can be found in Appendix E.

6.2.2 Maple Ridge Landfill. The Maple Ridge Landfill is located some 2000 metres to the southeast of Haney, adjacent to an unnamed tributary to Kanaka Creek. The landfill, opened in 1956, serves the needs of the residents of the District of the Municipality of Maple Ridge. Plate 6.2.1 is an aerial photograph of the site.

The landfill site covers an area of some 8 ha and lies adjacent to and 7.5 metres above an unnamed tributary to Kanaka Creek. The site is underlain by Whatcom Glacio-Marine deposits 7 metres to 91.5 metres thick consisting of silty clay, clayey silt, clay silt and sand. The

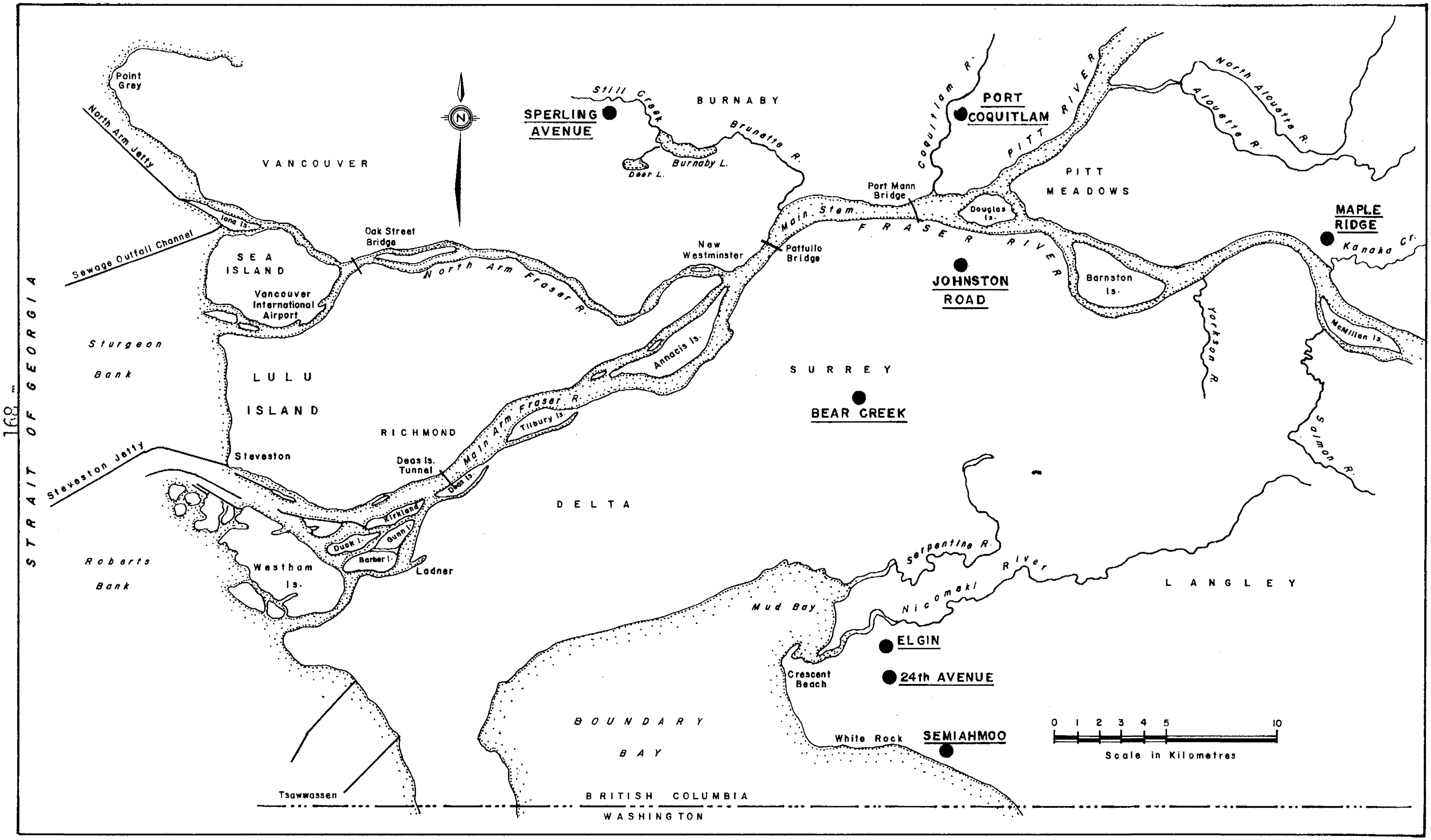


FIGURE 6.2.1 LOCATION MAP SHOWING THE DISTRIBUTION OF SMALL MUNICIPAL LANDFILLS IN THE FRASER RIVER ESTUARY AREA

PLATE 6.2.1 GENERAL VIEW OF THE MAPLE RIDGE LANDFILL SITE

coefficient of permeability of the material directly under the site is in the order of 10^{-5} to 10^{-7} cm/sec and should minimize any leachate discharge to groundwater. The location of the site is shown in Figure 6.2.2.

The site is permitted to receive some 180 m^3 of refuse per day and presently receives about 32 tonnes per day (Greg Miller, District of Maple Ridge, personal communication).

Fill is placed in 1.5 to 1.8 metre lifts, is compacted by bulldozer and is covered daily. Present fill height along the east flank is 7.5 to 9.0 metres.

The existing site was, as of early January 1979, close to being full. Application has been made to the PCB (December 15, 1978) to extend the landfill eastwards about 90 metres, or not closer than 30 metres from the unnamed tributary of Kanaka Creek. This application is presently under review and several options have been put forward as alternatives to an area expansion. The present and proposed areas are shown on the site plan, Figure 6.2.3. An option the municipality is now considering is to increase the fill height. Under the terms of the present permit, the municipality is required to minimize the passage of surface water through the waste fill. To meet this requirement and at the same time increase the useful life of the existing fill area, it is proposed to slope the final grade of the existing surface at 4% towards Cottonwood Drive. Preliminary computations by the Pollution Control Branch of the capacity available for future landfilling by increasing the height and grading the existing area, indicate the fill life can be extended by about 2-1/2 years. This estimate takes account of intermediate cover (20% volume), side and back slopes (5:1) and a final clay layer 0.9 metre thick. The permitted discharge quantity of 153 m^3/day has been used as the basis of the calculation, but no account was taken of the compaction which would normally reduce the placed refuse volume by a factor of at least 2.

The municipality is presently providing drainage control on the site. On April 18, 1979, observations indicated that leachate and

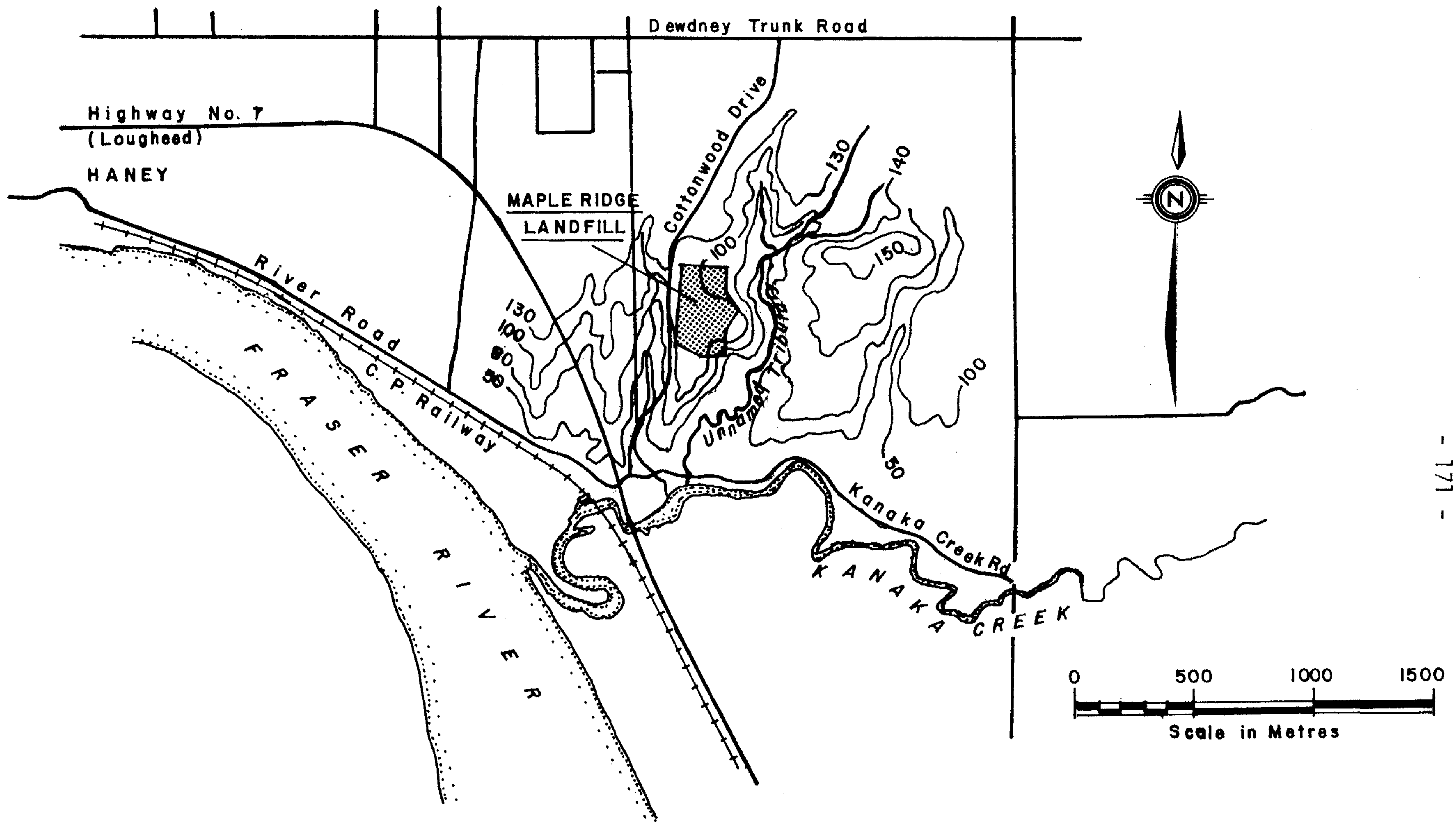


FIGURE 6·2·2 LOCATION MAP - MAPLE RIDGE LANDFILL

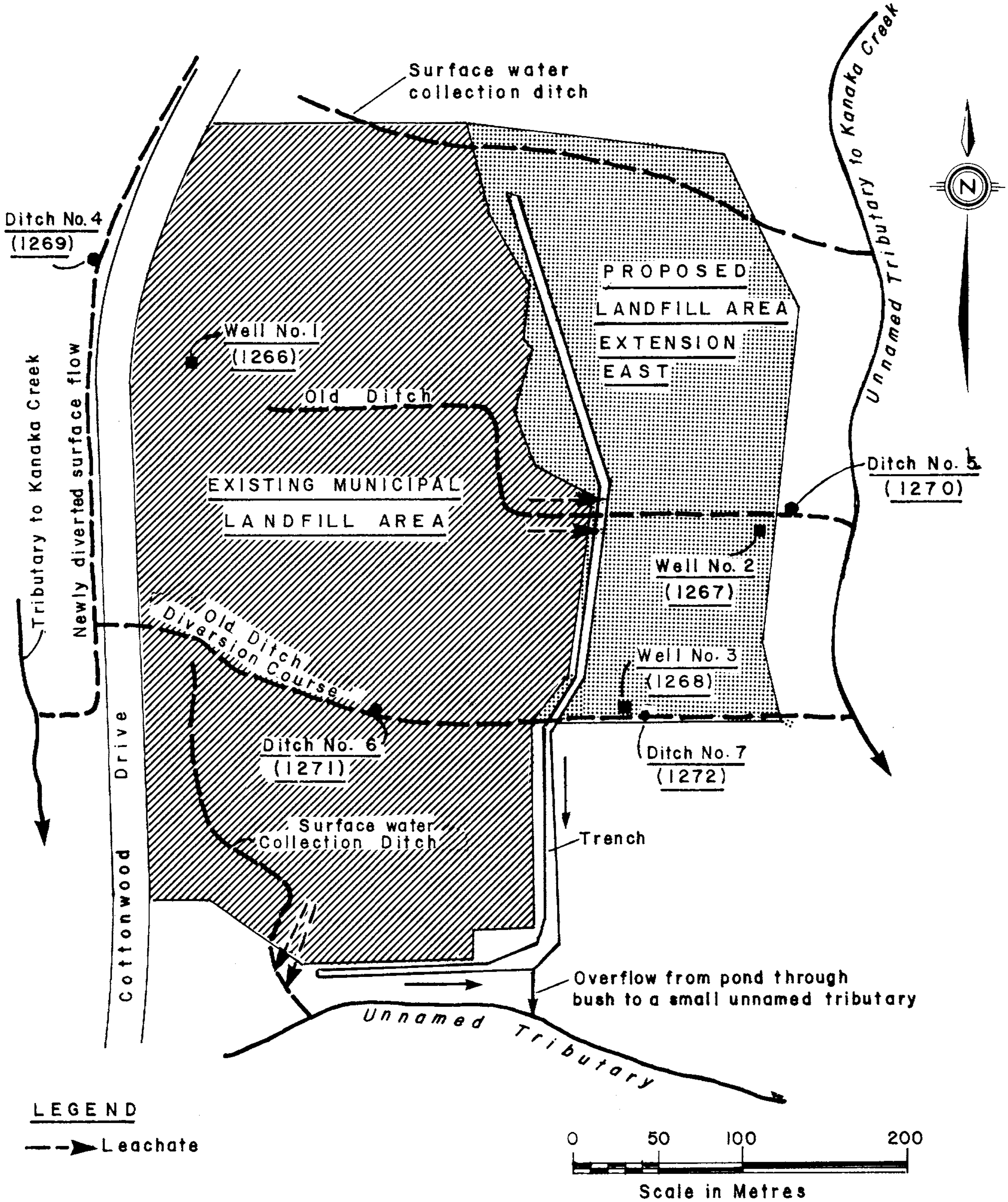


FIGURE 6.2.3 SITE PLAN - MAPLE RIDGE LANDFILL

seepage are presently being collected in a trench cut out along the south and east boundaries of the fill. A lagoon is to be constructed at the southeast corner and a flow measuring device will be installed. Methods of treating and handling the collected leachate have not been ascertained. Leachate now being collected in the trench is discharged into a small tributary of Kanaka Creek.

During April 1979, leachate was observed entering a surface drainage flow at the southwest corner of the fill but it did not enter the trench system. Instead, it was connected to another small unnamed tributary. Two major leachate discharges enter the trench along the eastern boundary. The trench drains into the former small tributary which, presumably, connects to the tributary along the eastern boundary of the fill and flows into Kanaka Creek. Recent changes in the drainage layout at the landfill have precluded the continued use of some PCB monitoring stations and a new sampling program will be set up. Three wells and four surface locations have been monitored at the landfill site. The monitoring locations and layout are shown in Figure 6.2.3. Site 1266 was a control, but was discontinued in April 1977, as it is now within the fill area; sites 1268 and 1267 are located between the fill and the unnamed eastern tributary. A comparison of the data from Well Sites 1267 and 1268 to data from Well 1266 (Table 6.2.1) before it was discontinued, suggests a migration of pollutants in an easterly direction; however, without data on the gradients, this is at best a suggestion. Well 1266 could possibly be located in inert materials or much older refuse.

Four surface flow monitoring stations have been used at the landfill, but recent diversions will require changes in the sample locations. Ditch Station No. 1269 was a control for the flow along Cottonwood Drive which was, until recently, diverted to run along the southern boundary of the fill. Ditch Station No's. 1271 and 1272 were established to determine the quality of this discharge after passing by and through the fill area and before entering the tributary of Kanaka Creek. Ditch Station No. 1270 monitored the leachate quality entering the tributary from approximately the center of the fill area. A comparison of the leachate ditch data with the control data (Table 6.2.2) shows

TABLE 6.2.1 PCB WELL MONITORING DATA* - MAPLE RIDGE LANDFILL (July 1975 to February 1978)

Parameter	Well #1 (1266)**		Well #2 (1267)		Well #3 (1268)	
	Mean	Range	Mean	Range	Mean	Range
pH	6.6	6.4 - 6.9	7.1	6.8 - 7.7	6.9	6.3 - 7.8
Spec. Cond.	88	73 - 112	1573	850 - 2120	1567	425 - 3620
TOC	19.3	8 - 29	80	42 - 188	101	13 - 186
Chloride	2.8	1.7 - 6.1	159	106 - 185	146	41 - 276
Alkalinity	36.1	25.5 - 43.5	673	264 - 869	735	158 - 1610
Ammonia	.119	.04 - .38	7.5	.635 - 16.1	37	.71 - 121
Total Kjeldahl	2.19	1 - 4	9.8	5 - 19	42	2 - 129
BOD	12	10 - 20	15	10 - 30	70	18 - 185
Total Chromium	.05	.018 - .086	.089	.006 - .51	145	.009 - .65
Total Copper	.09	.03 - .18	.21	.02 - 1.08	.29	.03 - 1.23
Total Iron	36.2	17.1 - 62.5	94	20.2 - 458	148	19.2 - 557
Total Lead	.007	<.001 - .014	.066	<.001 - .4	.093	.008 - .6
Total Zinc	.18	.09 - .32	.31	.04 - 1.2	.62	.15 - 1.1

* Based on 7-8 samples, reported as mg/l except for specific conductance (µmhos/cm) and pH.
** Discontinued April 22, 1977.

TABLE 6.2.2 MAPLE RIDGE LANDFILL, PCB SURFACE FLOW MONITORING DATA* (January 1976 to February 1978)

Parameter	Ditch #4 (1269)		Ditch #5 (1270)		Ditch #6 (1271)		Ditch #7 (1272)	
	Mean	Range	Mean	Range	Mean	Range	Mean	Range
pH	7.4	6.9 - 8	7.1	6.4 - 7.7	7.3	6.3 - 7.7	7.7	7.5 - 8.0
Spec. Cond.	78	46 - 113	1550	475 - 3210	650	123 - 2830	702	283 - 2350
TOC	4.8	1 - 9	209	1 - 525	33	7 - 84	26.8	13 - 64
Chloride	2.3	1.2 - 3.3	102	15.5 - 240	42.5	4.3 - 190	48	14.4 - 165
Alkalinity	36.2	18.1 - 56.1	517	153 - 1510	370	51.7 - 1330	325	108 - 1080
Ammonia	.03	.01 - .089	23	4.75 - 32	18.09	.90 - 84	17.35	4 - 62
Total Kjeldahl	.37	.11 - .66	31	5 - 106	20	1.28 - 87	19.6	5 - 68
Total Phosphorous	.085	.022 - .225	1.64	.030 - 8.26	2.34	.11 - 12.2	.226	.064 - .9
BOD	10	10	358	10 - 651	20	10 - 54	26	10 - 93
Total Chromium	.006	.005 - .010	.071	.005 - .49	.019	.005 - .053	.012	.005 - .045
Total Copper	.008	.001 - .020	.107	.001 - .75	.029	.003 - .13	.023	.003 - .11
Total Iron	2.5	0.5 - 8.0	150	2.4 - 1020	45.6	1.7 - 228	9.6	1.4 - 40
Total Lead	.007	.001 - .034	.008	.001 - .043	.025	.002 - .100	.015	.001 - .082
Total Zinc	.035	.005 - .080	5.8	.03 - 36.3	.718	.03 - 3.87	.068	.02 - .23
Fecal Coliform**	436	130 - 920	12 475	800 - 24 000	2222	490 - 5400	16 110	330 - 24 000

* Based on 6-8 samples, reported as mg/l except for fecal coliform (MPN/100 ml), specific conductance (μmho/cm) and pH.

** Based on three samples.

an across-the-board rise in the concentration of measured parameters. The new trench now collects these discharges at one point.

It is anticipated that a water quality study presently being carried out by the GVRD in connection with a Regional Park may give some indication of the impact of the leachate on Kanaka Creek.

A rough estimate of the leachate constituent loadings from the Maple Ridge Landfill can be made based on an annual precipitation rate of 1500 mm, an evaporation loss of 225 mm, and the concentration data from ditch site #5. Ditch sites #6 and #7 data are not used as there is no estimate of dilution as a consequence of the diverted ditch. Calculated loadings are presented in Table 6.2.3.

6.2.3 Johnston Road Landfill - District of Surrey. The Johnston Road Landfill is located in the District of Surrey, just to the east of the Port Mann Freeway. The fill was operated from 1967 to 1969 (Kelly, 1971), from the closure of the Bear Creek Landfill to the opening of the Port Mann site.

The fill occupies the top end and parallels the western side of a small ravine that starts at the corner of Johnston Road and 122nd Avenue. It would appear that drainage was culverted through the site as a series of manholes can be seen in the fill. The drainage (or the culverts) terminates in an open stream further down the ravine.

Discharges from the site do occur and can be seen in an open ditch parallel to the manholes.

The creek that occupies the ravine once flowed directly towards the Fraser River before it was intercepted by CNR ditches. With the construction of the Port Mann fill, the creek was directed to the west around the site. This can be seen in Figure 3.3.5 in the previous Section on the Port Mann fill.

In 1972, the Greater Vancouver Regional District monitored the creek above the Port Mann site as control for their study of leachate from the Port Mann site (GVRD, 1972). A summary of those data is presented in Table 6.2.4. Leachate still flows from the site, but no information has been collected since 1972.

TABLE 6.2.3 MAPLE RIDGE LANDFILL - LEACHATE LOADINGS

Parameter	Average Concentration (mg/l)	Average Daily Loading (Kg/day) (0.003 m^3/sec)
COD*	570.0	159.0
TOC	209.0	58.0
Chloride	102.0	29.0
Alkalinity as $CaCO_3$	517.0	144.0
Ammonia	23.0	6.5
Total Kjeldahl	31.0	8.6
Chromium	0.071	0.02
Copper	0.107	0.03
Iron	150.0	42.0
Lead	0.008	0.002
Zinc	5.8	1.62

* COD from BOD at 1.6:1 ratio.

TABLE 6.2.4 MONITORING DATA - GVRD 1972 - JOHNSTON ROAD LANDFILL

Parameter	No. of Samples	Maximum (mg/l)	Minimum (mg/l)	Mean (mg/l)
Temperature (°C)	16	21	3	9.9
pH (units)	17	7.5	6.7	7.4
DO	17	13.4	90	11.0
Spec. Cond. (μmhos/cm)	17	943	290	658
Total Solids	16	837	259	506
Total Filterable Solids	14	740	36	410
Total Volatile Solids	14	294	9	124
Total Dissolved Solids	16	828	254	496
BOD	4	8.0	< 1	3.2
COD	13	42	< 4	15.8
T. Alk. as $CaCO_3$	16	89	31	71.2
Chloride	16	159	15	39.1
Sulphate	7	410	12.6	253.5
Total Phosphate	10	-	< .01	9/10 b/d
Ammonia	12	5.2	< .01	3.0
Phenolics	3	< .01	< .01	< .01
Sodium	16	30.3	15.0	21.8
Potassium	16	67.0	3.4	16.2
Calcium	16	150	9.7	81.5
Sodium	16	26	3.8	9.8
Copper	15	b/d	< .01	7/15 b/d
Lead	16	b/d	b/d	b/d
Iron	16	3.90	0.47	1.29
Zinc	16	0.44	0.06	0.19
Cadmium	16	b/d	b/d	b/d
Chromium	16	b/d	b/d	b/d
Nickel	16	b/d	b/d	b/d
Arsenic	1	< .1	< .1	< .1
Manganese	14	1.46	0.18	0.42
Aluminum	1	< 1	< 1	< 1
Total Coliforms MPN/100 ml	17	43 000	40	5325
Fecal Coliforms MPN/100 ml	17	23 000	< 30	1705

b/d = below detection

The site was used for municipal and commercial refuse as evidenced by some remnant piles of gyproc and appliances that missed being covered. Much of the site is overgrown with saplings and remains vacant except for some trailers parked along the east side of the road.

6.2.4 Bear Creek Landfill - District of Surrey. The Bear Creek Landfill is located in the District of Surrey, midway between the King George Highway and 40th Street south of 88th Avenue. The site was operational up to some time in 1967, receiving municipal and commercial garbage.

The fill area is bounded by Bear Creek on the west and an unnamed tributary to Bear Creek on the east. The fill limits can be determined from settlement and the presence of iron staining in the creeks. Fill height is estimated to be 2.5 to 3.0 metres (Kelly, 1971). Watkins (1970) reported that the landfill bordered the creeks for 400 metres.

Kelly (1971) reported that groundwater percolates through the dump into Bear Creek and that deposits of iron sulfide left by slimes growing during the winter months have left a rusty colour along the stream bed. Watkins (1970) reported that along the 400 metres of fill area that borders the creek, leachate seeps into the stream at many points and at some points forms stagnant pools that, in turn, drain into the creek. The site was visited in April 1979, and the above situation prevailed. Iron staining is also evident in the tributary, thereby indicating drainage to both water courses.

Kelly (1971) reported the upper soil to be littoral beach deposits of medium to coarse sand resting on clay. Depending on the depth of the clay, all or part of the leachate would go the creeks. If the sand extends to some depth, partial discharge to groundwater could be anticipated. The fill upon completion was capped with a till-like material (Kelly, 1971).

Some analysis (Watkins, 1970; Kelly, 1971) of Bear Creek above and below the site as well as of leachate composite are presented in Table 6.2.5. It is not known if any monitoring and analyses have been done at the site since 1971.

TABLE 6.2.5 SAMPLE ANALYSES FOR BEAR CREEK LANDFILL

Determination*	July 1970**			May 1971	
	1	2	3	1	2
pH	-	-	-	-	-
Sulphate	-	-	-	8.8	6.4
Chloride	-	-	-	8.0	9.2
Ammonia-Nitrogen	-	-	-	.8	.19
Zinc	-	-	-	.09	.15
Copper	.005	.01	.03	< .01	< .01
Lead	-	-	-	< .05	< .05
Iron	.08	.8	.5	1.8	.7
Aluminum	.008	.01	.1	-	-
Barium	.004	.008	.07	-	-
Boron	.04	.06	.5	-	-
Chromium	.001	.001	.01	-	-
Sodium	9.0	11.0	110.0	-	-
Potassium	2.3	5.4	90.0	-	-
Calcium	4.0	10.0	90.0	-	-
Magnesium	1.6	6.0	50.0	-	-
Manganese	.008	.10	2.5	-	-
Total Dissolved Solids	79.0	118.0	896.0	-	-
Volatile Dissolved Solids	1.0	49.0	208.0	-	-
Fixed Dissolved Solids	78.0	69.0	688.0	-	-

* Values in mg/l except pH.
** After Watkins, 1970.

Sample Site:

Sample 1: Sample was taken from Bear Creek upstream from the dump.
Sample 2: Sample was taken from Bear Creek downstream from or adjacent to the dump.
Sample 3: The sample was a composite made up of leachate found percolating into the creek.

The site is zoned for parks and part of the site has been used for a running track. The rest of the site remains vacant.

6.2.5 Elgin Landfill - District of Surrey. The Elgin Landfill is located in south Surrey, south of Crescent Road between 140th Street and 144th Street. The site covers an area of some 1.2 ha, with the fill along the north face about 6-7 metres high. Elgin Landfill was reported as being used up to 1965 or 1966; when it started is not known.

An open ditch along the north face drains towards an unnamed tributary of the Nicomekl River. Surface water draining from the eastern portion of the site was directed towards the ditch.

During the April 1979 site inspection, leachate seeps in the form of iron-precipitation were noted along the north boundary as were pools in the area of the eastern boundary. No analytical information is available concerning refuse constituents or leachate compositions for the Elgin Landfill.

6.2.6 Port Coquitlam Landfill. The Port Coquitlam Landfill is located in the City of Port Coquitlam 1000 metres north of the Red Bridge in a slough on the east side of the Coquitlam River.

Refuse was reported as being used to reclaim a portion of the slough leading to the Coquitlam River. The fill is estimated to be greater than 2.4 metres and has been in place since at least 1971. At that time it was used only for yard rubbish (Kelly, 1971). The site was closed to that disposal in early 1979.

The site was covered with a material resembling a till (Kelly, 1971) and is presently used for storing City equipment and materials.

Groundwater would be expected to move freely through the fill into the slough and thence to the Coquitlam River. There was no visual evidence of leachate at the time of the April 1979 visit. No analytical data are available concerning the Port Coquitlam Landfill.

6.2.7 Sperling Avenue Landfill - Corporation of Burnaby. The Sperling Avenue Landfill is located in North Burnaby in a boggy area west of Sperling Avenue between Laurel and Still Creek.

Municipal refuse from the northern half of Burnaby went to the Sperling Avenue Landfill between 1963 or 1964 and 1967; after 1967, it was disposed of at the Stride Avenue Landfill. The fill operation consisted of excavating continuous trenches 1.8 metres deep and some 10.7 to 12.2 metres wide in peat or peat and soft clay, and then placing some 3.7 to 4.3 metres of refuse in the trenches. The excavated peat was then used as a final cover. No analytical data are available concerning the refuse or leachate at the Sperling Avenue Landfill.

6.2.8 Semiahmoo Bay and 24th Avenue Landfills - District of Surrey. Two other very small and old landfills are located in the District of Surrey.

The Semiahmoo Bay Landfill was closed in the early sixties and now supports a baseball diamond. Drainage would be towards Campbell River. There was no visual indications of leachate during the April 1979 site inspection.

The 24th Avenue site is situated at the head of the creek flowing past the Elgin Landfill site. The 24th Avenue Landfill, reportedly closed in the early fifties, is marked by a jog in 24th Avenue taken to avoid potential settlement problems. No analytical data are available for either the Semiahmoo Bay or 24th Avenue Landfills.

6.3 Miscellaneous Dumps

6.3.1 Introduction. This section covers those dumps not included elsewhere in this report. These dumps are usually small, discrete, aesthetically displeasing and, on occasion, a source of concern with regard to small watercourses.

A complete inventory of miscellaneous refuse disposal carried on within the Fraser River Estuary study area defies any adequate accounting; however, an attempt will be made to provide an overview. Generally, miscellaneous refuse disposal can include permitted industrial refuse disposal sites, local opportunistic type landfilling with refuse, and truckload quantity littering (convenience dumping).

Permitted industrial refuse disposal encompasses an extremely diverse array of materials including offices wastes, construction wastes, scrap metals, asphalt roofing manufacturing discards, cement plant washouts, boiler ash, house demolition material, etc. These dump sites range from low lying areas within a particular industry operating area, which usually accommodates refuse loads of 1 to 10 m^3/day, to large dumps sites in low lying areas adjacent to the Fraser River where volumes of 10 to 100 m^3/day may be disposed of. The refuse consistency and volume dumped at any particular site may vary considerably in the case of demolition materials or be quite uniform in the case of an asphalt roofing plant waste.

Refuse dumping as part of land development operations has been accepted as an economic alternative to the expense of trucking in "clean fill" for activities such as dyke construction or industrial site preparation. This practice of dumping accommodates as much material as required for completion of a project over the short construction period and may include such material as land clearing refuse, road construction wastes, demolition material, etc.

Convenience dumping includes all varieties of refuse in volumes from garbage bag quantities to several truck loads, disposed of in "locations of convenience" with a fair concentration of them along vacant stretches of the Fraser River shoreline.

Figure 6.3.1 is a map of the study area showing the permitted industrial fills and the occurrences of dumped refuse.

6.3.2 Permitted Industrial Disposal Sites. A listing of 25* refuse disposal sites permitted and active, and four permitted but since closed, was supplied by the B.C. Ministry of Environment from their EQUIS data retrieval system (Clark, 1975; Ellis and Clark, 1977). They are listed together in Table 6.3.1 with pertinent information extracted from the respective PCB permits.

---

*This number does not include the few agricultural refuse disposal sites such as silage or cattle manure dumps that are also under PCB permit.

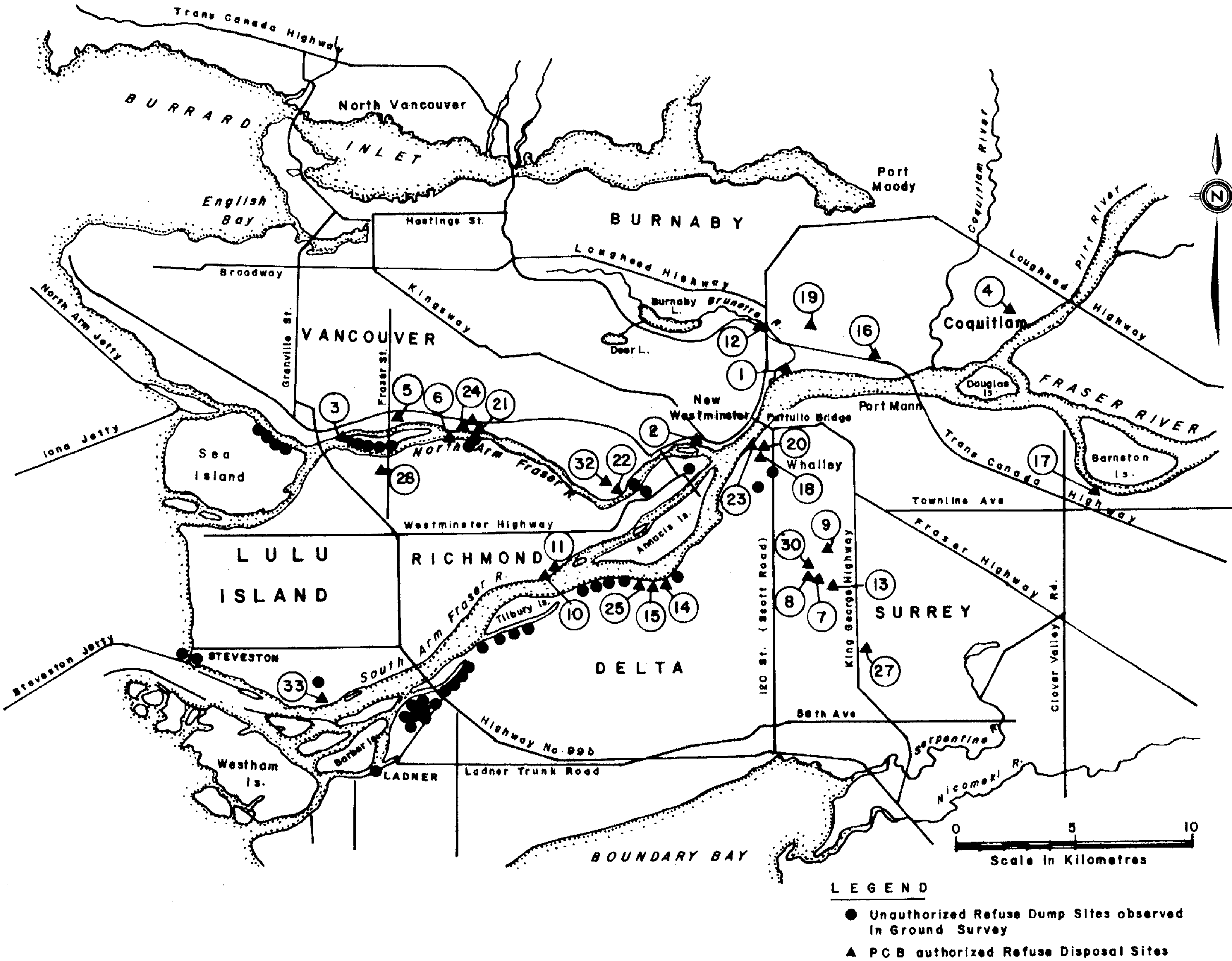


FIGURE 6·3·1 AUTHORIZED AND UNAUTHORIZED MISCELLANEOUS REFUSE DISPOSAL SITES IN THE FRASER RIVER DELTA AREA

TABLE 6.3.1 INDUSTRIAL REFUSE SITES

Map No.	Company Name	Permit No.	Refuse Type	Volume Under Permit	Receiving Waters	Monitoring Requirements	Comments
1	Canadian Forest Products Ltd. (New Westminster)	PR-1655	ash, boiler ash, sand & gravel	4.0 yd 3/day	-	-	leachable refuse, leachate high pH, alkalinity
2	MacMillan Bloedel Ltd. (New Westminster)	PR-1655	ash, burner ash, cinders, sand and gravel	1.0 yd 3/day	-	-	probable leachate generation to groundwater
3	Western Canada Steel Ltd. (Richmond)	PR-2086	ash, cinder, bricks, misc. metal	18.0 yd 3/day	-	-	probable leachate generation to groundwater
4	Kennametal Inc. (Port Coquitlam)	PR-2351	ash, granular carbon refractory brick, refractory lining	1.5 yd 3/day	surface water	3 sample sites PCB #0301198 #0301199 #0301200	leachate probably toxic with additions of effluent treatment sludge
5	Weldwood of Canada Ltd. (Vancouver)	PR-2425	ash, boiler ash, sand & ash from multiclone collector 4% paint and glue	3.0 yd 3/day	-	-	leachate likely to be moderately toxic
6	Mainland Foundry & Engineering Ltd. (Richmond)	PR-2510	ash/foundry, foundry ash, sand, gravel, inorganic sludge	4.0 yd 3/day	-	-	probable leachate generation to groundwater
7	Reliance Foundry Comp. Ltd. (Surrey)	PR-2250	foundry waste, sand, gravel, inert debris	5.5 yd 3/day	surface water	2 sample site PCB #0301122 #0301123	leachate found to be very high in metal contamination
8	Rober Industries Ltd. (Surrey)	PR-3082	foundry waste, sand, gravel, inert debris	1.0 yd 3/day	-	-	probable leachate generation
9	Letson and Burpee Ltd. Foundry (Surrey)	PR-2085	foundry waste, sand, gravel, inert debris	4.0 yd 3/day	groundwater	-	probable leachate generation
10	Con-Forces Products Ltd. (Richmond)	PR-3867	cement plant wastes, paper and wood	15.0 yd 3/day	-	-	-
11	Lafarge Canada Ltd. (Richmond)	PR-4409	cement plant wastes, inert & non-combustible debris	70.0 yd 3/day	-	-	-

TABLE 6.3.1 INDUSTRIAL REFUSE SITES (Continued)

Map No.	Company Name	Permit No.	Refuse Type	Volume Under Permit	Receiving Waters	Monitoring Requirements	Comments
12	Crane Canada Ltd. (Coquitlam)	PR-3141	porcelain/clay products, inert	3.0 yd 3/day	-	-	-
13	Fairey & Co. Ltd. (Surrey)	RR-3578	porcelain/clay products, broken brick	15.0 yd 3/day	-	-	registered refuse
14	Mr. & Mrs. White (Delta)	PR-5135	asphalt waste	10.0 yd 3/day	groundwater	-	probable leachate generation
15	Industrial General Prod. Ltd. (Delta)	PR-4923	asphalt waste	10.0 yd 3/day	groundwater	-	probable leachate generation
16	Columbia Bitulithic (Coquitlam)	PE-2516	asphalt waste, earth spoils, sand & gravel	-	groundwater	-	probable leachate generation
17	Cloverdale Demolition & Salvage (Surrey)	PR-4441	demolition waste, lumber, brick, hog fuel	23.0 yd 3/day	-	-	probable leachate generation
18	United Auto Wrecking Ltd. (Surrey)	PR-1685 AR-5245	demolition waste, lumber, demolition material, scrap metal	1000 yd 3/day	groundwater	2 sample sites #0301062 #0301063	leachate generated at both sides
19	Pacific Disposal Ltd. (Coquitlam)	PR-2259	demolition mixed inert	50 yd 3/day	groundwater	2 sample sites PCB #0301284 #0301285	probable leachate generation, not detectable in receiving waters
20	B.C. Land Recovery (Surrey)	No permit	demolition constr. waste, earth fill	-	-	-	probable leachate generation, short term fill operation
21	MacMillan Bloedel Ltd. (Vancouver)	RR-4264	forest ind. sludge from epoxy finishes	3.0 yd 3/day	groundwater	-	probable leachate generation, registered refuse
22	Belkin Packaging Ltd. (Richmond)	RR-4932	forest ind. cardboard, paper screen waste	24.0 yd 3/day	groundwater	-	probable leachate generation, registered refuse
23	Weldwood of Canada Ltd. (Surrey)	RR-4973	forest ind. glue residue from lagoons	-	-	-	probable leachate generation, registered refuse
24	MacMillan Bloedel Ltd. (Vancouver)	RP-1667	ash, grate ash from hog fuel furnace, sand, slag, charcoal	20.0 yd 3/day	-	-	leachate toxic at natural pH

TABLE 6.3.1 INDUSTRIAL REFUSE SITES (Continued)

Map No.	Company Name	Permit No.	Refuse Type	Volume Under Permit	Receiving Waters	Monitoring Requirements	Comments
25	Vito Steel and Barge Construction (Delta)	PR-4468	gyproc, hog fuel, red cedar, bark mulch	unspecified	groundwater, ditch, small creek, Fraser River	-	leachate highly toxic, under order by PCB to isolate dump with impermeable dykes
29	Weldwood of Canada Ltd. (Vancouver)	AR-2426	boiler ash	-	groundwater	-	permit application withdrawn in 1974, connected to Municipal Sewer
30	Holmes Insulation Ltd. (Surrey)	AR-2647	asbestos products	-	groundwater	-	dumping ceased in 1973, permit never issued
32	Domtar Construction Materials Ltd. (Vancouver)	AR-3128	general	-	groundwater	-	permit application withdrawn in 1974, dumping ceased
33	Crown Zellerbach Canada Ltd. (Richmond)	AR-3460	hardboard, paper	-	groundwater	-	permit application withdrawn in 1974. Dump site used.

The volume of refuse dumped at these active sites totals approximately 1000 m^3 (1200 yd^3/day) and, of this volume, 700 m^3/day (1000 yd^3/day) represents a short-term fill operation in North Surrey.

It is not likely that all the industrial disposal sites are permitted, some exist simply because individuals may believe that the consequences of the dumping do not warrant the bureaucratic endeavour — an example in point may be some of the concrete plants.

An estimated 20 concrete batch plants located within the study area dispose of excess concrete from mix trucks to the ground within the plant site at the end of each working day. This waste is often dumped directly onto the foreshore of the Fraser River (Plate 6.3.1).

6.3.3 Refuse for Land Development. From a four-hour ground survey of the North and South arms of the Fraser River from Garry Point to the Pattullo Bridge, five active dyking projects were observed to be using refuse as fill (i.e.: South side of Deas Island Slough, Northeast side of Sea Island, River Road dyke improvement in North Delta). Evidence of active dumping of mixed fill over vast quantities of refuse on the river side of the dyke can also be seen at Brittain Steel, in New Westminster. Similar situations can be observed in numerous other areas; for example, near the Ladner Sewage Treatment Lagoons, where demolition refuse mixed with soil fill was dumped on the slough side of the River Road in Delta.

6.3.4 Convenience Dumping. A quick survey can reveal countless incidents of convenience dumping or large volume littering within the study area. A recent four-hour survey by car indicated examples from a truck load of banding iron dumped on the 5700 block of River Road in Delta to household garbage dropped along the dyke bordering North Richmond. The areas most commonly used appear to be secluded, vacant municipal or private lands, or areas of convenience such as ravines or pulloffs along dyke roads. The accreted lands west of Ferry Road in Delta were examined on foot, and approximately a dozen dumpings of

PLATE 6.3.1 BATCH PLANT AND SOLID WASTE DISPOSAL ALONG THE FRASER RIVER FORESHORE

refuse were observed on the 3.5 or more hectares above high tide. Plate 6.3.2 shows part of the estimated 1000 m^3 of refuse that has been dumped on the foreshore at Garry Point in Richmond. Much of the convenience dumping occurs directly onto the foreshore areas.

6.3.5 Impact of Miscellaneous Refuse Dumps. Discharge from two sites, Reliance Foundary and Kennametal Inc., have increased the heavy metals levels in drainage ditches downstream from the sites (EQUIS, 1978). The data for these sites are summarized in Table 6.3.1. The ditch below Reliance Foundary flows into Bear Creek; the one below Kennametal Inc. eventually flows into the Pitt River. The effects of the discharge have not been investigated.

The fate or consequence of the leachate from many of the other sites, permitted or otherwise, is not known; most of the leachates are believed to go to ground. It can be anticipated that there is little reason for concern, although it should be noted that there have been instances of well contamination from very small fills. Such an instance was the one of manganese poisoning through the disposal of some 150 dry cell batteries near a domestic water well in Japan (McKee et al, 1963).

The impact of convenience dumping by a company or an individual is usually totally out of proportion to its volume or its minimal effect on water quality. It constitutes a visual insult to the environment. Clearly, the control of convenience dumping lies with the local government in these areas in which it occurs. Whether they choose to control the situation will depend upon whether or not there is a perceived public health problem or civic pride is adversely affected.

PLATE 6.3.2 CONVENIENCE REFUSE DUMPING AT GARRY POINT IN RICHMOND

7 DISCUSSION

All the leachate emanating from the landfills in the study area reaches the Fraser River, most of it at this time directly through surface waters, some through groundwaters, and the remainder through sewers discharging at the Annacis Island STP. To discern or define the impact of a direct leachate discharge on the Fraser River Estuary is difficult, if not impossible. If effects on the Estuary are to be defined, they are more likely to be caused by the cumulative discharges from all the sources within the Vancouver Lower Mainland. Local and specific impacts can be seen where wood waste leachate has entered small tributaries, and adjacent to one refuse landfill where metals may have accumulated in Fraser River sediments. While the impact of leachate discharges may be difficult to define, the presence of the discharges is rarely hard to discern because they are often marked by black plumes or iron staining.

It is possible to arrive at some estimate of the amount of specific pollutants entering the Fraser Estuary as a result of land-filling, although these estimates are qualified. The pollutant loadings are discussed, as are the specific problems that have occurred from the placement of wood waste in areas such that the resulting leachate has degraded small tributary waters. The loadings and the impact of the leachates from the four different landfill groups are discussed separately.

7.1 Large Municipal Landfills - Active

The five landfills in this group are Burns Bog, Richmond, Braid Street, Port Mann and Leeder. They are generally comparable in their depths of fill and physical settings, if not in their geologic settings or size. Except for wood waste, almost all of the municipal, commercial and industrial solid waste generated in the study area is disposed of in the five landfills.

The average concentration and average daily loadings of leachate constituents from the Burns Bog, Richmond, Braid Street and Port Mann landfills are compared in Tables 7.1 and 7.2. Loadings were not estimated for the Leeder landfill, although they are probably similar to those for Port Mann as the daily tonnages placed are comparable, bearing in mind that the putrescible content in the Leeder landfill refuse would be lower and some ground attenuation will occur. In the subsequent discussions, the total values used are totals of only the four for which loadings estimates were made. The loadings were calculated, for the most part, from an extensive analytical data base and a non-existent flow base. It is in the consideration of large landfills that it becomes quite apparent that where loading information is required the current monitoring procedures are not adequate. It must be recognized, however, that while it is comparatively easy and inexpensive to monitor effluent flows through pipes, it would have been an expensive and a very difficult technical process to monitor the diffuse discharges from the landfills. With the installation of collection works and pumping stations at the Burns Bog and Port Mann landfills, it will become much easier to measure leachate flows and ascertain the loadings with some accuracy.

A comparison of the concentrations of the leachate constituents in Table 7.1 would, initially, suggest a wide variance in the average concentration for any given constituents between the four landfills. However, in consideration of the extreme variability of leachate concentrations reported in the literature (Table 2.2), the variability in concentrations at any one landfill, for example Burns Bog (Table 3.1.2), and the wide range of refuse composition and placement methods, the concentrations could be considered as quite comparable. A comparison can be made in terms of the yield of total leachate constituents, that is, tonnes of total solids* in the leachate per tonne of refuse placed per year (Table 7.3). The yields for Braid Street (Coquitlam), Port Mann

* Total solids calculated from the sum of available cations and anions where total solids analysis is not available.

TABLE 7.1 LEACHATE CONSTITUENTS - AVERAGE CONCENTRATIONS - LARGE ACTIVE LANDFILLS

Parameter	Burns Bog	Richmond	Braid St. (Coq.)	Port Mann	
pH (units)	7.4	7.0	6.4	7.5	
COD	255	700	842	250	
NFR	-	80	211	92	
T.Alk. as $CaCO_3$	-	-	400	695	
SO_4	-	75	33.3	10.6	
Chloride	281	300	173	90	
Total Phosphate	-	-	-	.30	
Ammonia	92.8	50	28.9	37.2	
Arsenic	.010	< .2	.01	< .15	4/4
Barium	-	-	-	.43	
Calcium	65*	-	119	177	
Chromium	.05	0.03	< .037	< .015	4/4
Copper	.01	.02	< .07	< .01	4/4
Iron	6.2	15	42.7	32	
Mercury	< 0.5- 1.0	< .10	-	< .1	4/4
Magnesium	32*	-	27.3	24	
Manganese	.355*	5	3.07	4.6	
Sodium	165*	250	108	76	
Nickel	< .01	< .2	< .13	< .08	3/3
Lead	.02	< .01	.052	< .08	4/4
Strontium	-	-	-	1.13	
Zinc	.33	.3	.59	< .02 - .3	
Silicon	-	-	-	8.2	
Aluminum	.39*	.5	-	.13	
Total Coliform	-	-	-	440/100 ml	
Fecal Coliform	-	-	-	-	
Tannins & Lignins	17.3	-	-	-	
Fluoride	.19	-	-	-	
Specific Conductance	2180	-	1915	1635	
Cadmium	< .0003	-	-	< .015	

* Dissolved
Units in mg/l except where indicated.

TABLE 7.2 LEACHATE CONSTITUENT LOADINGS - LARGE ACTIVE LANDFILLS (kg/day)

Parameter	Burns Bog	Richmond	Braid Street	Port Mann
COD	370	3150	846	435
NFR	-	360	210	80
T. Alk. as $CaCO_3$	-	NC	NC	540
SO_4	-	336	34	11
Chloride	682	1350	173	80
Total Phosphate	-	NC	NC	0.4
Ammonia	229	225	29	40
Barium	-	NC	NC	0.1
Calcium	177	NC	123	138
Cadmium	NC	NC	NC	NC
Chromium	0.027	0.13	NC	NC
Copper	-	NC	0.065	NC
Iron	16	67	43	27
Mercury	-	-	-	-
Magnesium	88	-	28	18
Manganese	1.14	22	3.3	1.4
Sodium	45	1120	110	63
Nickel	0.039	NC	<0.13	NC
Lead	0.003	NC	0.050	NC
Strontium	NC	NC	NC	0.9
Zinc	0.79	1.30	0.6	0.3
Silicon	NC	NC	NC	5.0
Aluminum	0.29	2.2	NC	0.12
Arsenic	0.031	NC	NC	NC
Fluoride	0.6	NC	NC	NC

NC = Not Calculated

TABLE 7.3 COMPARISON OF THE PERCENT YIELD OF LEACHATE CONSTITUENTS (Tonnes of constituents in leachate stream/year/tonne of refuse placed per year)

Parameter	Burns Bog	Richmond	Braid Street	Port Mann
Tonnes of refuse placed per year	227 000	245 000	151 000	66 000
Tonnes of constituents* leached per year	1 685	1 484	373	388
Percent yield tonnes/tonnes	0.58%	2.0%	0.5%	0.81%

* Tonnes of organics taken as 0.5 of tonnes COD.

and Burns Bog landfills are comparable although they are 1/4 to 1/2 of reported values (Cameron, 1978a). The larger value for Richmond landfill may be the result of higher flushing rates.

With respect to leachate concentrations, there is little that can be concluded from any of the specific values; however, several observations can be made from the data in Table 7.1.

The pH of 6.4 for Braid Street does not reflect the trend of increasing alkalinity reported in Section 3.4.4; however, this is an average of only seven analyses.

The high chloride levels reported for the Richmond and Burns Bog landfills most likely reflect the addition of saline water from the pumping of dredged sand onto the site. This is generally supported by higher sodium values in the leachate from these two landfills.

The higher sulphate values at Richmond landfill are probably the result of landfilling large quantities of gypsum board. Ammonia concentrations in Burns Bog leachate are higher without an obvious explanation. One possibility may be that hog fuel, which is used to a greater extent at the other fills, may be reacting with a portion of the ammonia resulting in lower concentrations at those sites.

The leachate from these large active landfills is characterized by pH, trace metal content, COD, ammonia concentration and toxicity. These five parameters are discussed in some detail.

7.1.1 pH. The mean of the pH values of leachates measured at the large landfills including the three large closed sites was, with the exception of Braid Street, 7.0 or greater. This is not extraordinary; however, it is unusual that it occurs at all but the one large landfill in the study area, and is noteworthy because the majority of leachates found elsewhere have acidic pH values.

An explanation put forth for the elevated pH has been the presence of large quantities of demolition debris and/or gypsum boards in the mattress fill (AESL, 1974). This may not be the full explanation, since at several of the landfills such mattress materials are not present in large quantities. Acidic conditions normally occur in refuse landfills through the production of volatile acids which, while not absent during aerobic biological activity, are enhanced under anaerobic conditions. Volatile acid concentrations in landfill leachates are generally lower under high infiltration conditions (Cameron, 1978a). At the same time, methane generation has been reported to be higher, suggesting that methane producers increase with infiltration, or more likely may not be suppressed by high volatile acid concentrations. It is suggested that the elevated pH values may be the result of low volatile acid content in the leachates as a consequence of one or more of the following factors or combinations:

- low biological activity, due to high infiltration rates;
- a high rate of volatile acid conversion to methane;
- low volatile acid production, due to aerobic conditions;
- buffering, as reflected by the alkalinities of the leachate.

7.1.2 Trace Metals. The concentration of trace metals measured in leachate from four of the five landfills (Burns Bog, Richmond, Braid Street and Port Mann) are, with the exception of iron, manganese, zinc, aluminum and arsenic, generally below 0.1 mg/l and at most times are below detection limits. (It should be noted that at the various laboratories, detection limits for any given constituent can vary by nearly an order of magnitude). Arsenic concentrations are normally below detection, but on one or two occasions there have been elevated levels at several of the landfills. This situation should be watched. Research at UBC into the movement of metals in refuse indicates that arsenic is not retarded as well as most other metals (Jackman et al, 1979). The trace metal concentrations observed could be explained, at least in part, by the following:

1) the elevated pH's do not favour the pickup or transport of most of the metals; a decrease of the pH by one unit can increase the solubility of many of the metal compounds a hundredfold;
2) the assumed lower biological activity could restrict the degradation of many refuse components; therefore, the release of bound metal ions from the refuse would be restricted;

3) precipitation of iron hydroxide in leachate ditches prior to the point of sampling can strip other metals;
4) precipitation of metal sulphide compounds in the lower portions of the fills. Sulphides have normally not been found in these leachates, although one analysis of Richmond landfill leachate showed a sulphide concentration of about 30mg/l. The general absence of sulphide in the leachates could suggest that (i) metal sulphides are precipitating, removing the sulphide from the leachate, or (ii) the sulphide is not being formed.

Loading calculations were not done for those metals where the concentrations were always below detection. When concentrations in the sample set were above and below detection, the loadings were calculated from average values, assuming a concentration of 1/2 the detection level for those below detection.

In absolute terms, there are quantities of trace metals entering the Fraser River system; however, in comparison to the sewage treatment plants and storm runoff, the large active refuse landfills are not major sources of trace metals, other than for iron, manganese, zinc and aluminum. It is estimated that an average of 150 kg of iron, 28 kg of manganese, 3 kg of zinc and 2 kg of aluminum enter the Lower Fraser River daily in the leachate from the Burns Bog, Richmond, Braid Street and Port Mann landfills. The importance of a metal source should not be based solely on quantity, since the form the metals are in determines their availability for biological uptake. A large portion of the metals in leachate are dissolved and many are chelated (Knox and Jones, 1979).

A considerable volume of metallic and chemical sludges and process washwaters are known to be deposited on landfills each year. The analytical data to date do not suggest that the metals are leaving the landfills in the leachate. Intuitively, one would believe that if any metals are coming out in the leachate, additions to the fills would increase throughput; however, the sludges in question are normally precipitations and it would appear that the conditions may not exist in the landfills, except for perhaps Braid Street, to resolubilize the metals.

7.1.3 COD - Chemical Oxygen Demand. The chemical oxygen demand of the leachate coming from the large active municipal landfills is estimated to be 4800 kg/day which amounts to about 2% of the COD load to the Fraser River from all sources in the study area. There is no information presently available as to the specific make-up of the organics of the leachate from the landfills in the Fraser River Estuary. It can be expected that almost all the organics will be soluble or colloidal, as generally greater than 95% of the total solids of leachate are filterable (EPS, 1979; Cameron, 1978b). If the Richmond landfill leachate is typical, then it can be expected that about 90% of the COD of the leachate can be removed with three to ten days of aeration, leaving some 10% non-degradable, or at least not readily biodegradable (Lee, 1978).

Chian (1977) analyzed the organic composition of the leachate from a number of landfills in the United States. He found that the largest organic group was the free volatile fatty acids and that they decrease rapidly as the age of a landfill increased. He also found that the most stable group of organics with increasing age of a fill were fulvic-like materials, with a relatively high percentage of carboxyl and aromatic hydroxyl group compounds. An analysis of leachate from a lysimeter in two-month old refuse showed that 73% of the leachate's total organic carbon (TOC) would pass a 500 molecular weight ultra-filtration (MW-UF) membrane, with 78% of that material being free volatile fatty acids. An analysis of leachate from a 13-year old fill showed some 94% of the leachate TOC passing a 500 MW-UF membrane, with the concentration of fatty acids being below detection. Chian concluded that most of the organics from this 13-year old site must be low molecular weight refractory compounds.

In a study of leachate in groundwater under one landfill in the United States, Robertson et al (1974) were able to identify some 41 common industrial organics. The 41 compounds represented some 10% of the total weight of the organics in the groundwater.

It is unlikely that the findings from either of the above two studies are directly comparable to the leachates entering the Fraser River as surface discharges, although they are likely indicative of the types of organics that may be present in the leachates.

At this time, about 3% of the COD in the Annacis Island STP is estimated to be contributed to by leachate from the Braid Street (Coquitlam) landfill. By 1980, it is anticipated that about 5% of the COD in the Annacis discharge will be due to leachate, as the leachates from the Burns Bog and Port Mann landfills are diverted to sewer.

7.1.4 Ammonia and Alkalinity. The estimated load of total ammonia entering the Fraser River Estuary in the leachates from the large active landfills is 475 kg/day, about 7.6% of the total ammonia load from all the landfills, stormwater discharges and effluent sources. At the present time, the leachate from the Braid Street Landfill accounts for about 1.5% of the Annacis Island STP ammonia load.

By late 1980, it is estimated that about 16% of the ammonia load in the Annacis discharge will be present as a result of leachate addition. The net effect of the leachate diversions from the Burns Bog and Port Mann landfills will be to increase the concentration of un-ionized ammonia in the Annacis discharge by as much as 14% on average, which may affect the toxicity of the Annacis Island STP discharge.

While the pH and alkalinity of the leachate are higher than the sewage at Annacis, it is unlikely that the overall pH or alkalinity of the Annacis discharge will rise measurably as a result of the leachate addition. Consequently, the ratio of un-ionized ammonia to total ammonia will not likely change in the Annacis discharge.

7.1.5 Toxicity. The fifth parameter, toxicity, is considered to be one of the most important measurements of leachate, in that it allows for some assessment of the consequence of discharging a complex waste like leachate (Cameron, 1978a). Within the study area, the leachates at the point of discharge to receiving waters, typically have a 96-hour

LC_{50} of 24 to 50%, whereas, the leachates from springs have a 96-hour LC_{50} of 6.5% to 18%. A summary of the available fish toxicity information for the large active landfills is presented in Appendix F. To date, no specific parameters have been identified as the source of the toxicity in the leachate. Most people believe than ammonia is responsible for a significant portion of leachate toxicity. Cameron (1978b) carried out regression analyses on lysimeter leachate using 96-hour LC_{50} data on fish, 48-hour Daphnia and ROB* toxicity data, and the analysis for some 30 chemical parameters. It was found that COD gave the best single parameter correlation for fish, and zinc for daphnia; but, that the best overall correlations were equations relating hydrogen un-ionized ammonia, tannin and copper concentrations to 96-hour LC_{50} toxicity, and zinc and tannin concentrations to daphnia toxicity.

A direct extrapolation of Cameron's findings to the Lower Fraser landfill leachates is not considered valid in the absence of the elevated metal concentrations and the existence of pH values greater than 7, which are atypical for lysimeter leachate.

7.2 Large Municipal Landfills - Closed

There are three landfills in this group: Kerr Road, Stride Avenue and Terra Nova. Of these three, Kerr Road and Stride Avenue were closed prior to the emplacement of objectives under the Pollution Control Act. Accordingly, the leachate discharges from these landfills were never included in the provincial regulatory process. The Terra Nova landfill was under permit during the latter stages of its filling operation; the permit expired on December 31, 1975. While the Terra Nova landfill has been closed to municipal refuse since late 1975, it has been and continues to be utilized for the storage of wood waste.

In these cases where refuse permits were never issued or have since been cancelled, a continuing control function for the leachate discharges is not in place and may be difficult to enforce.

---

* ROB - Residual Oxygen Bioassay

7.2.1 Loadings. The Kerr Road and Stride Avenue landfill sites were atypical in that they were gully fills; whereas, the Terra Nova site is comparable to the active operations of the Braid Street, Richmond and Burns Bog landfills. The average constituent concentrations in the leachate for the three closed sites (Table 7.4), show a general trend of higher concentrations from the two gully sites than from the Terra Nova site. On the other hand, a comparison of loadings for six constituents and the percent constituent yield per tonne of refuse placed (Table 7.5), suggests that mass loadings do not follow that trend. From the totals of the percent yields, there would appear to be a relationship between fill age and the percent yield, which is consistent with leachate decay models. The values for the loadings and percent constituent yields cannot be considered absolute. However, the methods of calculation were comparable; therefore, the numbers should be reasonable for comparative purposes.

It can be expected that leachate will emanate from these landfills for a considerable period of time. Presently, the yields are about 0.7% to 0.6% of the yield of the active sites for the same five parameters (COD, chloride, ammonia, iron and manganese); whereas, the mass loadings from the closed sites are, in total, about 3% of those from the active sites. Table 7.6 compares the total daily loads for the large municipal active and closed sites.

In comparison to the loadings to the Fraser River from the active sites, sewage treatment plants, or storm sewers, the loadings from the three closed sites would not appear to be significant. With the exception of COD, ammonia, and possibly iron, the constituent concentrations are exceeded by many, if not most, groundwaters (Davis et al, 1966). The trace metal loadings including iron, amount to less than 9 kg/day in total.

With the exception of iron, manganese and sometimes zinc, the concentrations of the trace metals were generally below detection. Possible reasons for the low metal concentrations were discussed in Section 7.1.2 and are not unexpected considering the higher pH's and alkalinity of the closed sites.

TABLE 7.4 AVERAGE LEACHATE CONSTITUENT CONCENTRATIONS FOR LARGE MUNICIPAL CLOSED SITES

Parameter	Kerr Road	Stride Avenue	Terra Nova (Popeye Creek)
pH (units)	7.9	7.8	7
COD	125	48	49
Chloride	100	43	195
Total Phosphate	.32	.09	.11
Ammonia	58.2	14.8	3.25
Iron	3.0	1.65	Not Available
Calcium	63.7	75	52
Magnesium	41.6	22.2	92
Manganese	.37	1.60	.57
Specific	1900	810	334

Units in mg/l unless otherwise noted.

TABLE 7.5 COMPARISON OF LOADING AND PERCENT (%) YIELDS FOR SIX LEACHATE CONSTITUENTS AT THE LARGE MUNICIPAL CLOSED LANDFILL SITES

Parameter	Kerr Road (2x10 tonnes)		Stride Avenue (.526x10 tonnes)		Terra Nova (.815x10 tonnes)	
	Loadings (kg/yr)	Yields (%/yr)	Loadings (kg/yr)	Yields (%/yr)	Loadings (kg/yr)	Yields (%/yr)
COD*	14 900/2	.0007	6400/2	.0012	19 000/2	.0012
Chloride	11 900	.0006	5700	.0011	8 500	.001
Total Phosphate	38	.000002	12	.000002	55	.000006
Ammonia	6 900	.0003	2000	.0004	1 060	.0001
Iron	1 100	.00006	600	.0001	475	.00006
Manganese	44	.000002	213	.00004	219	.00003
TOTAL		.0013		.0022		.0033

* Yield calculated as kg of organics at 1/2 of COD.

TABLE 7.6 COMPARISON OF CONSTITUENT LOADINGS IN LEACHATE FROM LARGE MUNICIPAL LANDFILLS, ACTIVE AND CLOSED (kg/day)

Parameter	Active	Closed
COD	4800	110
Chloride	2285	72
Ammonia	523	27
Iron	153	6
Manganese	28	1.3

550 : 90
611 : 100

The organic load from the closed landfills, as represented by the COD, is less than 0.05% of the total load to the Fraser Estuary. However, these organics are likely biologically much more stable than most organics discharged in the study area, if any conclusions can be drawn from Chian's (1977) work.

Lack of good flow data for the leachate discharges leaves the loading calculations suspect. This is most significant for the Stride Avenue landfill, which is influenced by storm water flows from above the site.

7.2.2 Toxicity. Limited toxicity data are available for the Kerr Road and Stride Avenue leachate discharges, but no toxicity information is available for the Terra Nova landfill.

Two bioassays using rainbow trout as test species were conducted on the leachate collected below Marine Drive at the Kerr Road landfill. The analysis gave a 96-hour LC_{50} of 24% and an LT_{50} at 100% concentration of 34 minutes (EPS, 1979).

Three bioassays using rainbow trout gave 96-hour LC_{50} results of greater than 90% non-toxic and 56% for the Stride Avenue leachate discharge at Marine Drive (EPS, 1979).

7.3 Wood Waste Landfills

Some 2 458 000 m^3 of wood waste were landfilled in the study area during 1977 (derived from data in Appleby, 1978), and it is estimated that more than 4 350 000 m^3 of wood waste are in fills along the Fraser River (EPS, 1978).

An estimated 5000 kg/day to 14 800 kg/day of COD enters the Fraser River Estuary in the leachate from the wood waste landfills; this amounts to from 2% to 5.5% of the total COD load to the estuary. Loading estimates of COD, total solids, nitrogen, phosphorous, iron, manganese and copper are presented in Table 7.7. While there was some limited qualitative data for wood waste leachate, there was no quantitative data. The loadings are estimates, based hopefully on sound assumptions, but at best can only be considered as estimates. In total, the COD load from

TABLE 7.7 SUMMARY OF DAILY LOADINGS FROM WOOD WASTE LEACHATES WITHIN THE STUDY AREA

Parameter	Loading/day (kg/day)
COD (chemical oxygen demand)	14800
Total Solids	9860
Total Nitrogen	24
Total Phosphorous	4
Copper	0.4
Iron	91
Manganese	11

the wood waste leachate is only 16% to 39% of the COD in the Annacis discharge. However, what is important to realize about the COD load from the wood waste landfills is that a portion is exerted in small tributaries rather than in the main stem of the Fraser River.

The organic make-up of leachate from wood waste is not likely to be as varied as that from refuse; however, it is a complex mix of at least lignins, resin acids and phenolic compounds. The acute toxicity of wood waste leachate has been demonstrated innumerable times in laboratories and in the field; at its worst, a 96-hour LC_{50} of 0.48% was measured in the laboratory and 8% in the field (Cameron, 1978b; and EPS, 1977). A summary of this toxicity data is presented in Appendix F. The quite general and rapid decay of acute toxicity has also been measured, and it was found that wood waste leachate becomes non-acutely toxic normally within 12 months to 4 years following the placement of fresh wood waste. However, this is not always the case, as is demonstrated by the one fill adjacent to the Fraser River which has been inactive since 1973, but still emits an acutely toxic leachate as of 1978 (EPS, 1978).

7.4 Small Municipal Landfills and Miscellaneous Refuse Dumps

One active and seven closed, small municipal landfills were documented. All the landfills are adjacent to small watercourses and, while leachate is believed to have entered most, data to suggest an impact on water quality were available only for the Bear Creek landfill. There is also some suggestion that leachates could be a source of trace metal build-up in the sediments adjacent to such discharges.

7.5 Impact on the Lower Fraser River and Estuary

7.5.1 Large Municipal Landfills - Active. Several qualities of a landfill leachate categorize it as an effluent having a potential for imparting an undesirable impact on the aquatic environment and its biota. The complex mixture of chemical constituents within leachates imparts an acutely toxic effect on fish. It is impossible to isolate any single parameter as the toxic fraction, but any one among ammonia (un-ionized), trace metals, tannins or COD can, by itself, be toxic. Organic contaminants which are not normally monitored in the analysis of leachates, could be present and could contribute to the acute toxicity, or they may have long-term implications through bio-accumulation. While specific organics and chlorinated organics have been identified in other leachates (Chien, 1977; Robertson et al, 1974), any suggested impact on the Fraser Estuary from such compounds is by inference only, due to an absence of data.

McBride et al, 1977, recorded sublethal effects in rainbow trout resulting from exposure to leachates. While the effects were noted at dilutions as great as 200:1, the fish appeared to be able to adapt; this was not the case at 20:1 dilutions. The leachate used in the studies was from a spring on the Burns Bog Landfill.

Very little data exists on the occurrence of trace metals or the organic contaminant accumulation in sediments of streams and rivers adjacent to landfills. The data that are available do suggest a possible accumulation of lead and zinc in the Fraser River adjacent to the Richmond Landfill, No. 8 Road ditch (Soper et al, 1977). No data exist to indicate the accumulation, or lack of accumulation, of metals or organic contaminants in the benthic biota of receiving streams under the influence of leachate. The presence of trace metals and possibly organic contaminants in leachates suggests that leachates are one source capable of contributing to the build-up of contaminants through the trophic levels in the Fraser Estuary. The significance of the contribution should be viewed in conjunction with that from other sources.

The Fraser River Estuary is an important zone for anadromous fish species making the change from fresh to salt water. Any discharges into these waters that could cause stress at low levels could affect transition capability.

To date, the toxicity and sub-acute toxicity work on leachate has been carried out using rainbow trout of 2 g and 30 g sizes, respectively. These may not be the most sensitive test animals. The influence of fish size and life history is important when dealing with acute and sub-acute toxicity. While specific data are not available for leachates, examples of selected toxicants from the literature are presented in Appendix H to illustrate the point.

Over the past few years, efforts have been made to control the surface leachate discharges emanating from the large active municipal landfills in the Lower Fraser Valley. Culmination of those efforts will likely see the diversion of leachates from the Burns Bog, Braid Street and Port Mann landfills to the Annacis Island STP, and the on-site control and treatment of leachate from the Richmond Landfill.

The diversion of leachate to Annacis Island STP requires some discussion. The benefits of sewering the leachates should be assessed against the additional impact that the leachate could cause by being discharged through the Annacis Island STP.

There are two aspects to the leachate diversion: increased toxicity as a consequence of ammonia content of the leachate; and production of chlorinated organics as a result of chlorinating the effluent from the Annacis Island STP.

Implicit in the diversion of leachate to Annacis, will be an increase in the concentration of un-ionized ammonia which may affect the toxicity of the Annacis Island STP effluent. As one of the toxic components of the present Annacis discharge is believed to be ammonia, then the addition of leachate will most likely have a detrimental effect on the entire Annacis discharge. Whether such an increase in toxicity of the effluent will result in a proportional increase of toxicity in the immediate mixing zone is not known, but one would expect some effect. It may be reasonable to argue that for chlorinated effluents, un-ionized ammonia may not be a causative toxin due to depressed pH following chlorination-dechlorination. Therefore, specific concerns related to leachate ammonia may be restricted to winter months, when chlorination is not practised. If the toxicity of the Annacis discharge is specifically related to heavy metals or pH effects, then the addition of the leachate will not likely increase the toxicity of the effluent. It may be conjectured that the increase in ammonia concentration could increase the solubility of heavy metals in the effluent through complex formation.

Consideration of the second point, that of the formation of chlorinated organics as a result of disinfecting a mixed effluent at Annacis, has implications that should not be restricted solely to leachate. The chlorination of any organic waste will result in the formation of some chlorinated organics. Jolley et al (1978) found that from 0.5% to 3.0% of the chlorine, applied either for disinfection or for anti-fouling, reacted with organics to form chloro-organic compounds. The yield of chloro-organics was lower in the presence of ammonia (Jolley et al, 1978; Pierce, 1978); however, that effect was offset to some degree with increasing organic content (Jolley et al, 1978).

The reaction of aqueous chlorine compounds with organics results in a myriad of chlorine contained compounds. In order to illustrate this point, several paragraphs of the summary section from The National Research Council Associate Committee on Scientific Criteria for Environmental Quality publication, "The Aqueous Chlorination of Organic Compounds: Chemical Reactivity and Effects on Environmental Quality" (Pierce, 1978), are quoted verbatim and presented in Appendix G.

Chlorination of the mix of organics in the leachate will in all likelihood result in the formation of some chloro-organics and chloro nitrogen-containing organics. The peat bog setting of most of the landfills in the study area will result in concentrations of humic material in the leachate and, if any correlations can be drawn from Chian's (1977) work, low molecular stable organics, free amino acids and alcohols will also be present.

Diversion of leachate to the Annacis Island STP will likely alter the mix of chlorinated organics in the effluent, as a result of different precursors. It does not follow, however, that the total concentration of chlorinated organics will increase in the effluent as a result of the leachate addition, due to the effect of ammonia, as reported by Jolley (et al), 1978.

Whether or not the effects from either the increased ammonia or chlorination of leachate organics will result in impacts in the Fraser Estuary can not be stated.

If the mode of treatment at the Annacis Island STP remains primary, there is a range of opinion as to the benefit to be gained, over a raw discharge, by the diversion of leachate into the sewer. On this point, there are a number of advantages and disadvantages to be weighed. The essence of these are presented for the reader's consideration.

The advantages of diverting leachate to the Annacis Island STP are seen to be:

1. The impact that the leachates have on the local receiving waters is removed. (In the case of Burns Bog leachate, the receiving waters are the municipal ditch system and Crescent Slough. In the case of Port Mann, the leachate flows onto the foreshore of the main stem of the Fraser.)
2. The leachate could be renovated to some degree in passing through the sewers and the primary treatment plant.
3. There would be a reduction in the number of individual discharges to estuary waters.

The disadvantages in diverting leachate to the Annacis Island STP are seen to be:

1. There may be an increase in the toxicity of the Annacis Island STP effluent.
2. A higher loading will be discharged to the river at one point, thereby exerting a greater demand on the river's assimilative capacity.
3. A wider spectrum of organics would be exposed to chlorination-dechlorination than would otherwise be the case.

If, on the other hand, biological stabilization becomes the treatment mode at the Annacis Island STP, then some benefits become apparent over a raw discharge, such as reductions in toxicity, BOD, COD and metals. Whether or not there is a reduction in the potential for the production of diverse chlorinated organics is not known, since many of the precursors relative to secondary treatment are biologically stable. On the other hand, Swedish research on the mutagenic effect of chlorinated pulp wastes shows that mutagenic effects can be reduced with biological treatment (Erickson et al, 1979), which suggests that some such chlorinated organics are degradable.

7.5.2 Large Municipal Landfills - Closed. There has been no documentation of impacts resulting from the leachates produced at the three large closed sites at Kerr Road, Stride Avenue and Terra Nova.

The Kerr Road Landfill discharge is the least apparent, as it enters the North Arm of the Fraser River at the MacMillan Bloedel/Canada White Pine complex.

The discharge from the Stride Avenue Landfill is the most accessible, flowing through and alongside a number of truck farms as it enters the general drainage of the Burnaby Flats. It is understood that a number of farms are irrigated from the drainage ditch and, in that context, specific ion concentration in the leachate could be of concern, if irrigation is carried out over a long period of time. For example, the maximum recommended level of manganese for long-term irrigation is 0.2 mg/l (EPA, 1973), whereas, the average concentration in the leachate at Marine Drive is 1.60 mg/l. These concerns should be reviewed in terms of specific soil types, crops and farming practices in the area.

The discharge from the Terra Nova Landfill can be seen at certain river stages as a characteristic black plume where it enters the Fraser River. Changes as a result of leachate going into Mill Creek cannot be readily seen as the water quality of the creek has been affected by other upstream activities.

The organic content of the leachate from the three closed sites is in all likelihood biologically quite stable. Quantitatively, the leachates are not a significant source of organics; however, the nature of the organics may prove to be of some concern and require study.

The continuing storage of wood waste at the Terra Nova Landfill and the subsequent generation of wood waste leachate may necessitate some management in line with other wood waste landfills.

7.5.3 Wood Waste Landfills. Leachate from wood waste landfills has not been seen to cause an impact in the main stem of the Fraser River. On the other hand, wood waste leachate continues to have a significant impact on small tributary streams and drainage waters throughout much of the Fraser River Estuary.

Examples of direct physical loss of Fraser River foreshore due to wood waste piles and fills can be provided; however, these losses are rather more symptomatic of industrial encroachment than the landfilling by wood waste. Direct loss of foreshore habitat due to wood waste leachate is not well documented; however, the foreshore shown in Plate 7.1 is one example.

It is on a number of small tributaries in the Lower Fraser River that the impact of wood waste becomes most apparent. The location of four such tributaries: Scott Creek, School House Creek, an unnamed drainage at Scott Road, and an unnamed drainage at the northeast corner of Burns Bog, are shown in Figure 7.1.

The leachate flowing into School House Creek had, on April 10, 1978, an LT_{50} at 100% of 5 minutes. On May 18, 1978, a bioassay was conducted on the stream water itself and all the fish died within 16 hours. The 96-hour LC_{50} (July 19, 1978) for the drainage water coming from the Burns Bog area (not the City of Vancouver's Landfill) after the addition of wood waste leachate was 13.5%; while on April 10, 1979, the LT_{50} was 14 minutes.

Leachate flowing into the large drainage channel at Scott and River roads had, on April 10, 1979, an LT_{50} of less than 5 minutes, and on April 24, 1979, a 96-hour LC_{50} of 8%. The fact that the channel had been lost to fish long before the present day wood waste leachate addition, is not considered valid justification for its present use. During April, 1979, it was not possible to find an inflow to the channel in the Scott and River road area that did not contain wood waste leachate.

Scott Creek, a small stream feeding into the Coquitlam River, has occasionally been black with leachate. A bioassay conducted on the

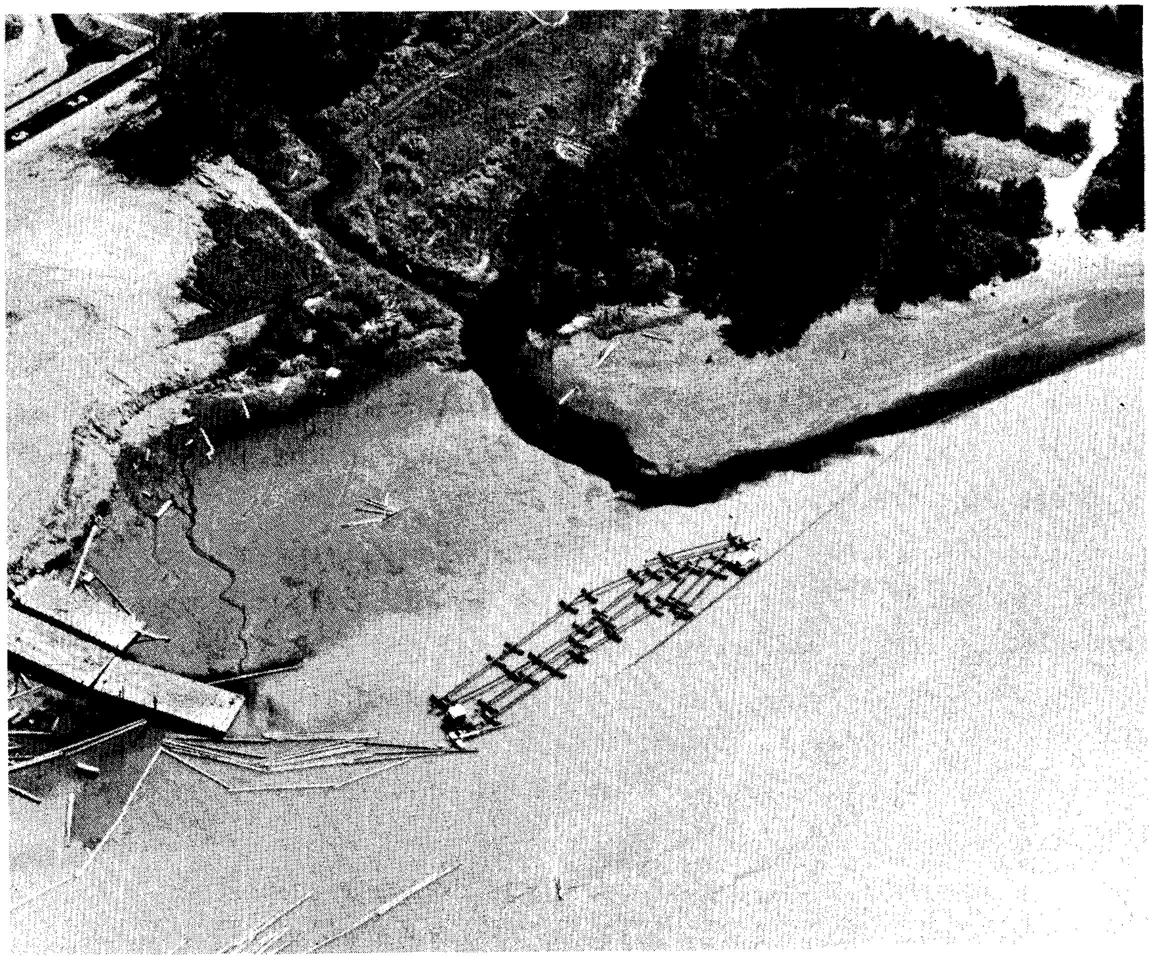


PLATE 7.1 FRASER RIVER FORESHORE HABITAT LOSS DUE TO WOOD WASTE LEACHATE

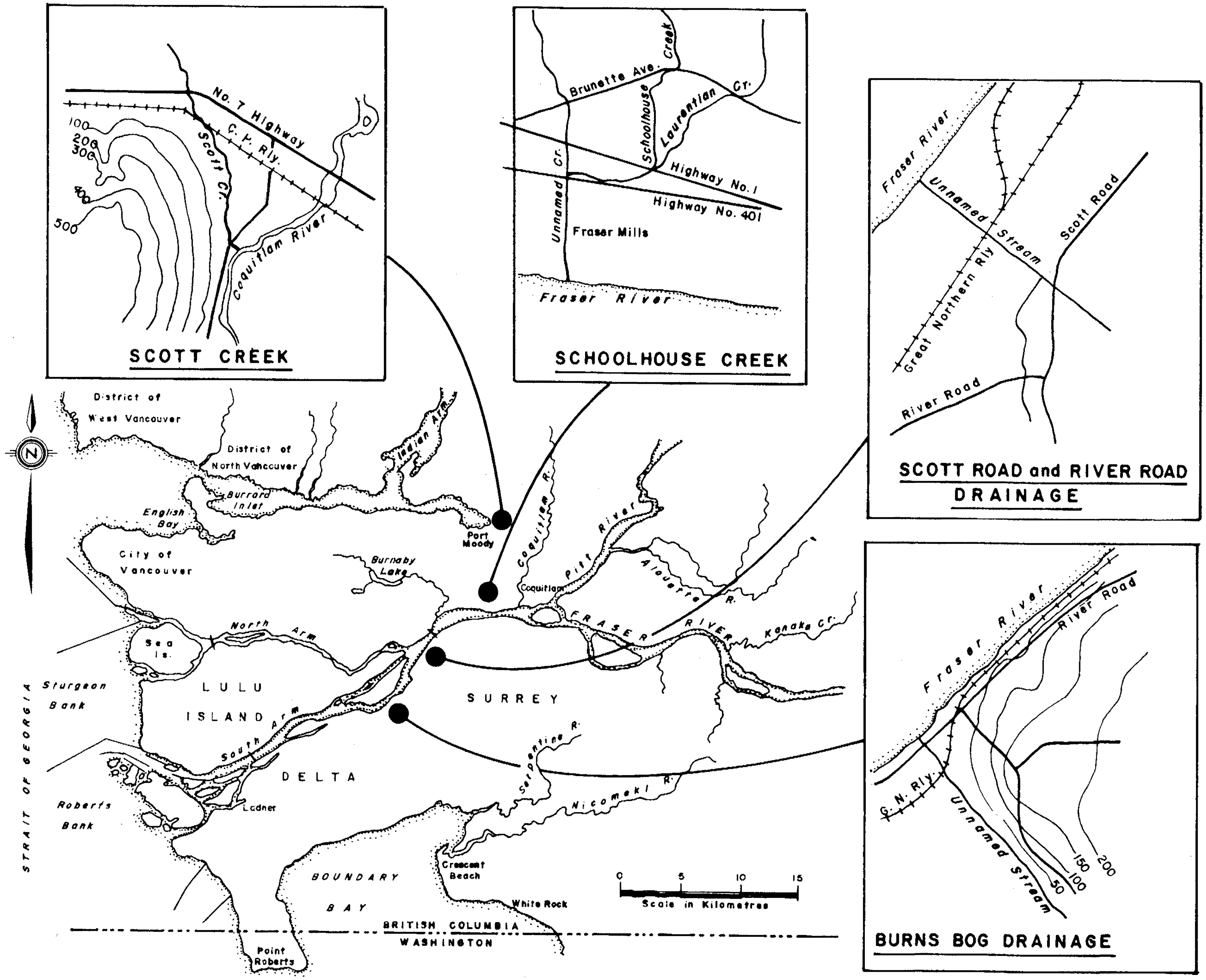


FIGURE 7·1 WOOD WASTE LEACHATE - LOCATIONS OF TRIBUTARY DRAINAGE leachate entering Scott Creek (April 6, 1977) had a 96-hour LC50 of 10%. Successful efforts in recent months have been taken to route the leachate away from Scott Creek. In some, but certainly not in all cases, the discharge of leachate to small receiving waters has resulted in the formation of a slime and/or fungal growth, and has sometimes caused colouring of the creeks due to the iron content (Thomas, 1977).

The discharge of wood waste leachate, particularly to small receiving waters, is covered under the Pollution Control Objectives for the Forest Products Industry of British Columbia. Those objectives specify that receiving waters may show only a negligible increase of wood waste leachate. At the same time, Section 33 of the Fisheries Act is quite specific regarding the discharge of deleterious materials.

The practicalities of such a situation are often such that enforcement of the letter of the law becomes difficult. However, it is fair to say that the control and treatment of wood waste leachate has not been a consideration in the location of wood waste landfills nor in the use of wood waste as a fill in the Fraser River Estuary. As a result, wood waste leachate has emanated from landfill sites without restriction and has been allowed to flow into the existing drainage courses.

As the landfilling of wood waste is widespread throughout the Fraser River Estuary, it is reasonable to expect that the water quality of many small tributaries will decrease unless specific steps are taken to minimize the impact of wood waste leachate. Wood waste leachate problems are not unique to the Lower Mainland.

Hogfuel leachates draining to swamp areas or small drainage ditches, such as those draining into the Fraser Estuary, have been reported to cause discolouration, reduced dissolved oxygen, decreased pH, bacterial slime growth, and acute toxicity. Schermer (1974) reported that a drainage ditch to Mill Creek in Oregon which had received wood waste leachate had a characteristic greenish, black colour with high COD (500 mg/l) and a pH between 5 and 5.5. Dissolved oxygen in the ditch was less than 1 mg/l and the only apparent life was a greyish looking algae. Peters et al (1976) reported that a runoff stream from a two-year old cedar landfill in the Snohomish River area near Everett, Washington, was intensely coloured, had a pH of 4.3 and a BOD_5 of 715 mg/l. In this case, the median survival for 10 test fish was 10.4 hours at 10:1 dilution. In swampy areas containing fresh cedar debris on the Quinalt Indian Reservation, tropolones were found to be at the 0.05 mg/l level, but neither tropolones nor lignins were ever detected in water containing older wood debris, which suggested that the substances had degraded with time.

Bioassays of groundwater taken at various times from test wells down slope from a hogfuel dump contained by a dyke indicated that the water, after reaching the test well by percolation through both soil and gravel, was non-toxic (Schermer and Phipps, 1976). This finding substantiated the findings of fish toxicity studies by the same researchers on laboratory-generated leachate that had been passed through soil or gravel columns. They concluded that "forcing" wood waste leachate into the groundwater system resulted in an immediate reduction in the COD, tannins, and toxicity of the leachate. However, as the amount of leachate interaction with soil or gravel increased, the materials lost their ability to remove those substances.

Peters et al (1976) also discovered that interaction with soil and soil micro-organisms apparently metabolized or bound the substances by contact, thus changing the leachates to a non-toxic form. It was also found that tropolone toxicity could be eliminated by chelatable iron.

The apparent ability of soil to remove wood waste leachate toxicity coupled with the short time (1 to 4 years) that wood waste landfills normally generate toxic leachates, suggests that wood waste could be landfilled in such a way that small tributaries would not become degraded. The disposal of wood waste could be managed to ensure that leachate is either diverted away from small tributaries for the period of time the leachate is acutely toxic, or that the leachate is renovated prior to entering small tributaries.

7.5.4 Small Municipal Landfills and Miscellaneous Refuse Dumps. With the exception of some visual evidence of iron staining and fungal build-up immediately adjacent to several small fills, there is little documentation of any impacts from leachates in this group, other than for Bear Creek.

Watkins (1970) reported the existence of abundant fungus growth in Bear Creek adjacent to the fill area. He also noted an abundance of coho fry in the stream upstream of the fill site and extending about 180 m downstream into the fill area. One hundred and eighty metres further downstream, very few fry were present except for small groups in the deeper pools. Fry were again seen in abundance immediately downstream of the fill area. Watkins (1970) suggested that the absence of fry in the area of the fill reflects the avoidance of the area by fry, thus making it useless as a rearing area.

In comparison with the other effluent sources it is doubtful if the leachates from this group of landfills will affect the Fraser River Estuary beyond the immediate receiving waters.

In summary, it is estimated that the leachates coming from all the landfills in the study area, contain between 4% and 8.7% of the COD, 6-7% of the ammonia, 9% of the iron and 1.9% of the zinc entering the lower Fraser each day from all effluent and storm water sources. The principal impacts from these leachates will occur in two forms: increased ammonia levels at the Annacis Island STP from leachate additions which may affect the toxicity of the total discharge; and, degradation of a number of small tributary waters, as a result of wood waste leachate.

REFERENCES

A.E.S.L. 1974. Leachate Study - Port Mann Sanitary Landfill. Report prepared for the District of Surrey by Associated Engin. Ser. Ltd.

Appleby, P.W. 1978. Hog Fuel Availability Study South Central Region of B.C. Report prepared for the British Columbia Wood Waste Energy Coordinating Committee.

Atwater, J.W. 1978. Personal information

B.C. Research. 1979. Effects of Leachate from Richmond Landfill on Water Quality in the Fraser River. Unpublished information.

Benedict, A.H., Hall, K.J. and F.A. Koch. 1973. Preliminary Water Quality Survey of the Lower Fraser River System. Westwater Research Center, University of British Columbia, Technical Report, No: 2.

Bird and Hale Ltd. 1978. Municipal Refuse: Statistics for Canadian Communities of over 100,000 (1976-1977). A report prepared for the Waste Management Branch, Environmental Protection Service.

Boyle, W.C. and R.K. Ham. 1974. Biological Treatability of Landfill Leachate. Journal of the Water Pollution Control Federation, Vol. 46, pp. 860-872.

Bjorkman, V.B. 1979. Physical-Chemical Treatment and Disinfection of Landfill Leachate. Unpublished Masters thesis, University of British Columbia.

Burkhalter, D.W. and C.M. Kaya. 1977. Effects of Prolonged Exposure to Ammonia on Fertilized Eggs and Sac Fry of Rainbow Trout (Salmo gairdneri), Transcripts of the American Fisheries Society, Vol. 106, No. 5.

Cameron, R.D. 1975(a). Investigation of Leaching from Simulated Landfills, University of British Columbia.

Cameron, R.D. 1975(b). A Comparison of the Standard TL_m Bioassay with the Rapid Toxicity Assessment for Municipal Refuse and Hog Fuel Leachates and Solid Wastes; Technical Report No. 6 prepared for the Government of British Columbia, Department of Lands, Forests and Water Resources by the University of British Columbia, Faculty of Applied Sciences.

Cameron, R.D. 1977. Leachate Treatment - Richmond Landfill, in the Richmond Landfill Assessment Study Report EPS-8-PR-77-2 (Appendix K), Environmental Protection Service, Vancouver.

Cameron, R.D. 1978(a). Unpublished report to the Environmental Protection Service, Vancouver.

Cameron, R.D. 1978(b). Toxicity of Landfill Leachates, University of British Columbia.

Chian, E.S.K. 1977. Stability of Organic Matter in Landfill Leachates; Water Research, Vol. 11, p. 215.

City of Vancouver. 1973. A brief submitted to the Public Inquiry into Municipal Type Waste Discharges on Land.

City of Vancouver. 1974. Unpublished data.

City of Vancouver. 1975(a). Unpublished information.

City of Vancouver. 1975(b). Evaluation of City of Vancouver, Burns Bog Sanitary Landfill, submitted to the British Columbia Pollution Control Branch.

City of Vancouver. 1976. Unpublished data.

City of Vancouver. 1979. Unpublished information.

Clark, M.J.R. 1975. B.C. Water Resources Service Discharge & Environmental Data Base (EQUIS). British Columbia Pollution Control Branch, Ministry of Environment, Report No. 75-9.

Canadian National Railway. 1976. Unpublished information.

Conestago - Rovers and Associates and Waterloo Engineering Associates Inc. 1978. Recommended Landfill Leachate Methodologies. Draft report of an April, 1978, conference at Halifax, Nova Scotia, report prepared for Fisheries and Environment Canada, Waste Management Branch, Ottawa.

Cook, P.M. 1963. Soil Report on Lots 1, 2, & 3, Block 31, DL 20 and Lots 4 & 5, Block 30, DL 66, NWLD (Leeder Pacific Ltd. Property, Coquitlam B.C.).

Corbett, J.R.E. 1975. Treatment of a Low Strength Landfill Leachate with Peat. Thesis Submitted in Partial Fulfillment of M.Sc. Department of Civil Engineering, University of B.C.

Davis, S.N. and R.J.M. DeWiest. 1966. Hydrogeology, published by John Wiley and Sons.

District of Surrey. 1972. Unpublished data from Surrey P.C.B. files.

District of Surrey. 1978. Port Mann Sanitary Landfill, report prepared by Engineering Department.

Econotech Services Ltd. 1977. Consequences of Leaching from Pulp & Paper Mill Landfill Operations, CPAR Project Report 363-2, Prepared for Fisheries and Environment Canada. Distributed by CPAR Secretariat, Environmental Protection Service Ottawa, Ontario.

Ellis, J.D. and M.J.R. Clark. 1977. Computer System For Biological Information; British Columbia Pollution Control Branch, Ministry of Environment, Report No. 77-4.

Environmental Protection Agency(EPA). 1973. Water Quality Criteria 1972. A report prepared by the National Academy of Sciences, Ecological Research Series, U.S. EPA Report No: R3-73-033, Washington, D.C.

Environmental Protection Service. 1976. Unpublished information.

Environmental Protection Service. 1977. Unpublished information.

Environmental Protection Service. 1978. Unpublished information.

Environmental Protection Service. 1979. Unpublished information.

Eriksson, K.E., Kolar, M. and K. Krinstad. 1979. Studies on the Mutagenic Properties of Bleaching Effluents, Scientist Papperstudnig No. 4.

Equis. 1978. Equis Printout 1974-1975 Data, British Columbia Pollution Control Board.

E.V.S. Consultants. 1975. Leachate Toxicity Measurements Using the Residual Oxygen Bioassay Method. A Report Prepared for Environmental Protection Service, Pacific Region.

Fisheries and Marine Service. 1976. Unpublished data.

Golder Associates. 1977. Leachate Containment Investigations - Richmond Landfill, a report prepared for the Department of Public Works.

Great Vancouver Regional District. 1972. Data collected to support a brief to the British Columbia Pollution Control Board.

Great Vancouver Regional District. 1973. A brief submitted to the Public Enquiry on Municipal Waste Disposal, Vol. 1.

Great Vancouver Regional District. 1975. Unpublished data.

Greater Vancouver Regional District. 1976. Unpublished data.

Greater Vancouver Regional District. 1978. Reports on Quality of Drainage Water - Coquitlam Landfill.

Greater Vancouver Sewerage and Drainage District. 1976. Request for Proposals for Solid Waste Disposal Services.

Hoos, L.M. and G.A. Packman. 1974. The Fraser River Estuary Status of Environmental Knowledge to 1974. Special Estuary Series No. 1, Regional Board, Pacific Region, Environment Canada.

Institute of Forest Products (I.F.P.). 1957. Conversion Factors for the Pacific North West Wood Products. Seattle, Washington.

Jackiman, et al. 1979. Unpublished progress report, University of British Columbia.

Jolley, R.L., G. Jones, W.W. Pitt and J.E. Thompson. 1978. Chlorination of Organics in Cooling Water and Process Effluents. Water Chlorination - Environmental Impact Health Effects, Vol. 1, Proceedings of the Conference at the Environmental Impacts of Water Chlorination, Oct. 22-24, 1975. Editor, R.J. Jolley. Ann Arbour Science.

Knox, K. and P.H. Jones. 1979. Complexation Characteristics of Sanitary Landfill Leachate. Water Research, Vol. 11, pp. 819-846.

Kaller, J.J. 1970. Report on Solid Waste Management. Report prepared for the Greater Vancouver Regional District

Kelly, H.G. 1971. Pollutants From Refuse Dumps. Major Essay Submitted in Partial Fulfillment of M.Sc. Department of Civil Engineering, U.B.C.

Lee, C.J. 1979. Treatment of a Municipal Landfill Leachate. Department of Civil Engineering, University of British Columbia.

Levelton, B.H. 1971. Analysis of Effluents from Landfill. Report prepared for Terra Nova Development Ltd.

McBride, J.R., Donaldson, E.M. and G. Derksen. 1979. Toxicity of Landfill Leachate to Underyearling Rainbow Trout (Salmo gairdnei). Bulletin of Environmental Contaminants and Toxicity, Vol. 23, pp. 806-813.

McKee, J.E. and H.W. Wolf. 1963. Water Quality Criteria. The Resources Agency of California.

Peters, G.B., Dawson, H., Hrutfiord, B.F. and R.P. Whitney. 1976. The Effects of Aqueous Leachate from Western Red Cedar on Some Aquatic Organisms. Journal of the Fisheries Research Board Canada, Vol. 33, pp. 2703-2709.

Pierce, R.C. 1978. The Aqueous Chlorination of Organic Compounds: Chemical Reactivity and Effects on Environmental Quality. National Research Council Associate Committee on Scientific Criteria for Environmental Quality.

Province of British Columbia. 1975. Pollution Control Objectives for Municipal Type Waste Discharges in British Columbia.

Province of British Columbia Pollution Control Board. 1977. Unpublished Water Quality report.

R.M. Hardy and Associates. 1978. Geotechnical Assessment of Stability and Leachate Control - Surrey Sanitary Landfill. Report prepared for the District of Surrey.

Robertson, J.M., Toussaint, C.R. and M.A. Jorque. 1974. Organic Compounds Entering Ground Water from a Landfill. U.S. Environmental Protection Agency. EPA-660/2-74-077.

Russo, R.C. et al. 1974. Acute Toxicity of Nitrite to Rainbow Trout (Salmo gairdneri). Journal of the Fisheries Research Board Canada, Vol. 31, pp. 1653-1665.

Schermer, E.D. and J.B. Phipps. 1976. Wood Waste Leachate: Its Interaction with Groundwater. Work Performed Under Contract 75-054, 76-046; Washington State Department of Ecology.

Schermer, E.D. and J.B. Phipps. 1974. A study of Wood Waste Leachate. Report prepared for the Washington State Department of Ecology.

Schreck, C.B. and H.W. Lorz. 1978. Stress Response of Coho Salmon (Oncorhynchus kisutch) Elicited by Cadmium and Copper and Potential Use of Cortisol as an Indicator of Stress. Journal of the Fisheries Research Board Canada, Vol. 35, pp. 1124-1129.

Sinley, J.R. et al. 1974. The Effects of Zinc on Rainbow Trout (Salmo gairdneri) in Hard and Soft Water. Bulletin of Environmental Contaminants and Toxicity. Vol. 12, No. 2.

Steelson, J. 1974. Proceedings Woodwaste Leachate Symposium, Grays Harbour College. Editor, J.B. Phipps.

Smith, C.E. and W.G. Williams. 1974. Experimental Nitrate Toxicity in Rainbow Trout and Chinook Salmon. Transcripts American Fisheries Society, Vol. 103, pp. 389-390.

Soper, P.W. and H.F. McAlpine. 1977. Richmond Landfill Assessment Study. Environmental Protection Service, Department of Fisheries and Environment, Vancouver.

Soper, P.W. 1978. Unpublished report to the Environmental Protection Service.

Thomas, P.R. 1977. Consequences of Leaching from Pulp and Paper Mill Landfill Operations. Report prepared for the Canadian Forestry Service, Environment Canada.

Thurston, R.V. et al. 1978. Acute Toxicity of Ammonia and Nitrate to Cutthroat Trout Fry. Transcripts American Fisheries Society, Vol. 107(2), pp. 361-368.

Unies Ltd. 1973. Surrey Bend Landfill - Preliminary Environmental and Cultural Assessment. Report prepared for the Greater Vancouver Regional District.

U.S. Department of Health, Education and Welfare. 1969. Development of Construction and Use Criteria for Sanitary Landfills, Cincinnati, Ohio.

Watkins, J.V. 1970. A Study of Leachates From Sanitary Fills and their Effects on Receiving Waters. Unpublished Report. Department of Fisheries and Forestry, Pacific Region.

Wilson, D.G. 1977. Ed. Handbook of Solid Waste Management. Van Nostrand.

APPENDIX A

LEACHATE TREATABILITY

1. BACKGROUND

This section on leachate treatability was taken from a paper by Dr. R. Cameron prepared at the request of EPS, Pacific Region. The paper drew heavily on Dr. Cameron's knowledge of the literature, his experience, and personal contacts. Most references to studies, if not all the references in this section, can be found in Cameron (1977), Chian et al (1977), and Conestago - Rovers (1978).

Studies of municipal refuse leachate treatability are limited in number as well as in field application. In the few cases where leachate treatment has been attempted on a pilot-or full-scale basis, the results have generally been much less satisfactory than the results from laboratory scale work. Some laboratory work has shown that reasonable effluent standards can be met with a relatively simple, single-stage system, while other work has indicated the need for highly sophisticated and costly advanced waste treatment systems. The previously mentioned variations in hydraulic loading coupled with dramatic changes in leachate concentrations with time, when added to the preceding factors, show that it is difficult if not impossible to prescribe a specific treatment method for a given leachate. The following discussion is therefore limited to an outline of the methods found to be reasonably successful and the limitations of the methods. While one might have to design a leachate treatment system based on information from the literature, proper design should be based on, at least, the results from a laboratory study.

2. BIOLOGICAL TREATMENT SYSTEMS

Leachate from a new or middle-aged landfill will usually be characterized by a relatively high BOD_5. Removal of organics in leachates from young sites can be achieved relatively inexpensively by using a biological treatment system. Anaerobic systems such as digesters or filters have shown good success in the laboratory where conditions have been controllable. Attempts at using anaerobic systems in the field, however, have met with little success, largely due to variations in hydraulic and organic loading. Relatively low treatment efficiency and high capital costs also tend to preclude the use of anaerobic digestion.

Aerobic treatment studies have shown that long detention times are necessary for leachate treatment. The use of activated sludge systems has therefore not met with success. Aerated lagoon type treatment has been successful, both in the laboratory and in the field. Generally, for leachate with BOD_5 values in the tens of thousands, detention times may range from 20 to 60 days. With lower BOD_5 values detention times of as few as three days have been found to be sufficient. With the higher influent BOD_5 values, effluent BOD may or may not meet effluent standards so that some form of polishing treatment may be necessary. Almost invariably, studies using leachate having BOD_5 values in the hundreds or low thousands have produced an effluent having a satisfactory BOD value. In these aerobic systems, the addition of phosphorus as a nutrient has almost invariably been required. Less commonly, nitrogen has also been necessary. At least initially, pH adjustment may be required for satisfactory growth of microorganisms.

The aerobic treatment of leachate can also significantly reduce toxic metal concentrations as well as the toxicity of the leachate. Especially with lower strength leachates, these factors may be reduced to the point where they can meet effluent quality standards. Again, with higher strength wastes polishing treatment may be required.

Leachates from very old landfills have to be looked at with care when determining the types of treatment necessary. As landfills are biological reactors, the degradable organics may be reduced to such an extent that biological treatment is either not feasible or is not cost effective. In such cases, the method of treatment selected will have to be based on the types of contaminants to be removed.

As with leachates from old landfills, the effluent from a first-stage biological treatment system will have to be examined with great care to determine exactly what has to be removed. Trace organics may be removed using ozonation, and suspended materials may be removed in a sand filter. Activated carbon will reduce metals and dissolved organics. Reverse osmosis or ion exchange can effectively remove dissolved metals. Various chemicals may be used to oxidize organics or to precipitate dissolved metals. Each of these systems has been used, at least in the laboratory, and each has met with some success. However each system has problems, not the least of which is relatively high cost.

While most studies have shown that the sludge produced in biological treatment systems settles well, some suspended material will often be present in the effluent. This suspended material will usually have to be removed because of the interference it can cause if steps such as ion exchange or reverse osmosis are anticipated. A sand filter is fairly cheap and is effective in removing suspended materials. It will, however, remove little else. An activated carbon column might be selected in place of a sand filter because it can remove the suspended organics and, under the right circumstances, produce an effluent sufficiently low in dissolved materials to meet effluent standards. In this situation, while the suspended material will reduce the carbon column efficiency for removal of dissolved material, the cost of the column may be less than that of a sand filter followed by reverse osmosis or ion exchange.

If effluent standards are sufficiently stringent, or if new leachate strength is very high, polishing of the effluent from a sand filter or a carbon column may be required. The common methods used are those of reverse osmosis and ion exchange. Reverse osmosis efficiency can be dramatically reduced if any suspended material is present. The presence of certain inorganics such as silica can also foul the membrane. Ion exchange units, if used in either the cation or anion exchange mode, may require pH adjustment of the effluent. Mixed beds may not need this, but they probably will be more costly because of the different degrees to which anions and cations will have to be removed. It must be remembered that ion exchange replaces a contaminant with another ion; therefore, the process cannot really be considered a removal method but is simply a replacement.

Each of the foregoing systems entails a level of effort greater than that indicated by a description of the process. Biological treatment systems may produce sludge volumes as great as 5% of the influent volume. Sand filters may require very frequent cleaning and subsequent sludge disposal. Backwashing these filters will produce significant quantities of polluted waters. Activated carbon has to be regenerated. High temperature regeneration facilities are thereby required which will create an ash disposal or air pollution problem. Reverse osmosis can produce a contaminated reject stream of 10% to 40% of the original volume. Ion exchange units have to be regenerated.

Regeneration will produce an acid or alkaline liquid waste stream containing all of the contaminants which were removed. If chemical precipitation is chosen as polishing step, disposal of the sludge produced will have to be considered.

While several researchers have suggested specific polishing treatment methods following biological treatment, it must be borne in mind that the method chosen will be very specific for the effluent produced from biological treatment. Considering the great number of unknowns pertaining to effluent quality, the best way to approach a polishing treatment is to design it based on laboratory tests performed on the effluent produced. While this approach will take time and could result in an unacceptable discharge, it is probably the only way in which a cost effective polishing step could be chosen.

3. SOIL ATTENUATION AND LEACHATE RECYCLE TREATMENT

Other treatment methods which have been suggested for leachates include soil attenuation and leachate recycle. In soil attenuation, leachate is allowed to pass through the soil underlying the landfill. Here the contaminants can be attenuated through oxidation, absorption, dilution in the groundwater, chemical precipitation, ion exchange, biological reaction and filtration. Unfortunately, very little is known about these mechanisms with the exception of dilution and possibly biodegradation. This lack of knowledge appears to be leading to a regulatory agency concept of not allowing the use of soil attenuation mechanisms for leachate treatment. This is probably unfortunate because soil attenuation is the least expensive way in which leachate can be treated. It is unfortunate too, because attenuation undoubtedly does work. This can be appreciated by considering that very few incidents of leachate damage have been reported from the tens of thousands of landfills in existence. Given that cost is an important factor, this method of treatment should not be overlooked. It should be noted, however, that sand, gravel or fissured rock will provide no attenuation except for dilution and perhaps some biodegradation. Fine grained and organic soils tend to have the greatest attenuation capabilities.

Leachate recycle through the landfill is not a complete treatment method because at some time it will have to be stopped. The main purpose of leachate recycle is to enhance biological degradation of organics within the landfill. The result is that during leachate recycle, no leachate is produced and therefore no external treatment is necessary. When the recycle is stopped, however, treatment will likely be necessary. The treatment required will be similar to that previously discussed for leachate in the previous page of this appendix. However, the size requirements will be reduced because of the reduction in organic loading due to biodegradation within the landfill. While some attenuation of metals within the fill is considered possible during the recycle, it is likely that the total mass of metals discharged will be the same as if no recycle had been practised. The limited research conducted to-date on this aspect does indicate however, that peak metal concentrations are reduced through recycle. This would then allow for a reduced capacity of the treatment system as concentrations would be more uniform.

The use of leachate recycle as a complete treatment system is applicable only under conditions where water loss through evaporation and runoff is equal to precipitation. This may be slightly modified by taking into consideration the liquid capacity of the landfill. Care must be taken in using this factor as a sudden reduction in storage could occur with subsequent overloading of the recycle facilities.

Leachate recycle also may be applicable in high rainfall areas where some percentage of the liquid can be allowed to enter the soil below the fill. A very high degree of control would be necessary in this case. With the often great variability of underlying soils in deltaic areas such as those found in the Lower Mainland, it is unlikely that the sophisticated degree of liquid control necessary would be possible.

Once the recycle has stopped, a leachate treatment system will likely be necessary. In this situation, design of the system should be based on laboratory or pilot-scale tests performed on samples of the leachate being recycled. In this manner, no unacceptable discharges should occur.

It must be noted that experience at recycling leachate in full-scale landfills has been plagued with problems due to odour, increased gas production and the clogging of cover materials. Recycle facilities must therefore be designed to be sufficiently flexible so that these problems can be overcome.

The preceding treatment methods have received some attention from researchers and enough information is now being generated so that conservative designs can be put forward. One aspect which has received virtually no attention, although people are aware of it, is the disposal of resulting sludges and brines from treatment.

4. SLUDGE DISPOSAL METHODS

As previously mentioned, sludge volumes from biological treatment of leachates will be significant. Reasonable choices for disposal are spreading in the landfill, land application and incineration. Of these three, incineration undoubtedly would be the most costly because of the high capital costs of incinerators and the cost of dewatering devices which would be necessary. In addition, bottom ash and effluents or residues from necessary air pollution control equipment would have to be disposed, thus adding to the cost. While energy recovery could be practiced, the costs of incinerating sludge, disposing of the residues, and treating the leachate in the first place would probably exceed the costs of incinerating, composting or recovery of energy and materials from the raw refuse.

Land disposal of leachate sludges might be feasible under the right conditions. Transportation distances would have to be relatively short in order to keep costs down. Metal toxicity would have to be carefully examined. Application rates to land should follow current guidelines for sewage sludge disposal to land. These guidelines limit both the rate of nutrient and metal addition as well as the total mass of metals which can be applied. Dewatering of sludges might be necessary in order to keep transportation costs to a minimum. It should be noted that the effluent from dewatering devices would have to be treated by recycling these effluents back through the leachate treatment system. In light of some of the difficulties which have occurred in domestic waste treatment systems when this practice has been followed, great care should be taken in testing and designing such a system.

Sludge disposal to the landfill, in most cases, would be the least expensive method to follow. While some difficulties would likely arise in carrying out this practice, the volume of sludge produced per unit volume of refuse will be low enough so that incorporation of the sludge in the landfill should not create significant operating problems. It must be remembered, however, that sludge will continue to be generated for many years after refuse disposal operations have ceased, so that provision for this will have to be made.

The main question about sludge disposal in landfills is that of desorption of the metals through leaching. If complete desorption occurred, the treatment system would have to be operated forever, a not too appealing thought. While virtually no work has been done in this area, some research has been carried out showing that complexation and precipitation of metals occurs during biological treatment, both within the landfill and in external processes, so that metal leaching will not likely occur to a great extent. While this shows some hope, this method of disposal should be designed with sufficient flexibility so that treatment for metal removal could be incorporated in the treatment system if necessary.

Disposal of brines from reverse osmosis systems and regenerated fluids from exchange processes has received no consideration in regard to leachate treatment. Application to land or to the landfill are possibilities, but these should not be attempted until some data are available to show the feasibility of the approach. Deep well disposal is another possibility. This could be very costly and could lead to a number of adverse environmental effects. Along with the high costs of these treatment methods, the difficulties associated with residual disposal or treatment are probably sufficiently great to suggest that these treatment systems not be used.

5. OTHER METHODS OF LEACHATE TREATMENT

One method of leachate treatment which could be the least expensive is that of adding leachate to an existing municipal or industrial waste treatment system. Experimental research and full-scale studies have shown that up to 5% by volume of high strength leachate added to influent to activated sludge systems has created no difficulties. When volumes exceed this, decreases in BOD removal efficiency through reduced settling efficiency have been noted. Discharge of leachate to a primary sewage treatment plant might show some metal reduction. This would occur because, under the higher pH conditions of sewage, some metals would be converted from the dissolved (ionic) state to a suspended form and thus settle out in the sedimentation basin. In the case of the leachates from the landfills in the study area, this mechanism is unlikely since the pH's of the leachate are normally higher than those in the sewage. If the primary plant were discharging digested sludges to the receiving water, this method of treatment would have virtually no effect on metal reductions although the form could be different. Sludge disposal by other means would have to take into account the problems which might be encountered due to the increased metal content of the sludge. Adsorption of specific metals or organics could possibly occur on suspended materials found in municipal sewage flows. This has not been demonstrated and is not expected to effect any significant removals. Depending on the specific situation, if equilization was not provided this method could cause upset of the plant due to the great variations in hydraulic and contaminant loading added by the leachate. Consideration should also be given to possible corrosion problems caused by strong, low pH leachate in both the sewer system and the treatment plant.

Treatment of leachates from wood waste landfills has also received little study. The small amount of work which as been done, however, indicates that biological aerated lagoon treatment combined with nitrogen and phosphorus additions can produce a non-toxic effluent capable of meeting current regulatory requirements. While a few metals would be associated with the sludge from the treatment of wood waste leachate, the problems of sludge handling would likely be much less severe than those created by refuse leachate sludge, due to the very much lower metal concentrations in wood waste.

6. TREATMENT COSTS

Before a leachate treatment system is designed very serious consideration must be given to the costs of constructing and operating the treatment system. These costs will likely be high and when considered along with transportation and landfilling costs, together they could make other methods of disposal more attractive. Each system of treatment will have different costs at different locations. Costs will be sharply affected by land acquisition prices and factors such as pH adjustment, nutrient additions, dewatering and residue disposal. Very little cost data from leachate treatment plants are available. The few cost figures which have been developed from the literature have been put forward very reluctantly by those reporting such costs, mainly because of the great uncertainties concerning leachate treatment. The cost figures shown in the following Tables A-1, A-2, and A-3, must, therefore, be used with a great deal of care. It is felt that the costs are within a reasonable range and can be used to provide preliminary cost data.

TABLE A-1 LEACHATE TREATMENT COSTS, 1977, BY TREATMENT SYSTEM

Treatment Method	Flow Rate US gal/day	Influent BOD 25 000 mg/l	Influent BOD 5000 mg/l
		Treatment Costs US dollar/1000 US gal	
Activated Sludge	28 800	23.6	6.0
	2 880	41.4	11.9
Aerated Lagoon	28 800	17.9	4.1
	2 880	31.6	10.0
Anaerobic Filter*	28 800	17.9	5.9
	2 880	38.8	16.8
Aerated Lagoon + Sand	28 800	25.7	7.3
Filter + Activated Carbon	2 880	39.9	13.7
Aerated Lagoon + Sand			
Filter + Activated Carbon	28 800	27.6	9.2
+ Reverse Osmosis	2 880	44.6	18.4
Anaerobic Filter* + Sand	28 800	28.6	9.7
Filter + Activated Carbon	2 880	50	21.1
Anaerobic Filter* + Sand			
Filter + Activated Carbon	28 800	30.4	11.5
+ Reverse Osmosis	2 880	54.3	25.4

* Cost include credit for methane recovery at $1.50/1000 ft^3.
Source: Chian _et al_, 1977.

TABLE A-2 LEACHATE TREATMENT, 1977, BY CAPITAL AND OPERATING COSTS

Treatment Method	Leachate Flow US gal/day	Influent Strength	Cost (1977 U.S. $)
Aerated Lagoon	71 300	540 mg/l BOD	Capital $349 000 Annual O & M 24 200
Activated Sludge + Chlorination	145 300	12 750 mg/l BOD	Material and power costs only: $3/1000 U.S. gal.
Anaerobic Filter	1320-5280	5 000 mg/l COD	Capital $400 000
Aerated Lagoon + effluent spray on landfill	Not given	4 000 mg/l BOD	Total $20/1000 US gal. (Estimate)

Source: Conestago and Rovers, 1978

TABLE A-3 LEACHATE TREATMENT COSTS, 1974

Treatment Method	Leachate Flow US gal/day	Cost 1974 US $/1000 US gal
Aerated Lagoon	1 200 000	0.10
	600 000	0.14
	170 000	0.31
Activated Carbon	1 200 000	0.45
	600 000	0.63
	170 000	1.20
Reverse Osmosis	1 200 000	0.59
	600 000	0.63
	170 000	0.72
Ion Exchange	1 200 000	0.66

Source: Cameron, 1977.

APPENDIX B

MONITORING DATA

BURNS BOG LANDFILL

INTRODUCTION

Table B-1 shows the results of analyses for Wells No. 3 and 4. As the wells are located on the periphery of the site between the landfill lift and the intercepting ditches, the analytical results will provide an indication of the so-called raw undiluted leachate. This judgement must however be tempered again by understanding the dynamic and complex nature of the flow system and leaching processes.

Monitoring results for ditch control Site 7, 8 and 9 are shown in Table B-2. These results tend to confirm the rather poor background water quality of the local ditch water.

Table B-3 shows the monitoring results for Wells 1 and 5. This monitoring provides a check on the possibility of leachate loss through the perimeter ditch sidewalls into the adjacent unconsolidated peats. Site 5 is a monitoring well, located as shown in Figure 3.1.3 (Sect 3.1.4) some 150 metres west of the site boundary and perimeter ditch. Site 1 is a monitoring well at the eastern property perimeter and is used for control comparison. Data in Table B-3 show that there is no consistent contaminant trend between wells, however, notable increases are seen in total iron and ammonia.

Table B-4 shows the comparison of the upstream natural surface drainage water (Site 11) with the downstream mixed landfill and drainage waters prior to discharge to Crescent Slough (Site 15). Comparing the monitoring results for these sites, it is readily apparent that with the exception of only a few parameters, the downstream result of the Burns Bog leachate discharges, for the most part, show only slight increases. Colour, nitrogen forms, and chemical oxygen demand have the most significant increases. Metals and notably chromium show generally increased concentrations, but this must be tempered by the limited data base and detection limit variations. Chloride which is a conservative ion and is generally considered a good indicator of landfill leachate contamination, has a downstream level which is, on average, lower than the upstream. This is possibly due to background levels from salt water intrusion prior to diking. High chloride levels are present in the dredged sand water. Fluoride also shows a similar downstream increasing trend.

TABLE B-1 BURNS BOG LANDFILL - MONITORING DATA FROM WELLS 3 AND 4

Parameter	Well #3			Well #4		
	No. of Samples	Range Min. - Max.	Average	No. of Samples	Range Min. - Max.	Average
Colour	15	200 - 2250	750	14	75 - 1400	351
pH	26	5.3 - 7.6	6.7	23	6.2 - 7.8	7.03
Spec. Cond.	29	859 - 8300	3943	25	465 - 2800	8.17
Chloride	22	234 - 905	578	20	0 - 588	52.3
Fluoride	14	.05 - .13	.096	18	.1 - .48	.154
Hardness	22	146 - 676	367	20	195 - 441	275
Ammonia	22	5.7 - 290	98	20	0 - 6.05	2.24
Nitrate	20	.02 - 202	46	15	0.02 - 6.4	0.61
COD	22	2.5 - 760	460	20	16 - 347	136
Tannin & Lignin	1	15 - 15	15	18	2.2 - 20	85
As (T)	3	< 0.008 - .015	< .012	15	0 - .02	< .011
Cd	2	< .0005 - .0011	< .008	-	-	-
Ca	15	23.5 - 139	68	13	25.5 - 80	38.9
Cr (T)	2	.04 - .052	.046	16	0 - .04	.015
Cu (D)	2	.05 - .06	.055	3	<.001 - .004	.002
Cu (T)	2	.07 - .08	.075	16	.006 - .07	.031
Fe (D)	12	.4 - 8	2.4	18	.2 - 21	7.8
Fe (T)	11	.6 - 37	7.99	2	105 - 111	108
Pb (T)	4	.02 - .3	.13	18	< .01 - .23	<.041
Mg (D)	15	19.6 - 80	48.7	13	35.6 - 73.5	44.1
Mn (D)	4	.33 - .46	.405	3	.46 - .5	0.48
Mn (T)	2	.45 - .56	.505	16	.29 - .95	0.641
Hg (T) μg/l	2	.1 - .33	.215	-	-	-
Ni (T)	2	.05 - .05	.05	-	-	-
Na (D)	4	150 - 520	371	2	13.5 - 14.7	14.1
Zn (T)	2	.25 - .38	.315	16	.02 - .12	.050
Al (D)	3	.19 - .6	.34	4	<.01 - .5	0.22

Colour................. TCU
Specific Conductance... mmho/cm
pH..................... pH units
All others............. mg/l

T = Total
D = Dissolved

TABLE B-2 BURNS BOG LANDFILL - MONITORING DATA FROM STATIONS 7, 8, AND 9

Parameter	Station 7 No. of Samples	Station 7 Range Min. - Max.	Station 7 Avg.	Station 8 No. of Samples	Station 8 Range Min. - Max.	Station 8 Avg.	Station 9 No. of Samples	Station 9 Range Min. - Max.	Station 9 Avg.
Colour	16	300 - 900	552	15	375 - 750	538	15	500 - 1000	698
pH	27	1.6 - 6.9	4.1	30	2 - 7.2	4	24	2.4 - 6.8	4
Spec. Cond.	29	46 - 11 000	540	31	52 - 4120	469	23	62 - 1750	193
DO	8	5 - 9.9	8.21	17	2.6 - 12	6.9	7	3.8 - 8.5	6.34
Chloride	21	.4 - 68.5	8.95	38	1.8 - 438	46.3	16	.4 - 100	10.74
Fluoride	19	.03 - .23	.096	4	<.01 - .01	<.01	5	.04 - .1	<.08
Hardness	4	3.23 - 5.24	4.4	8	2.01 - 10	5.14	3	3.31 - 4.52	4.05
Ammonia	24	0 - 1.6	0.26	24	0 - 204	14.85	14	0 - 3.2	.28
Nitrate	20	<.02 - .6	.094	23	<.02 - .6	.085	4	<.02 - .02	.02
Kjeldahl N	14	.45 - 5	1.70	15	.44 - 221	27.8	4	.49 - 3	1.31
COD	15	55 - 349	137.8	13	72 - 393	158	16	63 - 225	138
Tannin & Lignin	4	10 - 25	15.9	5	7.6 - 18	12.3	5	6.1 - 25	16
As	1	<.005 - .005	<.005	3	<.005 - 92.7	31	1	<.005 - .005	<.005
Cd (T)	-	-	-	2	0 - .01	.005	1	0	0
Ca (D)	4	.32 - 1	.63	5	.1 - 45	.312	4	.4 - .77	.57
Cr (T)	1	<.005 - .005	.005	3	0 - .01	.005	2	0 - .005	.003
Cu (T)	1	.001 - .001	.001	7	.001 - .06	.023	2	.001 - .05	.026
Fe (D)	18	.1 - 4.1	.687	34	.1 - 13.7	1.90	10	.1 - 1.07	.457
Fe (T)	4	.2 - .6	.35	3	.2 - .4	.27	3	.2 - .2	.2
Pb (T)	3	.003 - .003	.003	5	.004 - .05	.017	4	.003 - .05	.015
Mg (D)	6	.59 - 2.2	1.03	7	.33 - .92	.575	6	.5 - 1.6	.79
Mn (T)	3	<.02 - .02	.02	5	<.02 - .5	<.116	4	.02 - .04	.025
Hg (T) μg/l	2	.05 - .07	.06	3	<.01 - .05	<.037	2	<.05 - .05	.05
Ni (D)	1	.01	<.01	4	0 - .01	<.008	3	0 - .01	<.0067
Na (D)	4	2.3 - 4.9	3.35	6	1.5 - 5	2.65	5	1.2 - 7.7	3.44
Zn (T)	1	<.005 - .005	<.005	3	0 - .11	<.04	2	<.005 - .12	<.0075
Al (D)	1	.22	.22	4	.16 - .24	.19	6	.13 - .5	.28

Colour.................TCU
Specific Conductance...mmho/cm
pH.....................pH units
All others.............mg/l

T = Total
D = Dissolved

TABLE B-3 BURNS BOG LANDFILL - MONITORING DATA FROM WELLS 1 AND 5

Parameter	Site 1 Control Well			Site 5 Well to West of Site		
	No. of Samples	Range Min. - Max.	Average	No. of Samples	Range Min. - Max.	Average
Colour	13	450 - 1200	799	15	60 - 900	246
pH	20	3.9 - 6.9	4.6	24	6.2 - 7.4	6.8
Spec. Cond.	24	45 - 11 000	643	27	300 - 2400	522
Chloride	18	2.2 - 74	10.68	21	5.5 - 21	8.4
Fluoride	1	.13	.13	4	.14 - .19	.16
Hardness	18	3.45 - 176	25.8	21	156 - 262	189
Ammonia	16	0 - 1.5	0.4	21	5.5 - 9.34	8.08
Nitrate	16	< .02 - 1.3	<0.16	19	<.02 - .7	<.14
Kjeldahl N	3	2 - 5	4	4	10 - 13	11.8
COD	18	115 - 506	248	21	57 - 311	123.2
Tannin & Lignin	-	-	-	-	-	-
As (T)	-	-	-	-	-	-
Cd (T)	-	-	-	-	-	-
Ca (D)	13	.49 - 5.6	1.99	14	23.4 - 33.3	29.2
Cr (T)	-	-	-	-	-	-
Cu (D)	2	.01 - .01	.01	4	.002 - .009	.00475
Fe (D)	15	.4 - 3.0	1.45	19	.3 - 9	6.15
Fe (T)	3	1.9 - 11.2	5.6	4	35.2 - 90	57.8
Pb (T)	2	.014 - .03	.022	4	.001 - .21	.078
Mg (D)	13	.54 - 3.4	1.38	14	23.2 - 29.6	26.5
Mn (T)	2	< .02 - <.02	<.02	2	.67 - 1.1	.89
Ni (T)	1	<.01	<.01	2	.06 - .1	.08
Na (D)	2	2.6 - 2.9	2.75	4	7.9 - 9.3	8.5
Al (D)	1	.82	.82	3	<.01 - .03	<.017

Colour.................TCU
Specific Conductance...μmho/cm
pH.....................pH units
All others.............mg/l

T = Total
D = Dissolved

TABLE B-4 BURNS BOG LANDFILL - MONITORING DATA FROM STATIONS 11 AND 15

Parameter	Station 11			Station 15		
	No. of Samples	Range Min. - Max.	Average	No. of Samples	Range Min. - Max.	Average
Colour	14	60 - 300	185	15	200 - 1250	663
pH	27	6.4 - 7.5	6.9	31	5.7 - 8	7.2
Spec. Cond.	26	56 - 6450	2294	28	630 - 5310	2374
DO	18	.9 - 8.4	4.9	26	.7 - 7.2	3.8
Chloride	13	97.6 - 1490	812	26	70.5 - 733	402
Fluoride	6	.35 - .51	.45	7	.17 - .35	.211
Hardness	6	97.4 - 734	573	8	249 - 537	388
Ammonia	14	.177 - 2.12	1.04	28	10.5 - 198	66
Nitrate	6	.04 - 2.6	.57	21	.09 - 3.51	1.01
Kjeldahl N	6	2 - 6	3.5	15	13 - 198	74
COD	13	44.2 - 170	84.7	12	125 - 350	204
Tannin & Lignin	6	5 - 2	3.9	7	8.9 - 20	16.3
As (T)	4	0 - .01	.004	5	0 - .02	.01
Cd (T)	3	0 - .0006	.00037	3	0 - < .0005	< .0003
Ca (D)	6	14.6 - 73	56	8	38.8 - 123	71.3
Cr (T)	4	0 - .01	.0004	5	0 - .04	.013
Cu (T)	4	.004 - .02	.011	5	.005 - .1	.03
Fe (D)	16	.3 - 2.6	1.23	13	.9 - 12.3	4.45
Fe (T)	5	.9 - 5.1	2.72	18	2.1 - 21.5	6.5
Pb (T)	7	0 - 0.2	.031	7	0 - .01	.005
Mg (D)	7	14.8 - 134	103	9	37 - 64	48.5
Mn (D)	2	< .02 - .4	< .21	10	.03 - .75	.41
Mn (T)	13	.08 - .57	.44	19	.29 - .92	.52
Hg (T) μg/l	4	< .05 - 1	< .29	4	.05 - < 1	< .30
Ni (T)	2	.03 - .05	.035	2	.03 - .03	< .03
Na (D)	6	870 - 67.5	472	6	259 - 308	283
Zn (T)	4	< .005 - .04	< .021	5	.02 - .12	.68
Al (D)	4	.02 - 1	< .315	3	.03 - < 1	< .36

Colour.................TCU
Specific Conductance...mmho/cm
pH.....................pH units
All others.............mg/l

T = Total
D = Dissolved

APPENDIX C

MONITORING DATA

RICHMOND LANDFILL

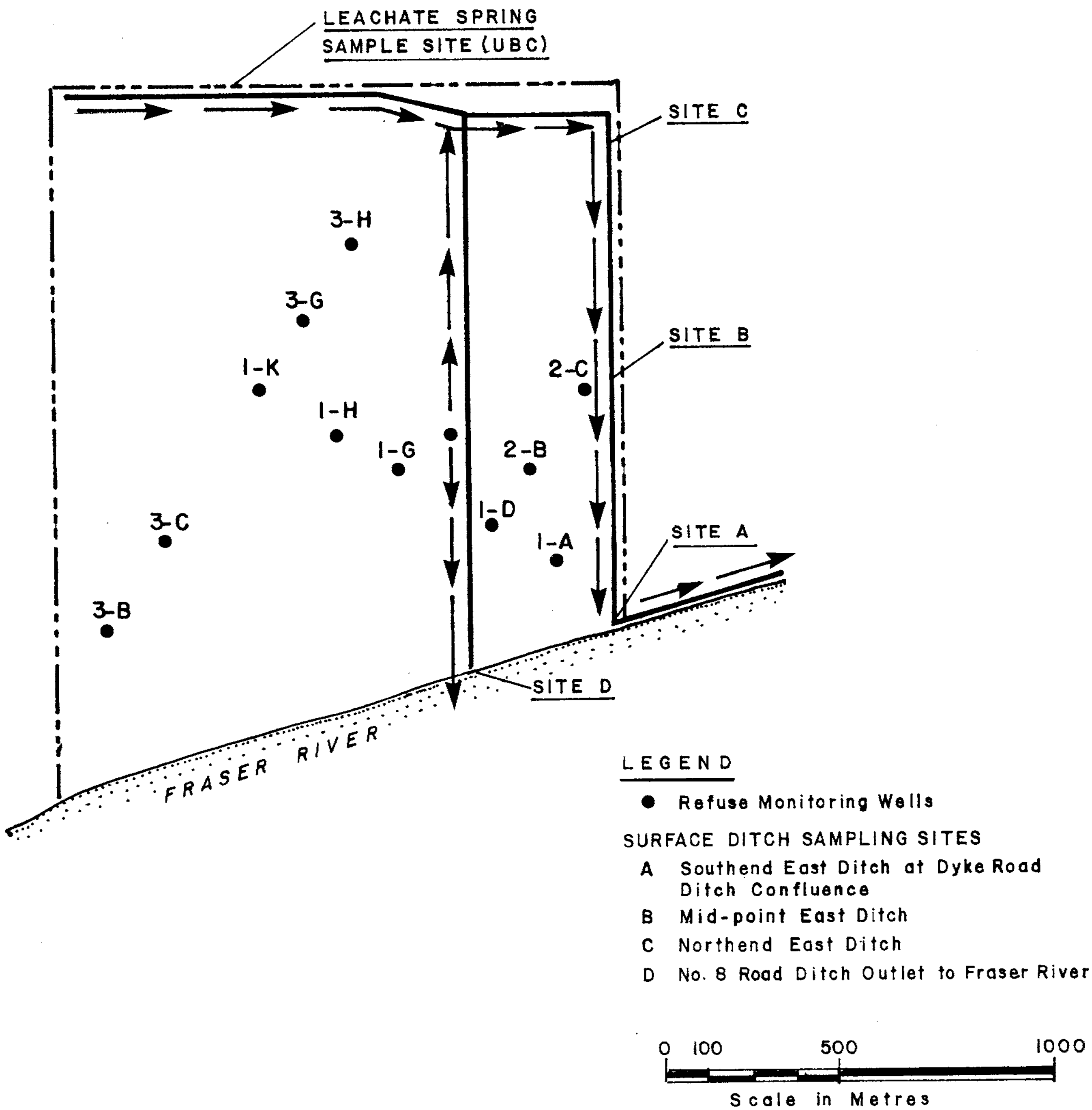
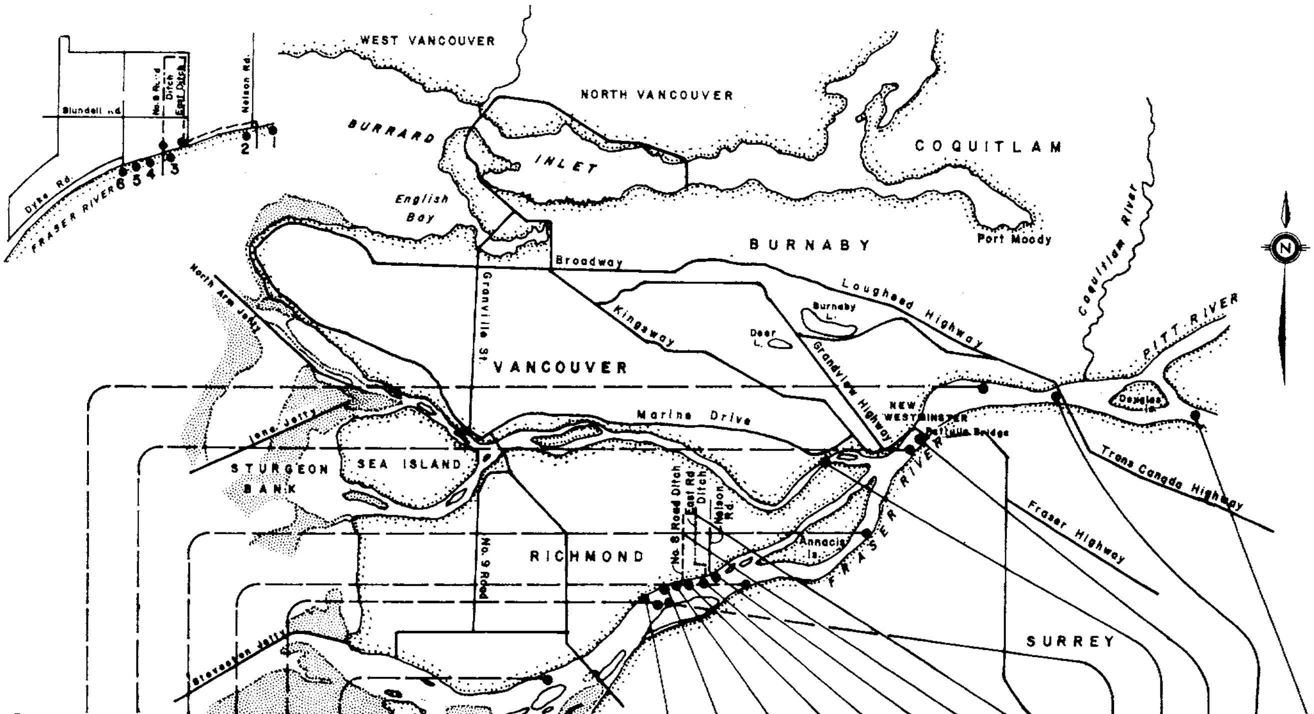


FIGURE C-1 RICHMOND LANDFILL LEACHATE SAMPLING SITES



EPS LOWER FRASER — November 1976 (ppm)

		F5	F4	F8A	F3	F2	F1
Hg	a	<.025	.141	<.025	<.025	<.025	<.025
	b						
Co	a						
	b	13	15	10	10	12	11
Cu	a						
	b	28	35	18	19	16	17
Fe	a						
	b	2.2 *	2.4 *	1.7 *	1.7 *	1.9 *	2 *
Mn	a						
	b	357	378	264	261	309	285
Ni	a						
	b	39	47	37	38	52	46
Pb	a						
	b	5	10	0	1	1	5
Zn	a						
	b	62	75	42	41	44	49
Ag	a						
	b	0	0	0	0	0	0
Cd	a						
	b	.3	0	.3	0	0	0
% Volatile Content		.7	1.6	<.3	<.3	<.3	<.3
PCB 1254		.01	0	0	.01	.01	.03
PCB 1260		0	0	.04	0	0	.01
Oligochaetes		158	602	2	0	5	0
Chironomid		26	0	2	1	1	0
Other		1	0	0	0	0	0

LEGEND
a EPS Laboratory
b UBC Laboratory
* Percent

RICHMOND LANDFILL - April 1977 (ppm)

Station No.		6	5	4	3	2	1	No. 8 Road Ditch	East Road Ditch
Hg	a	.07	.06	.06	.06	.05	.06	.11	<.05
	b								
Co	a	17	17	10	14	14	13	15	<9
	b	14	15	9	11	11	14	12	2.8
Cu	a	37	42	29	30	19	36	50	<1.1
	b	37	39	27	22	31	31	44	7
Fe	a	30×10^3	33×10^3	29×10^3	27×10^3	34×10^3	31×10^3	60×10^3	6.2×10^3
	b	2.2 *	2.4 *	1.9 *	1.9 *	1.9 *	2.0 *	4.2 *	2.6 *
Mn	a	720	650	460	530	510	630	1000	250
	b	496	434	271	311	279	387	742	425
Ni	a	55	63	46	48	46	51	52	13
	b	45	49	37	37	41	43	36	5
Pb	a	<13	26	41	19	<13	<13	100	<13
	b	6	15	30	52	2	14	94	0
Zn	a	82	94	130	70	54	88	170	14
	b	77	87	76	56	47	79	146	76
Ag	a	<4	<4	<4	<4	<4	<4	<4	<4
	b	0	0	0	0	0	0	0	0
Cd	a	<1.3	<1.3	<1.3	<1.3	<1.3	<1.3	<1.3	<1.3
	b	0	0	0	0	0	0	0	0
% Volatile Content		74	1.1	.2	.4	.2	.55	2.5	8.3
PCB 1254		.01	0	0	.01	.06	.14	0	0
PCB 1260		0	0	.03	.01	.01	.05	0	0
Oligochaetes		0	34-40	0	24-40	0	75		
Chironomid		4-12	36-54	8	112-124	4	240		

WESTWATER — July/August 1972 (ppm)

10	6	11	12	13
30	20	20	10	25
40	20	20	20	30
70×10^3	70×10^3	70×10^3	49×10^3	70×10^3
300	500	500	400	800
200	100	30	40	100
7	10	7	7	7
.27	.28	.36	.16	.19

FIGURE C-2 E.P.S. BENTHIC SEDIMENT SAMPLING LOCATIONS AND RESULTS (1976) RICHMOND LANDFILL SITE

TABLE C-1 SUMMARY OF LEACHATE DITCH SAMPLING - 1976 EPS STUDY RICHMOND LANDFILL SITE

Parameter	Site A			Site B		
	No. of Samples	Range Min. - Max.	Avg.	No. of Samples	Range Min. - Max.	Avg.
Alkalinity	5	660 - 875	785	4	740 - 1140	965
Cadmium	3	0 - <.01	-	3	0 - <.01	-
Calcium	5	170 - 230	184	2	190 - 250	220
Organic Carbon	3	30 - 118	68	2	40 - 75	58
Chloride	5	380 - 2200	1134	3	800 - 1900	1470
Chromium	5	0 - <.03	<.0067	5	0 - <.02	< .01
COD	6	120 - 320	227	3	200 - 290	257
Conduct(umho/cm)	4	2500 - 7500	4300	2	5400 - 7200	6300
Fluoride	1	.170	-	1	.14	-
Iron	7	2 - 42	22.5	3	28 - 93	58.3
Hardness	4	610 - 990	728	2	830 - 1030	930
Mercury	1	< 0.15	-	1	0	-
Manganese	2	2.9 - 120	61.5	-	-	-
Nickel	-	0 - <.05	-	-	0 - <.05	-
Ammonia	6	7.3 - 10	9.1	4	4 - 14	10
Sodium	3	500 - 1200	867	3	760 - 1100	983
Lead	5	<.02 - .22	.044	3	<.02 - <.05	-
pH (units)	5	6.9 - 7.5	7.4	3	6.8 - 7.4	7.2
Total Residue	4	1700 - 4800	3100	3	3300 - 4500	4000
Zinc	4	.08 - .22	.17	3	.16 - .5	.3
Copper	2	.01 - <.02	-	1	<.02	-

Source: Soper et al, 1977

Units: mg/l except for conductivity and pH

TABLE C-1 SUMMARY OF LEACHATE DITCH SAMPLING - 1976 EPS STUDY RICHMOND LANDFILL SITE (Continued)

Parameter	Site C			Site D		
	No. of Samples	Range Min. - Max.	Avg.	No. of Samples	Range Min. - Max.	Avg.
Alkalinity	5	800 - 1280	1136	4	337 - 920	682
Cadmium	3	0 - .01	-	3	0 - <.01	-
Calcium	4	200 - 370	283	2	86 - 210	100
Organic Carbon	2	95 - 300	198	3	40 - 115	70
Chloride	3	410 - 1600	1128	3	190 - 1700	873
Chromium	5	<.02 - .04	0.013	5	0 - .03	.015
COD	4	160 - 1900	720	3	110 - 300	217
Conduct(umho/cm)	3	2500 - 7150	5020	2	1270 - 6400	3840
Fluoride	1	23	-	1	0	-
Iron	4	1.8 - 17	6.4	3	18 - 47	34
Hardness	3	710 - 1480	1197	2	300 - 840	570
Mercury	1	<.15	-	1	0	-
Manganese	-	-	-	-	-	-
Nickel	-	0 - .05	-	-	0 - <.05	-
Ammonia	5	13 - 38	20.8	4	7.5 - 23	16.9
Sodium	3	700 - 900	780	3	120 - 980	510
Lead	5	<.02 - 1.01	-	8	<.02 - <.05	-
pH (units)	4	6.9 - 7.5	7.3	-	6.9 - 7.4	7.1
Total Residue	3	3400 - 4600	4000	3	750 - 4000	2340
Zinc	4	.09 - .82	.49	3	.11 - .21	.15
Copper	2	<.01 - .02	-	1	<.02	-

Source: Soper et al, 1977

Units: mg/l except for conductivity and pH

TABLE C-2 REFUSE UNIT MONITORING WELL RESULTS - 1976 EPS STUDY - RICHMOND LANDFILL SITE

Parameter	Range of Values for Wells with Screen Set in Refuse Unit
COD	82 - 11 000
Specific Conductance (umho/cm)	560 - 10 100
Total Residue	500 - 6080
pH (units)	5.3 - 11.6
Organic Carbon	18.0 - 1600
Inorganic Carbon	8.0 - 400
Alkalinity as $CaCO_3$	47 - 2200
Hardness as $CaCO_3$	222 - 1780
Chloride	5.0 - 3000
Sulphate	4.0 - 500
Ammonia - N	2.0 - 79
NO_3 and NO_2 - N	0.1
Calcium	30 - 770
Magnesium	6.0 - 150
Total Iron	2.8 - 490
Zinc	0.05 - 0.97
Sodium	7.7 - 1100
Nickel	0.05 - 0.1
Chromium	0.02 - 0.09

Units: mg/l except for conductivity and pH

TABLE C-3 LEACHATE SPRING ANALYSES - RICHMOND LANDFILL SITE

Parameter (mg/l)*	July 1977	January 1978
COD	1860	4720
BOD_5	1140	2980
Total Carbon	930	1830
Total Organic Carbon	810	1600
Total Residue	3190	6490
Total Volatile Residue	1470	2930
Total Dissolved Residue	3070	6470
pH (units)	6.2	6.3
Acidity ($CaCO_3$) @ pH 8.3	540	790
Alkalinity ($CaCO_3$) @ pH 3.7	1350	3050
Total Kjeldahl N	8.78	46
NH -N	.3	37.5
Total Phosphate P	4.67	3.1
Sulphate	250	83
Chloride	125	390
Sulphide	0.02	30
Boron	5.89	7.43
Calcium	535	1065
Sodium	128	358
Potassium	51	137
Magnesium	39	84
Iron	22.4	1.62
Manganese	4.3	7.76
Zinc	1.32	0.55
Aluminum	.36	1.26
Chromium	0.025	0.085
Copper	0.050	0.010
Nickel	0.002	0.012
Lead	0.051	0.023
Cadmium	0.002	0.001
Selenium	0.018	0.013
Arsenic	0.006	-

Source: U.B.C. Department of Civil Engineering, Leachate Treatability Study, 1978

*Except as noted

TABLE C-4 SURFACE DITCH SAMPLING RESULTS EAST DITCH (Site C) - RICHMOND LANDFILL SITE, 1978

Parameter (mg/l)*	February 22, 1978	February 28, 1978
pH (units)	7.0	7.4
T.Alk. ($CaCO_3$)	1730	1820
Sulphate	77.5	76.5
Chloride	382	420
Ammonia (N)	60.2	66.0
Nitrate (N)	1.19	.053
Specific Cond. (μmho/cm)	4420	4390
Non-filterable Residue	41.6	119
Total Residue	3290	3130
COD	539	940
Copper	< 0.02	0.02
Iron	18.8	25.8
Lead	< 0.10	< 0.10
Zinc	0.239	0.304
Sodium	306	262
Cadmium	< 0.010	< 0.010
Nickel	< 0.20	< 0.20
Manganese	4.08	4.27
Aluminum	0.80	0.43
Chromium	0.033	0.023
Arsenic	< 0.20	< 0.20
Mercury	< 0.10	< 0.10

*Except as noted.

TABLE C-5

CHARACTERISTICS OF SURFACE WATERS
IN THE ENVIRONS OF RICHMOND LANDFILL

Parameter	FRASER RIVER	NELSON ROAD AND NO. 7 ROAD DITCHES	EAST DITCH NO. 8 ROAD DITCH AND SURFACE SPRINGS ON FILL
pH	7.6	6.3 to 6.9	6.6 to 6.9
Organic carbon mg/l	3	10 to 31	45 to 400
Solids mg/l - floatable	nil	nil	nil
- Settleable	< 0.1	< 0.1	<1 to 78
- suspended	10 to 14	10 to 23	125 to 1353
- total	1060 to 1099	161 to 277	3689 to 5027
Ammonia mg/l	0.04 to 0.06	0.65 to 2.61	19.2 to 45.3
Phosphorus mgP/l	0.04 to 0.05	0.14 to 0.26	0.4 to 3.8
Total Coliforms MPN/100ml	22 000 to 79 000	2 400 to 49 000	340 to 70 000
Fecal Coliforms MPN/100ml	4 700 to 17 000	170 to 1 100	45 to 70 000
Temperature C	4.0	9.5 to 11.0	9.5 to 11.0
Dissolved oxygen mg/l	12.8 to 13.1	9.4 to 10.4	0 to 3.8
Toxicity 96-h LC50%	non-toxic	non-toxic	17 to 39
Iron (Dissolved) mg/l	< 0.1	0.6 to 1.8	10.5 to 22.0
Manganese (Dissolved) mg/l	< 0.05	0.05 to 0.15	4.3 to 8.5
Lead (Total) mg/l	0.001 to 0.012	0.007 to 0.021	0.008 to 0.195

B.C. Research: March to June 1979

TABLE C-6

CHARACTERISTICS OF LEACHATE FROM NORTH END OF NO.8 ROAD DITCH

PARAMETER*	NO. OF SAMPLES**	RANGE	MEAN
pH (pH units)	10	6.6 to 8.2	-
5 - day biochemical oxygen demand (BOD_5)	9	< 10 to 480	190
Chemical oxygen demand (COD)	10	132 to 859	480
Ammonia	11	8.7 to 91.0	49.2
Total Kjeldahl nitrogen (TKN)	10	21.1 to 97.0	57.2
Suspended solids	9	13 to 216	99.9
Total Solids	9	3 170 to 4 829	3 686
Oil and Grease	5	1.8 to < 5.0	< 5.0
Sulfate	4	40 to 90	62
Resin acids	1	- to -	not detectable
Polychlorinated biphenyl compounds (PCBs)	1		< 0.1 µg/l
Alkalinity (as $CaCO_3$)	5	300 to 2 290	1 055
Total coliforms (MPN/100ml)	3	270 to 35 000	-
Fecal Coliforms (MPN/100ml)	3	2 to 13 000	-
Toxicity 96-h LC50	5	7 to 39.0	-
Settleable solids	9	0.1 to 0.3	0.1
Iron (dissolved)	9	0.3 to 22.0	7.6
Manganese (dissolved)	9	2.3 to 4.8	4.1
Copper (dissolved)	9	< 0.05 to 0.07	< 0.05
Nickel (dissolved)	9	< 0.1 to 0.2	< 0.2
Cadmium (dissolved)	9	0.0003 to 0.001	< 0.001
Iron (total)	4	27 to 78	-
Manganese (total)	8	2.4 to 9.0	4.6
Tin (total)	5	-	< 5
Zinc (total)	5	0.06 to 0.07	0.07
Lead (total)	8	0.006 to 0.018	0.010
Chromium (total)	9	0.05 to 0.10	< 0.08
Mercury (total)	4	0.04 to 0.10	0.06 µg/l

*Values in mg/l except where stated
B.C. Research 1979.

*Samples were collected over a 12 week period March - June 1979.

APPENDIX D

WOOD WASTE

EPS AERIAL SURVEY 1977

INTRODUCTION

An aerial survey conducted by EPS during September 1977, revealed 35 wood waste landfills covering an area of 130 ha and containing an estimated 4,350,000 m^3.

Fill estimates were made using oblique aerial photography. A chartered Cessna 172 aircraft and a hand-held aerial camera operated through the co-pilot's open window position were used. The aircraft altitude was maintained as low as permissible, i.e., 240 to 300 metres in order to reduce haze effects and to obtain a large-image size on film.

The photographic equipment included an 80 m focal length lens in a Hasselblad Model 500 CM 2-1/4 SQ format camera equipped with a 70 mm film magazine. Kodak Aerocolour 2455 colour negative film allowing approximately 65 exposures per loading was used. Film exposures were determined based on an arbritrary A.S.A. film speed of 100 with the shutter speed fixed at 1/500th second. All film was processed by the National Air Photo Library in Ottawa, who provided continuous roll contact proof prints for selection. Selected negatives were subsequently enlarged and printed locally.

The areas and landfill heights were scaled using objects in the photographs of known heights and lengths such as box cars, scows, boom heights, etc.

It is assumed that the species mix of wood waste indicated in the Reid Collins report (Appleby, 1976) was representative of the 35 sites, however, on a site-by-site basis any given fill could contain from 0% to 100% of a particular wood species.

The surveyed area consisted of five sectors, A to E as shown in Figure D-1. There were no sites in Sector B. Table D-1 is a summary of the areas and volumes of the 35 sites. Figures D-2 to D-5 are location maps of the sectors A, C, D, and E, respectively. At some locations wood waste fill is in place adjacent to short-term storage or chip loading operations. In these instances an estimate has been made as to what percentage is fill.

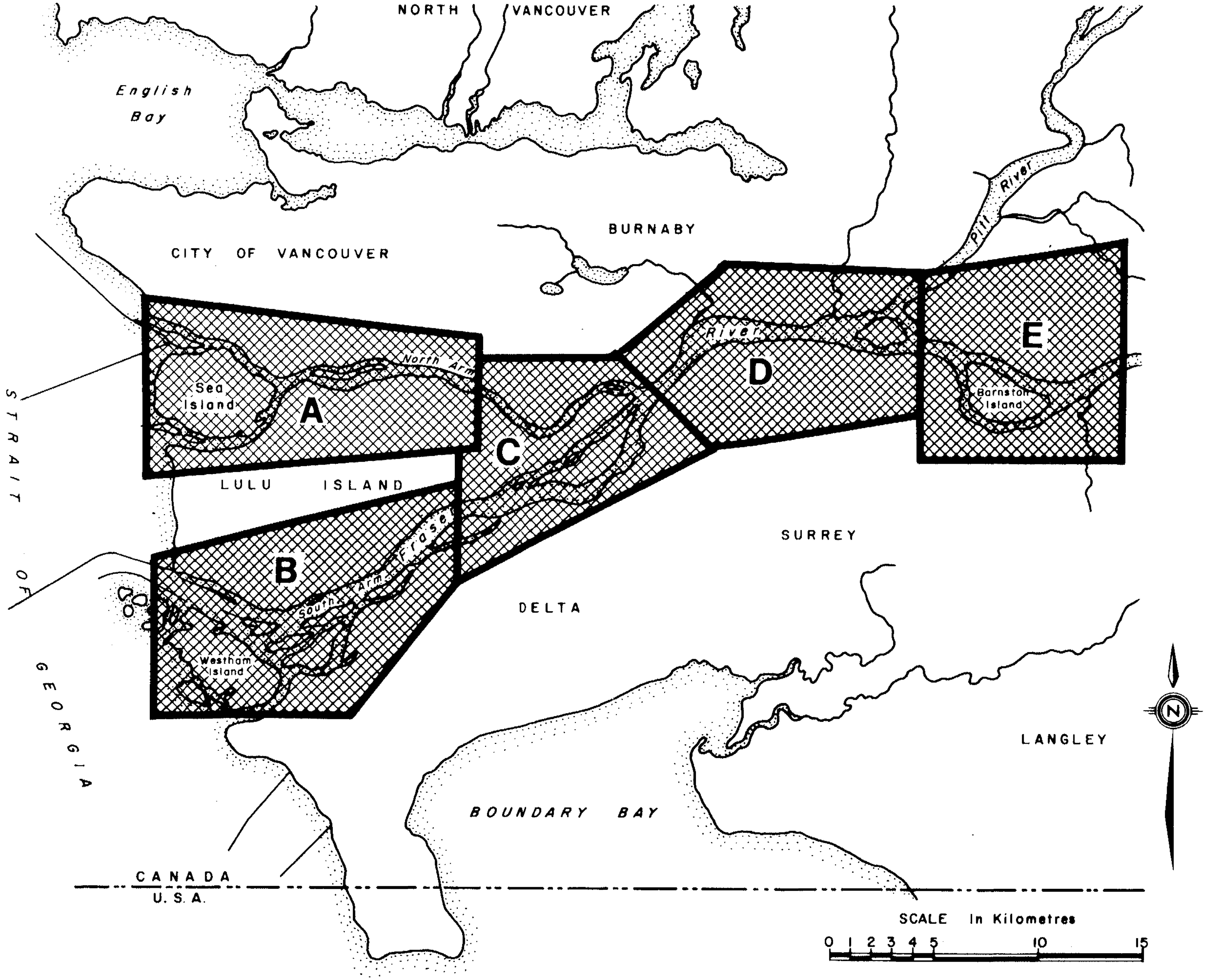


FIGURE D-1 WOOD WASTE DISPOSAL SITES - DESIGNATED AREAS IN THE AERIAL SURVEY

TABLE D-1 SUMMARY OF THE AREAS AND VOLUMES OF THE WOOD WASTE SITES - 1977 WOOD WASTE AERIAL SURVEY

Site	Height of Fill	Area m	Area Hectare	Volume m	Volume of fill m
A-1	7.6	2 830.2	0.3	21 509.5	21 509.5
A-2	15.3	8 139.7	0.8	124 537.4	87 176.0
A-3	15.3	2 325.6	0.2	35 581.0	3 558.0
A-4	8.0	2 809.4	0.3	22 475.2	22 475.2
A-5	20.7	4 695.7	0.5	97 201.0	97 201.0
A-6	20.7	4 695.7	0.5	97 201.0	97 201.0
				398 504.9	329 120.7
C-1	15.3	25 755.0	0.3	394 051.5	23 643.1
C-2	8.3	858.5	0.09	7 125.6	7 125.6
C-2	2.2	11 135.2	1.1	24 497.4	24 497.4
C-3	3.4	85 863.7	8.6	291 936.6	29 193.7
C-4	1.5	772.1	0.02	258.2	232.4
C-5	2.0	17 860.5	1.8	35 721.0	35 721.0
C-6	4.0	24 297.4	2.4	97 189.6	97 189.6
C-7	2.0	25 700.0	2.5	51 400.0	51 400.0
C-8	2.0	3 343.4	0.3	6 686.8	6 686.8
C-9	10.0	13 416.2	1.3	134 162.0	134 162.0
				1 043 028.7	409 851.6
D1-6	2.0	16 856.4	1.7	33 712.8	33 712.8
D-8	2.0	13 912.0	1.4	27 824.0	27 824.0
D-9	9.0	7 600.0	0.8	68 400.0	68 400.0
D-10	16.0	1 296.0	0.13	20 736.0	20 736.0
D-10	2.0	1 080.0	0.11	2 160.0	2 160.0
D-10	3.0	1 200.0	0.12	3 600.0	3 600.0
D-11	1.0	1 655.0	0.17	1 655.0	1 655.0
D-12	2.0	2 482.7	0.25	4 965.4	4 965.4
D-13	2.0	65 117.5	6.51	130 235.0	130 235.0
D-14	10.0	2 540.0	0.25	25 400.0	25 400.0
D-15	2.4	1 215 000.0	121.5	2 916 000.0	2 916 000.0
D-16	2.0	165.0	0.02	330.0	330.0
D-17	2.0	6 742.9	0.67	13 485.8	13 485.8
D-18	4.0	3 600.0	0.36	14 400.0	14 400.0
D-19	2.0	10 500.0	1.05	21 000.0	21 000.0
D-20	10.5	6 528.0	0.65	68 544.0	68 544.0
D-20	4.0	4 352.0	0.44	17 408.0	17 408.0
				3 369 856.0	3 369 856.0
E1-6	10.0	28 174.2	2.8	281 742.0	197 219.4
E-7	3.0	1 545.8	0.15	4 637.4	463.7
E-8	10.0	800.9	0.08	8 009.0	8 009.0
E-8	2.0	236.7	0.02	473.4	473.4
E-9	8.0	2 809.4	0.3	22 475.2	22 475.2
E-10	4.0	3 070.1	0.3	12 280.4	12 280.4
				329 617.4	240 921.1
			TOTAL	5 141 007.0	4 349 749.4

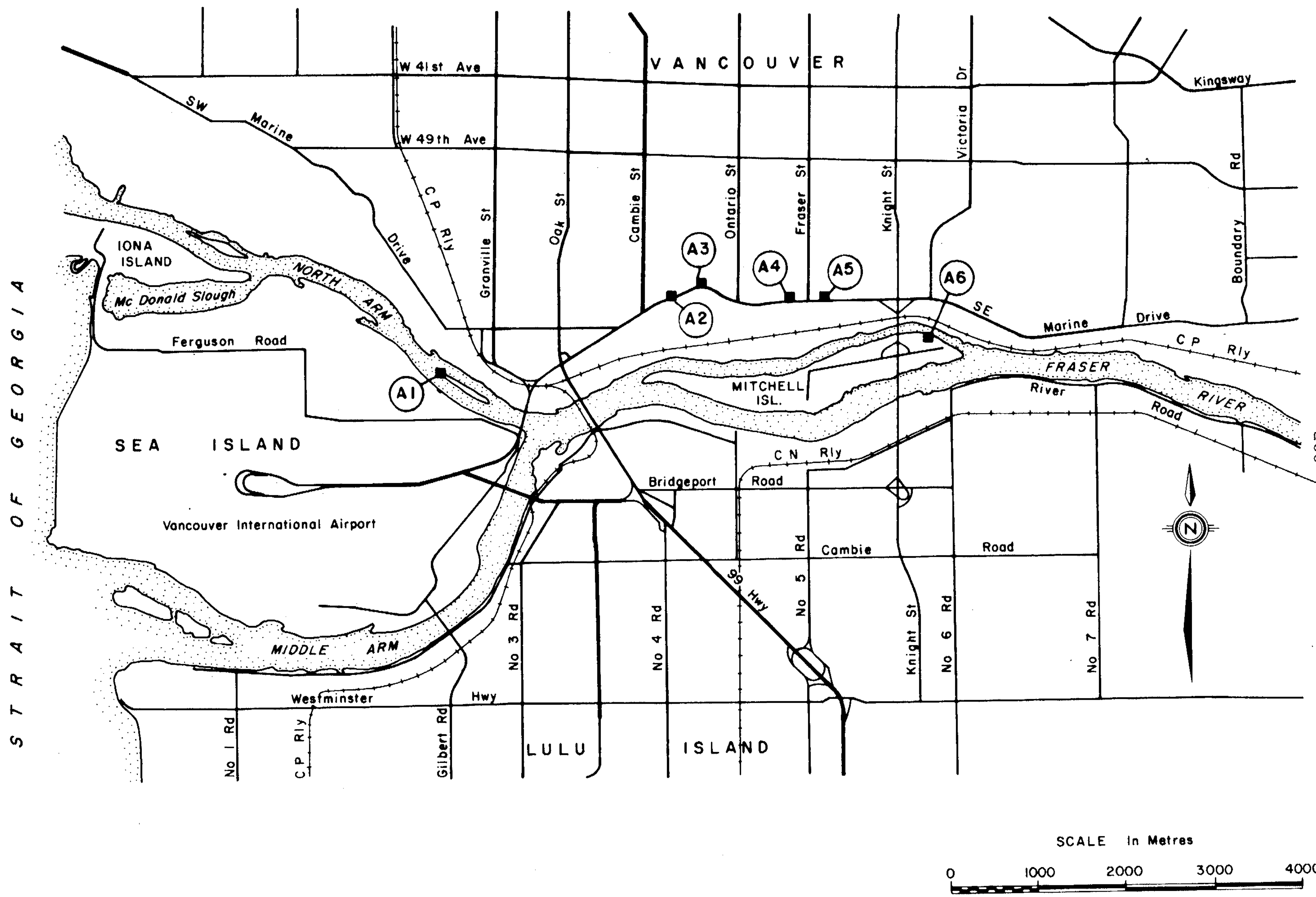


FIGURE D-2 WOOD WASTE SITES - AREA A

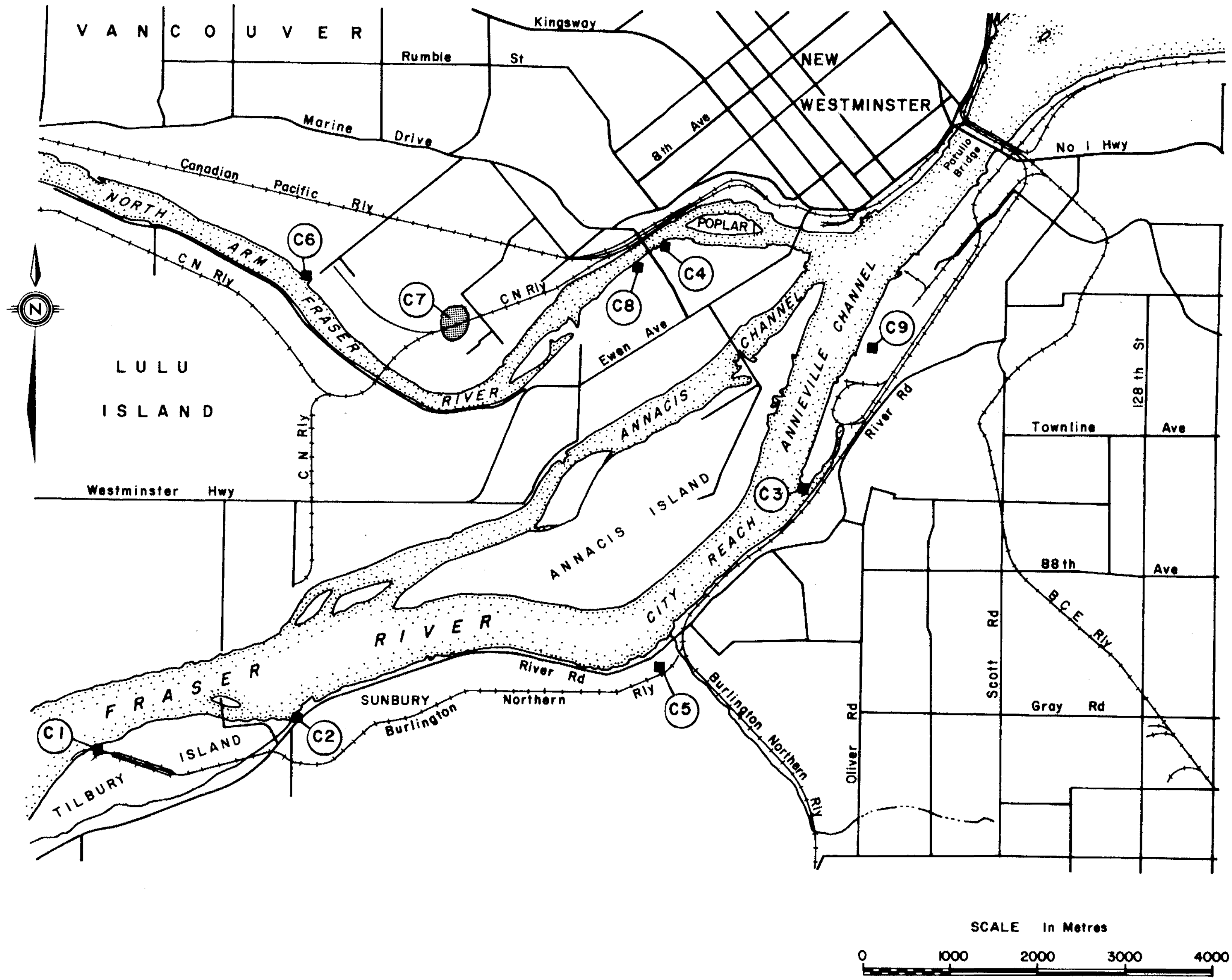


FIGURE D-3 WOOD WASTE SITES - AREA C

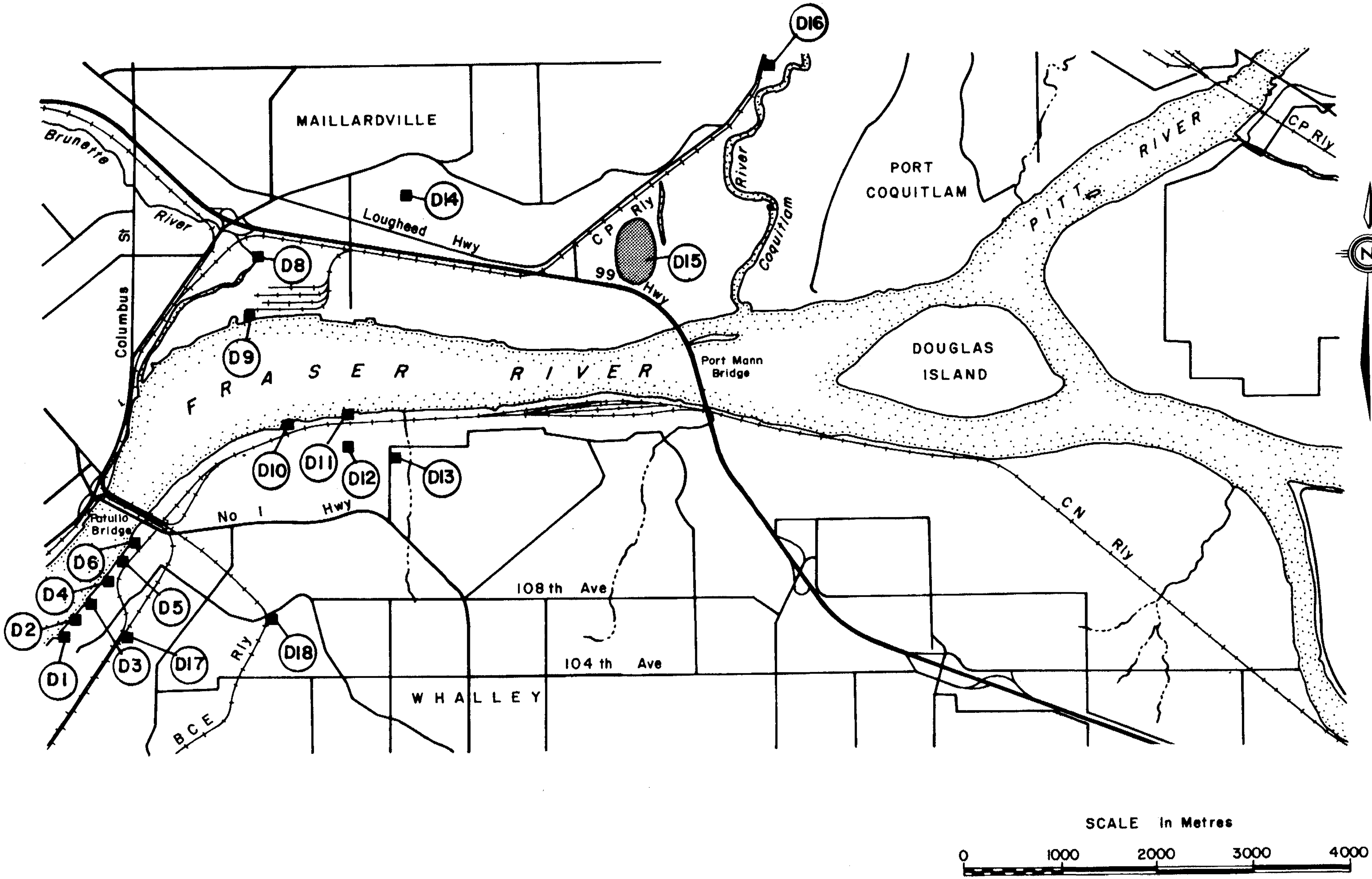


FIGURE D-4 WOOD WASTE SITES - AREA D

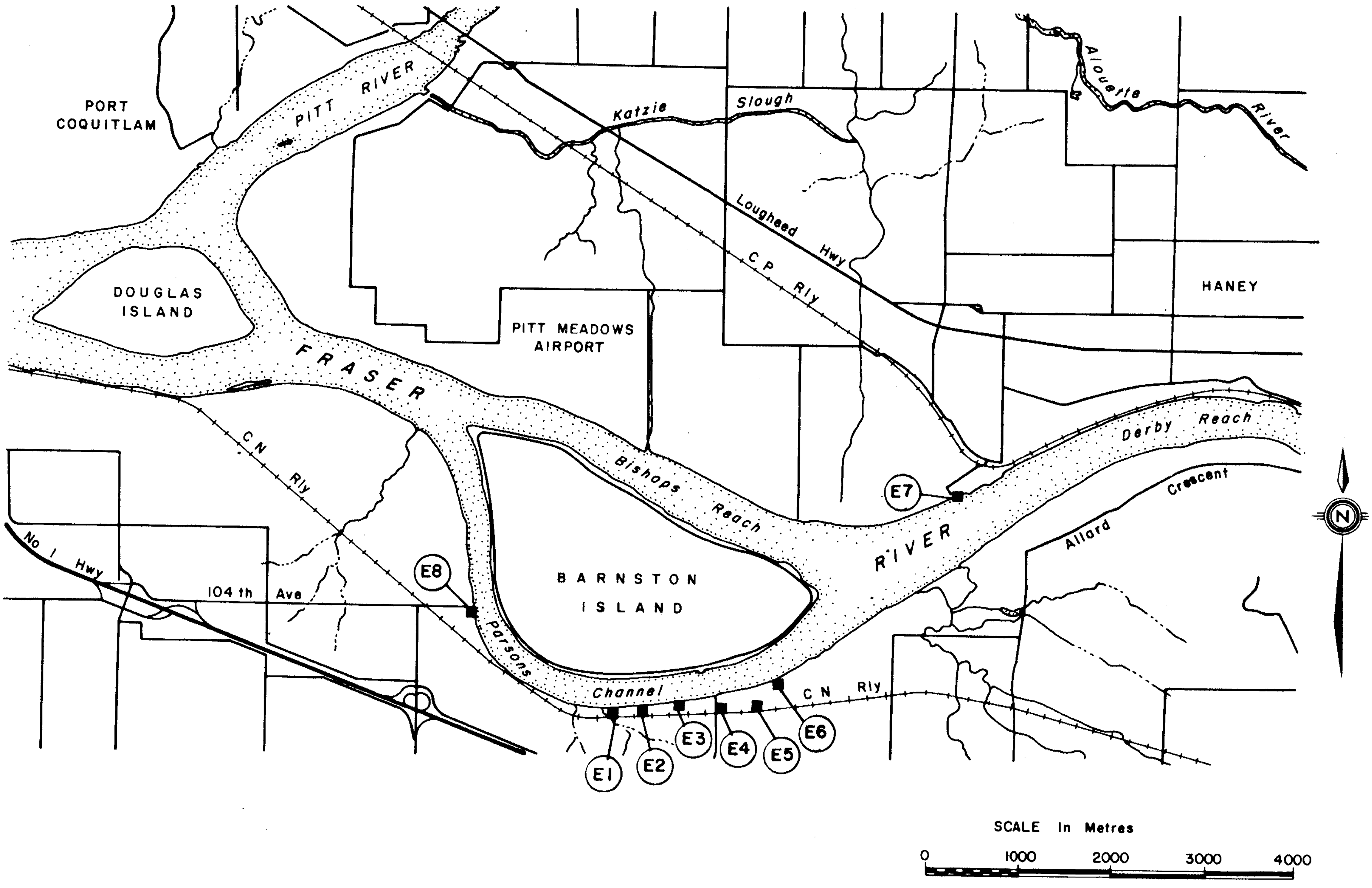


FIGURE D-5 WOOD WASTE SITES - AREA E

APPENDIX E

LOCATION MAPS AND SITE PLANS

CLOSED SMALL MUNICIPAL

LANDFILLS

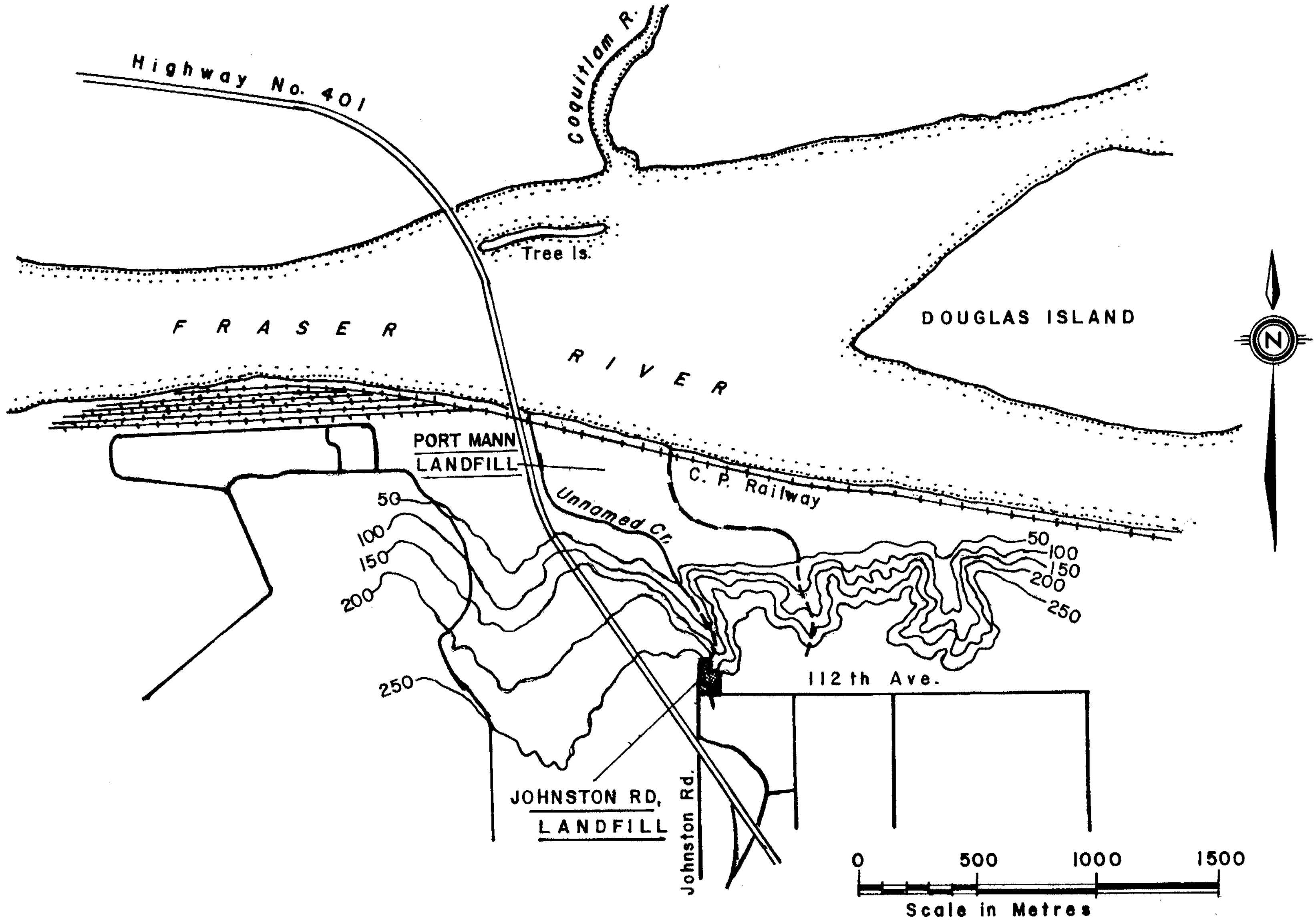


FIGURE E-1 LOCATION MAP - JOHNSTON ROAD LANDFILL

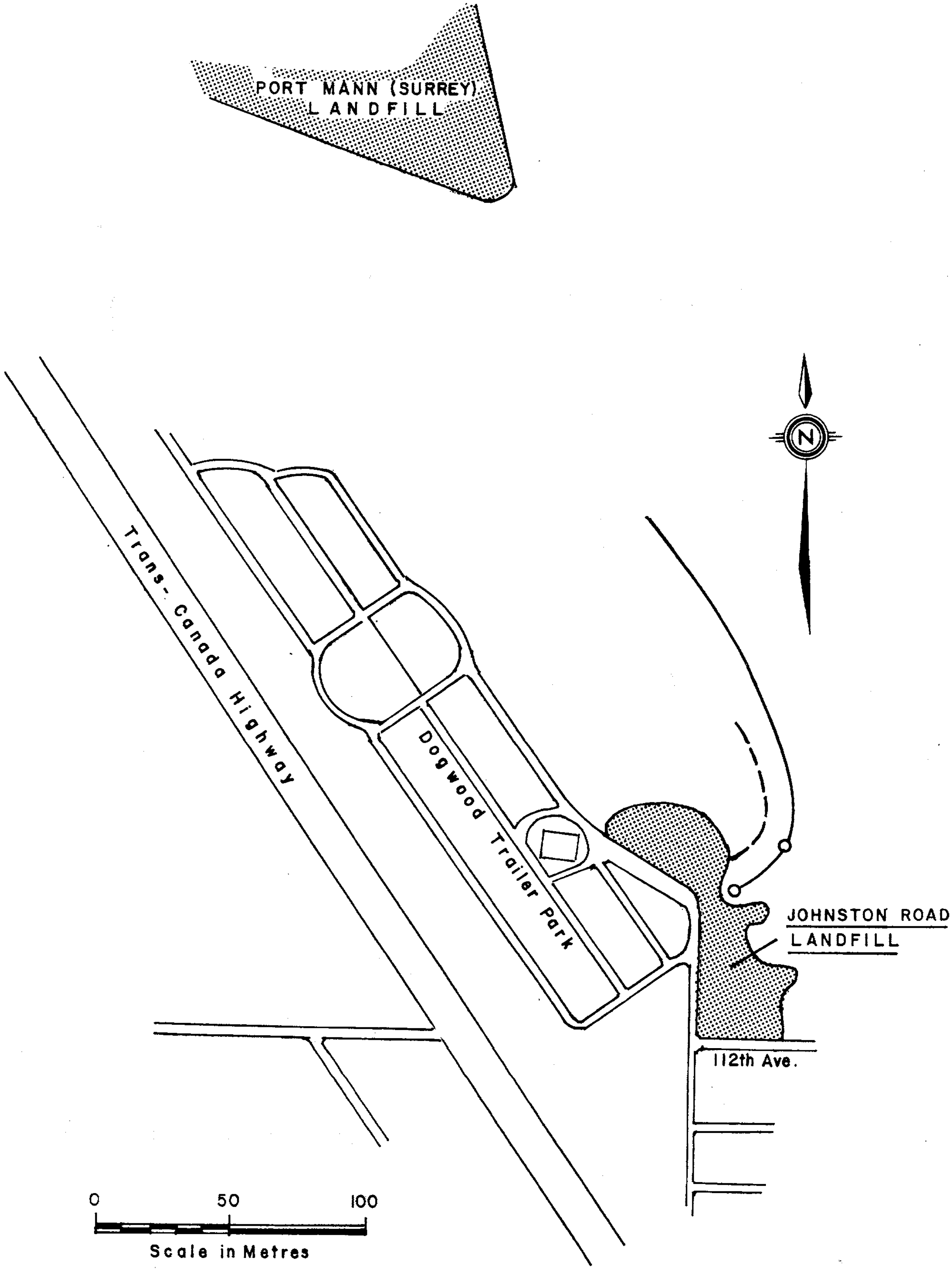


FIGURE E-2 SITE PLAN - JOHNSTON ROAD LANDFILL

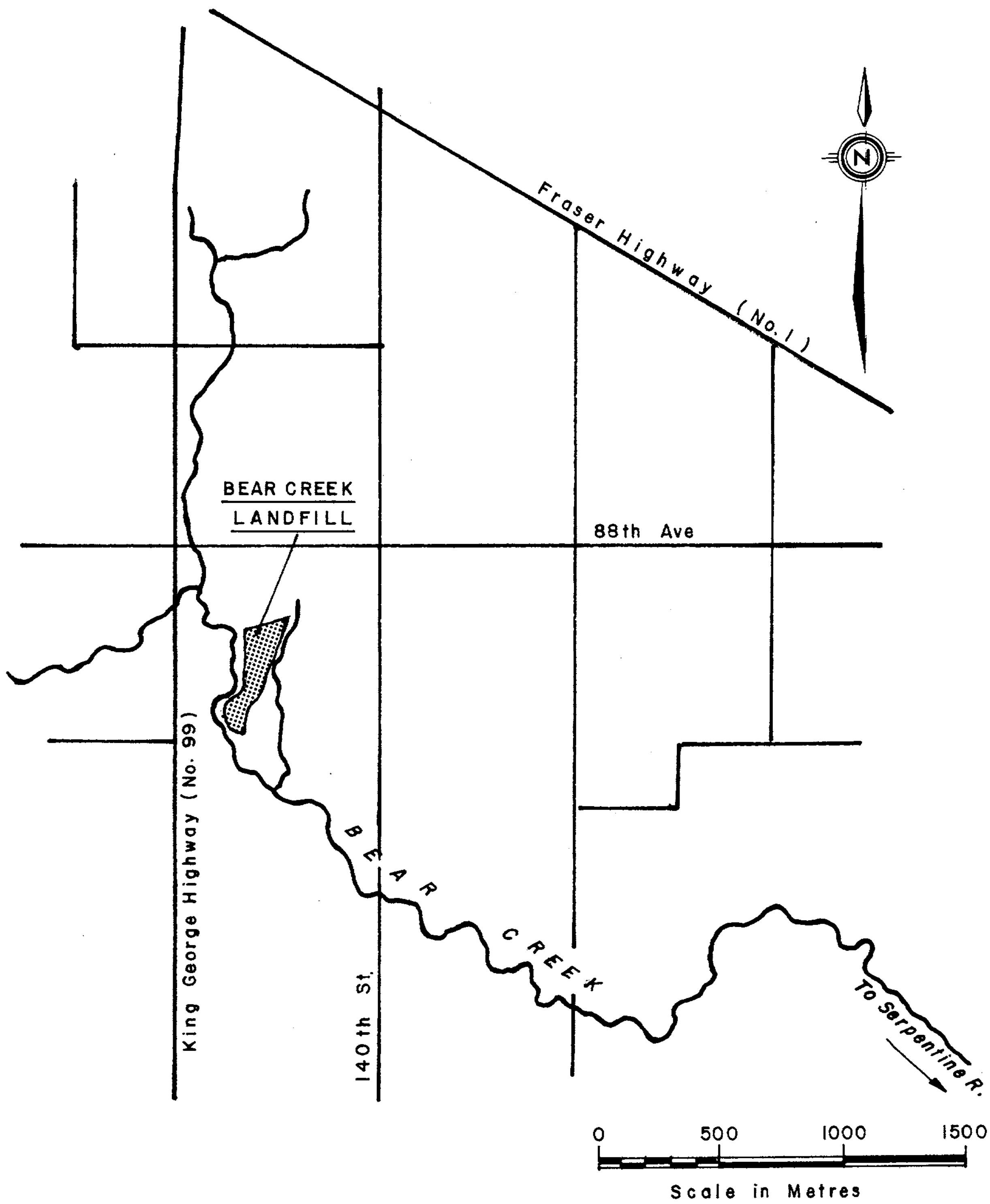


FIGURE E-3 LOCATION MAP - BEAR CREEK LANDFILL

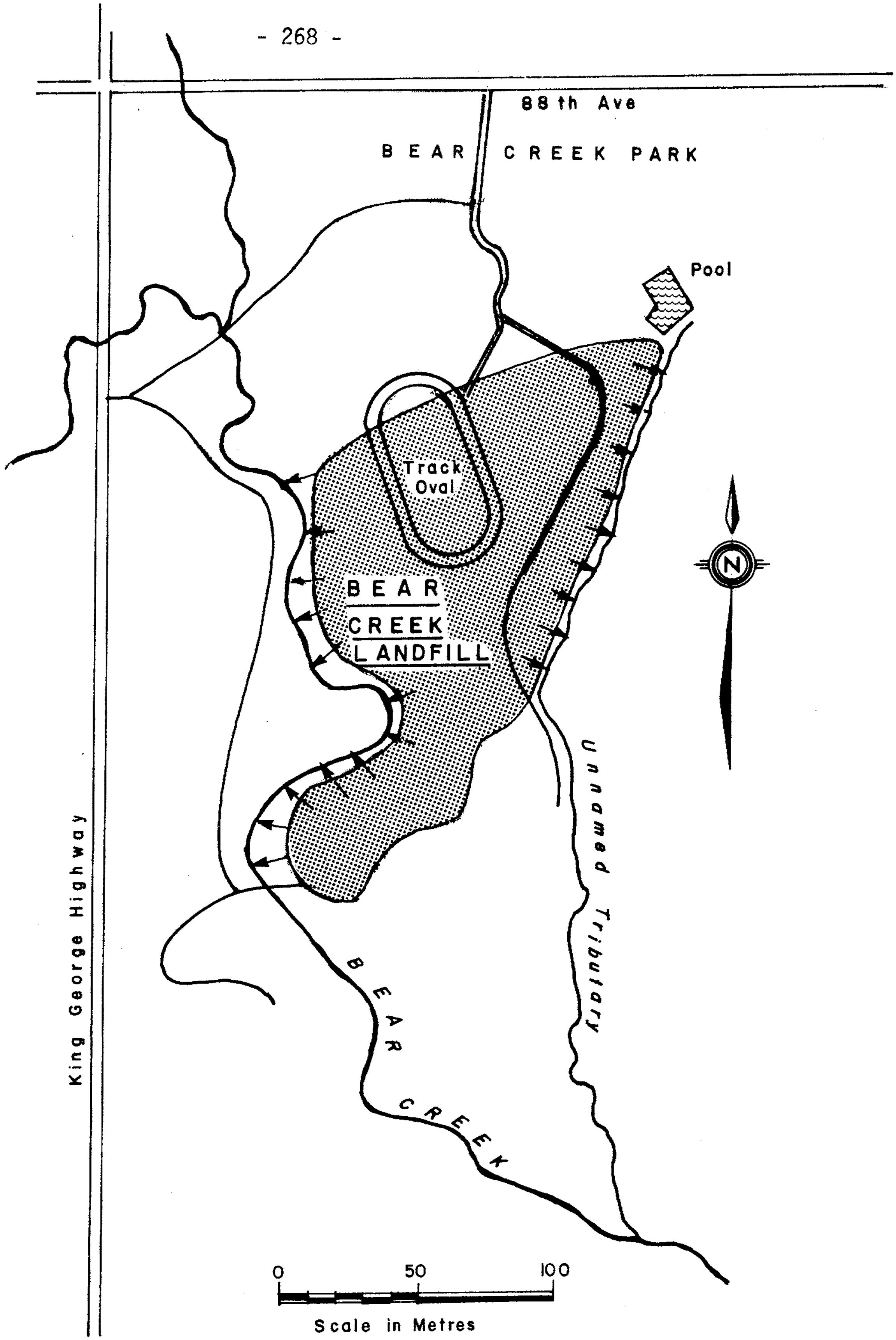


FIGURE E-4 SITE PLAN - BEAR CREEK LANDFILL

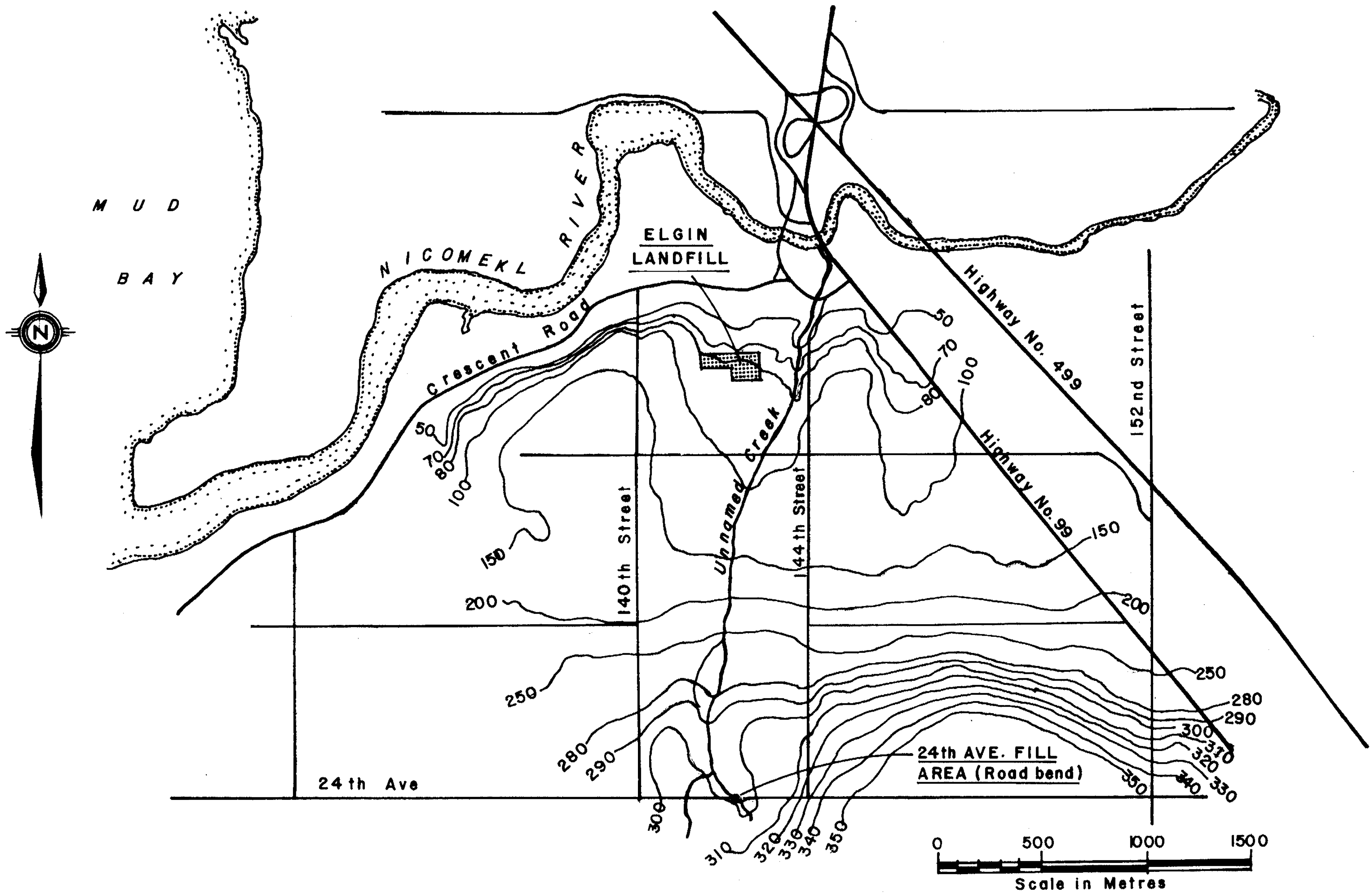


FIGURE E-5 LOCATION MAP - ELGIN LANDFILL

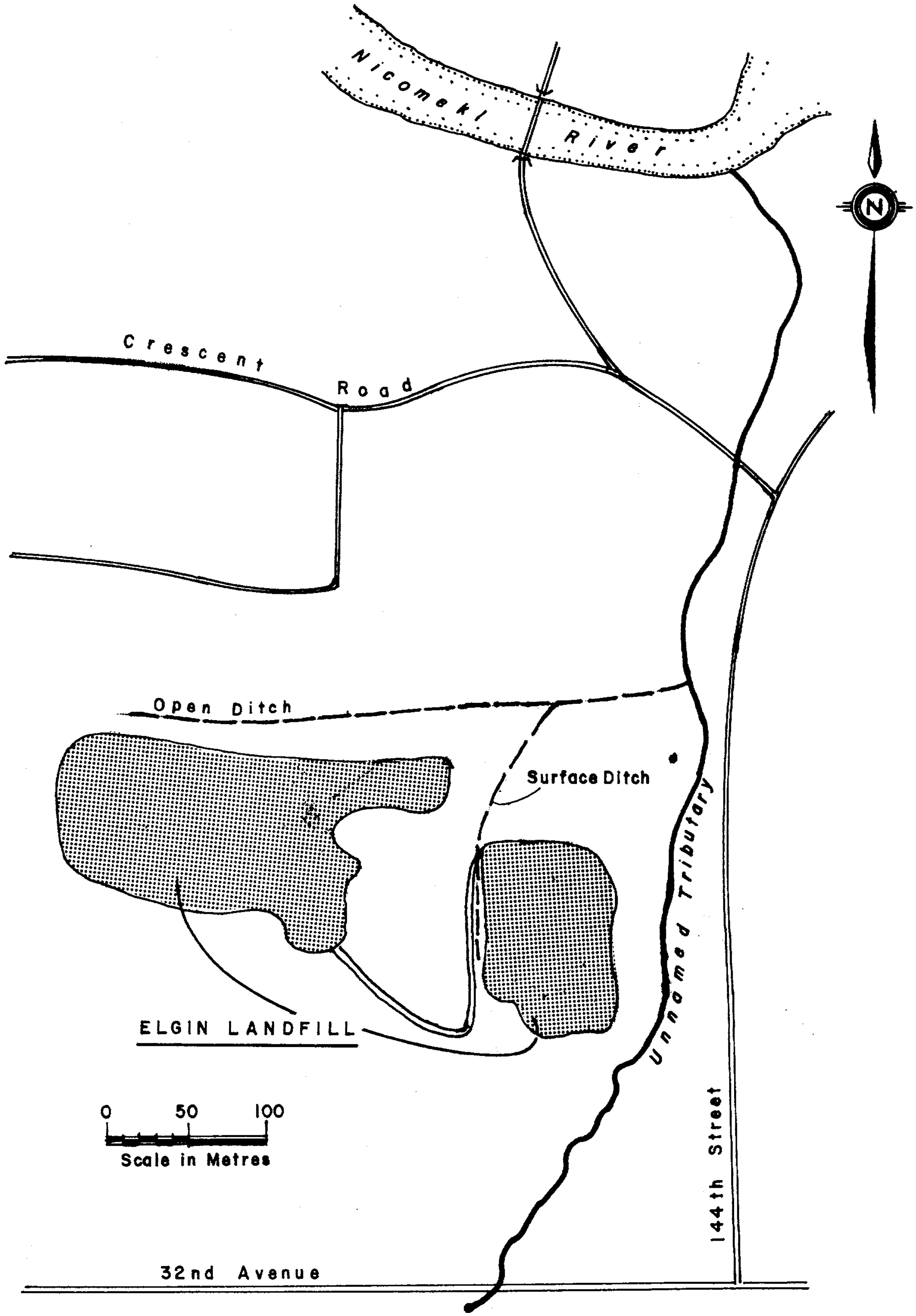


FIGURE E-6 SITE PLAN - ELGIN LANDFILL

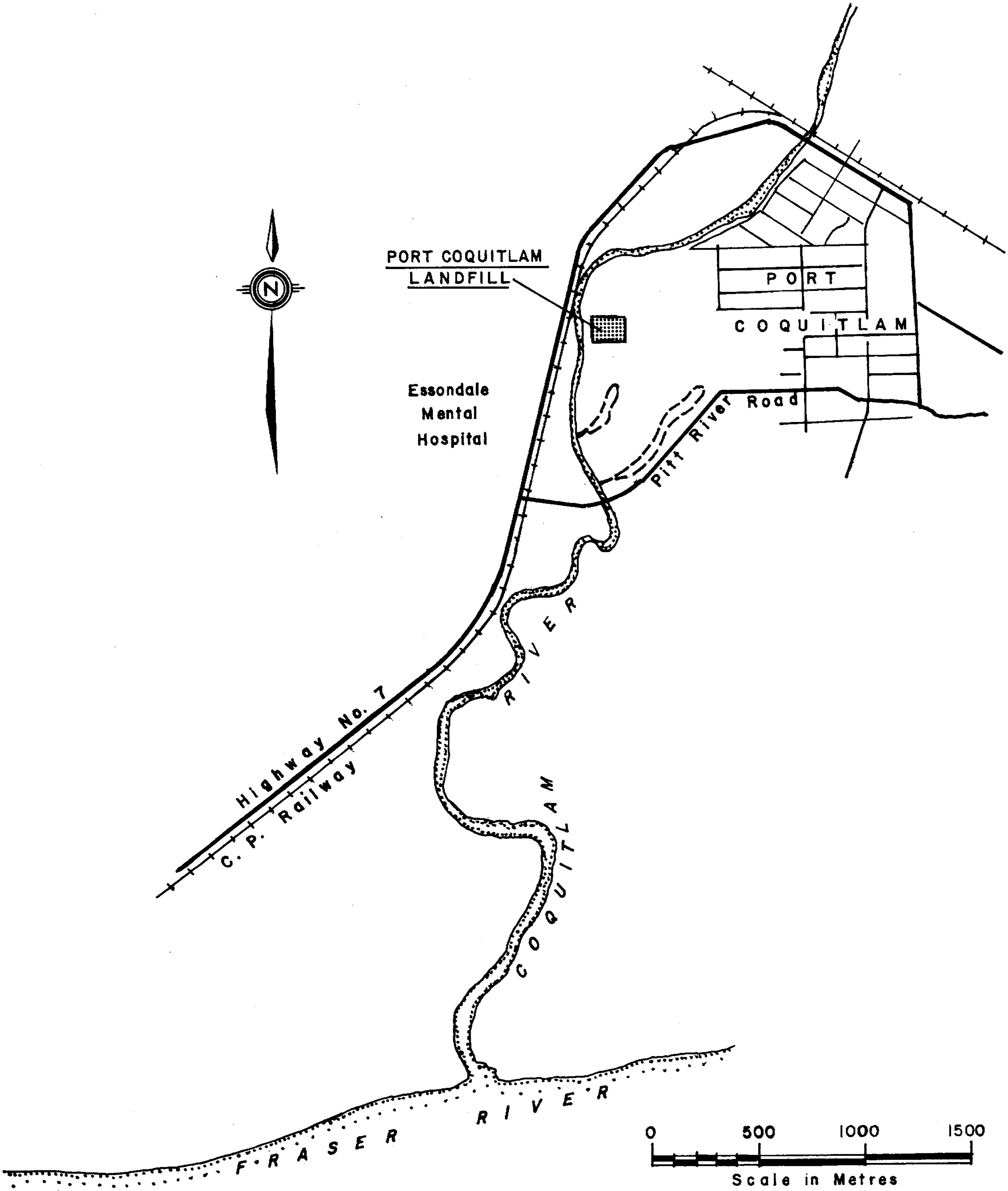


FIGURE E-7 LOCATION MAP - PORT COQUITLAM LANDFILL

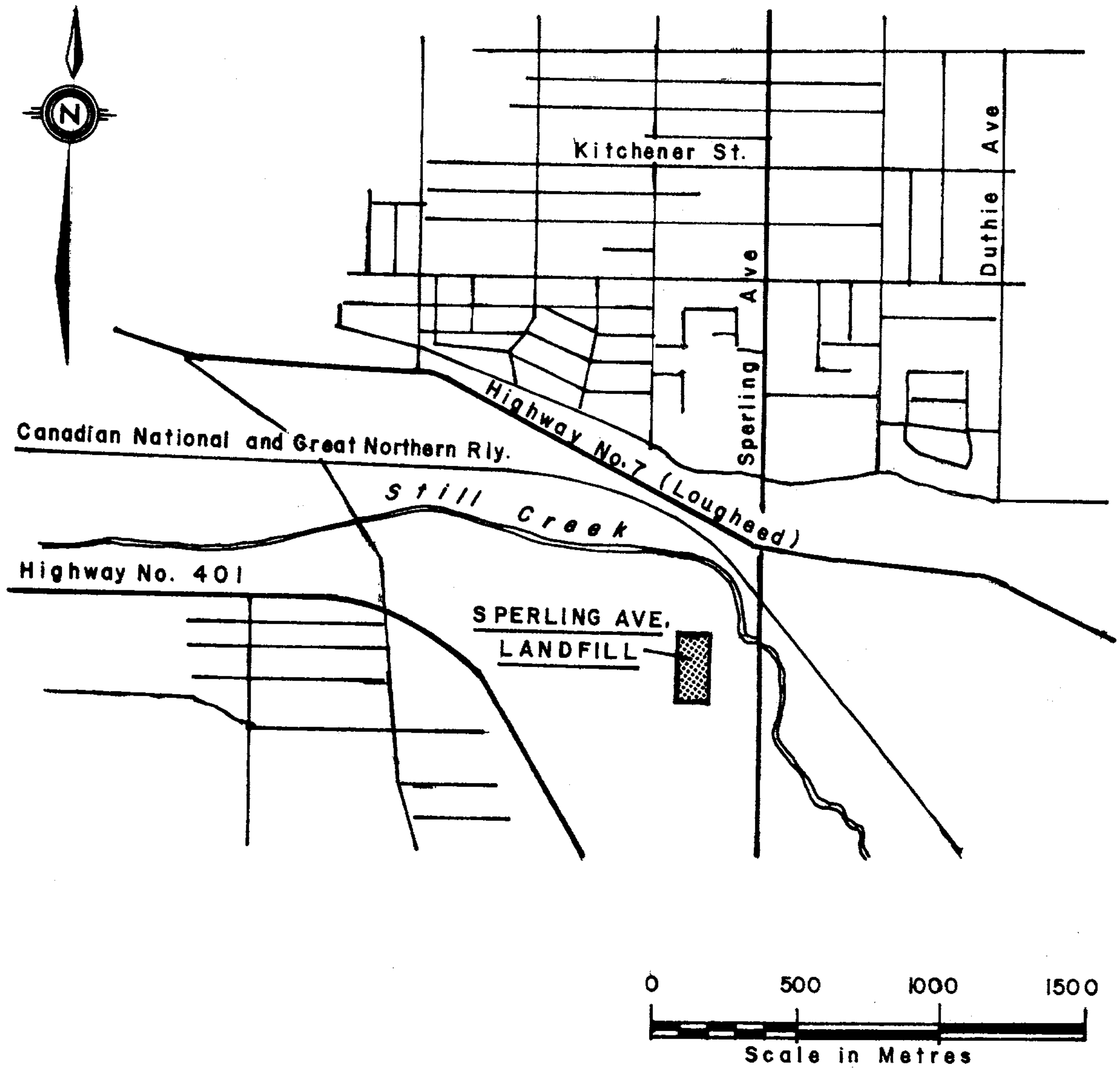


FIGURE E-8 LOCATION MAP - SPERLING AVENUE LANDFILL

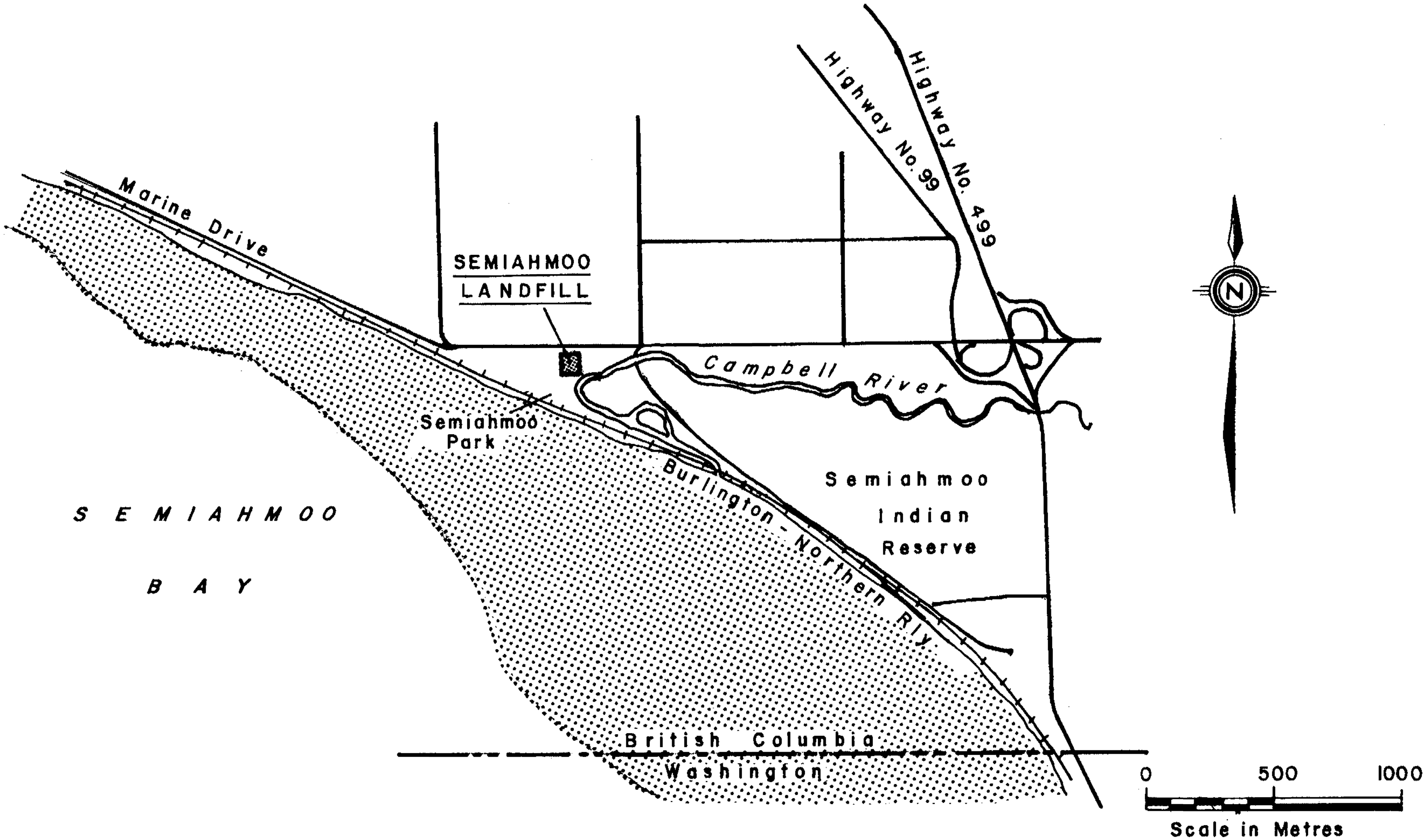


FIGURE E-9 LOCATION MAP - SEMIAHMOO BAY LANDFILL

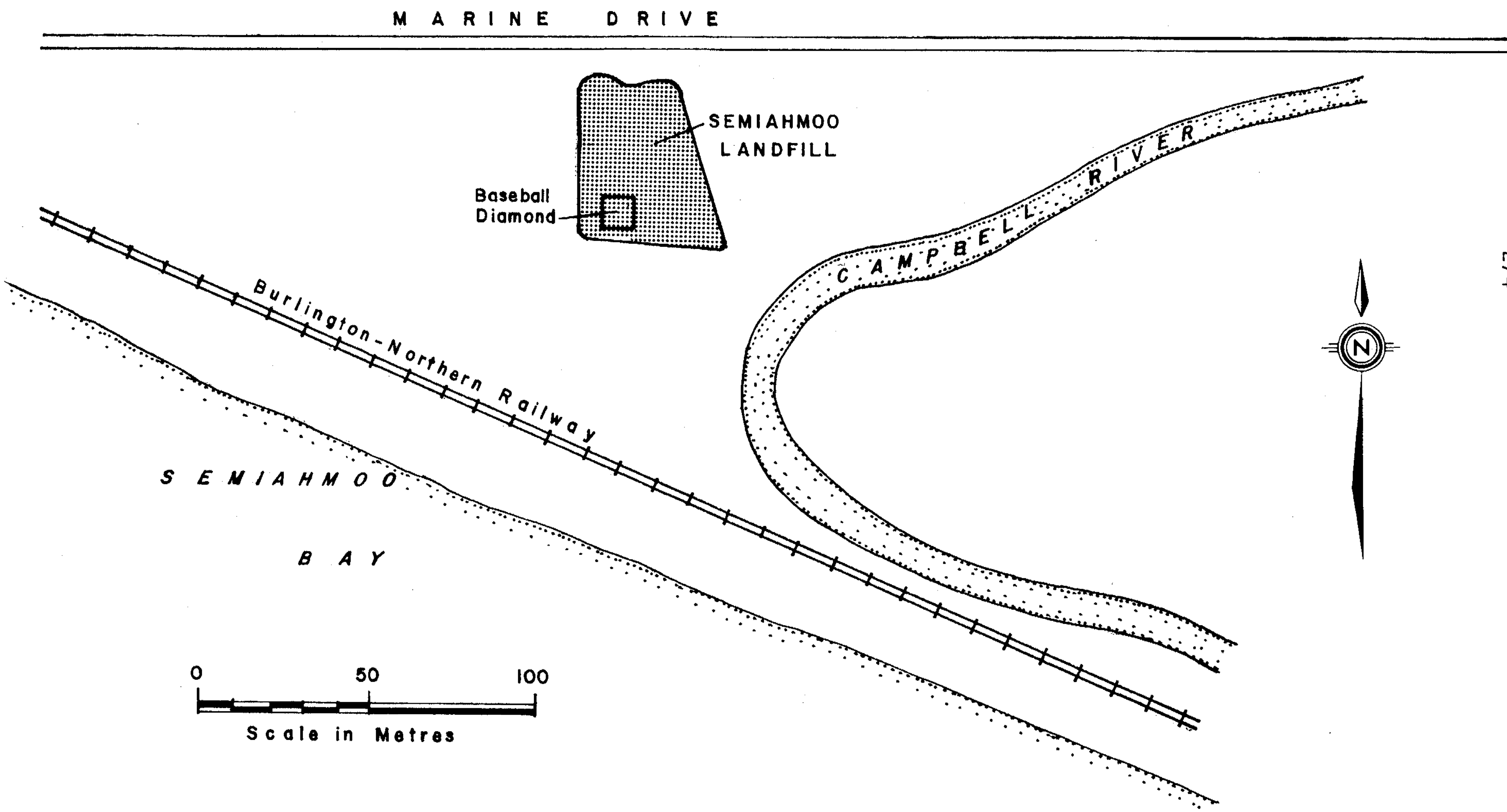


FIGURE E-10 SITE PLAN - SEMIAHMOO BAY LANDFILL

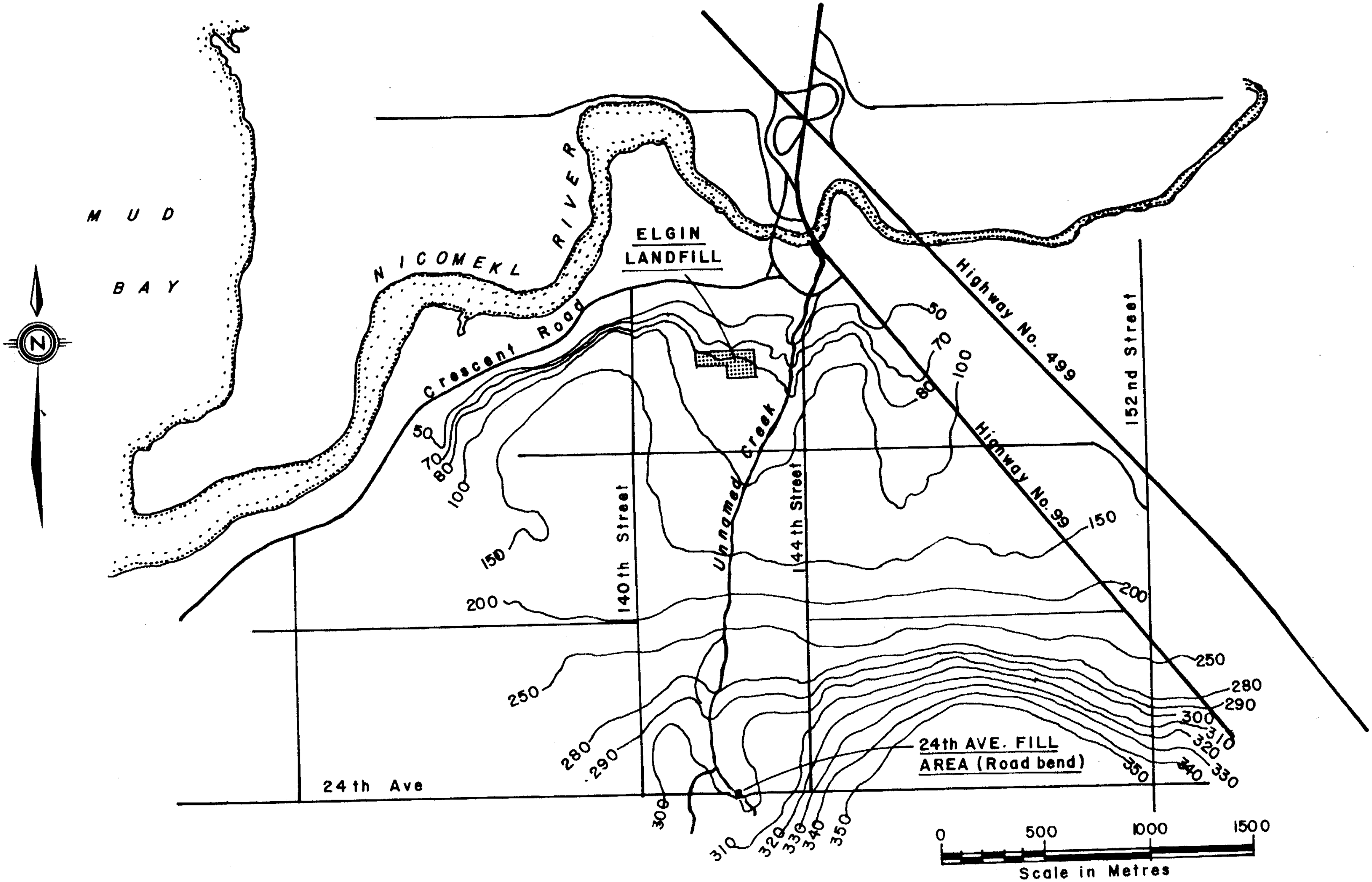


FIGURE E-11 LOCATION MAP - 24TH AVENUE LANDFILL

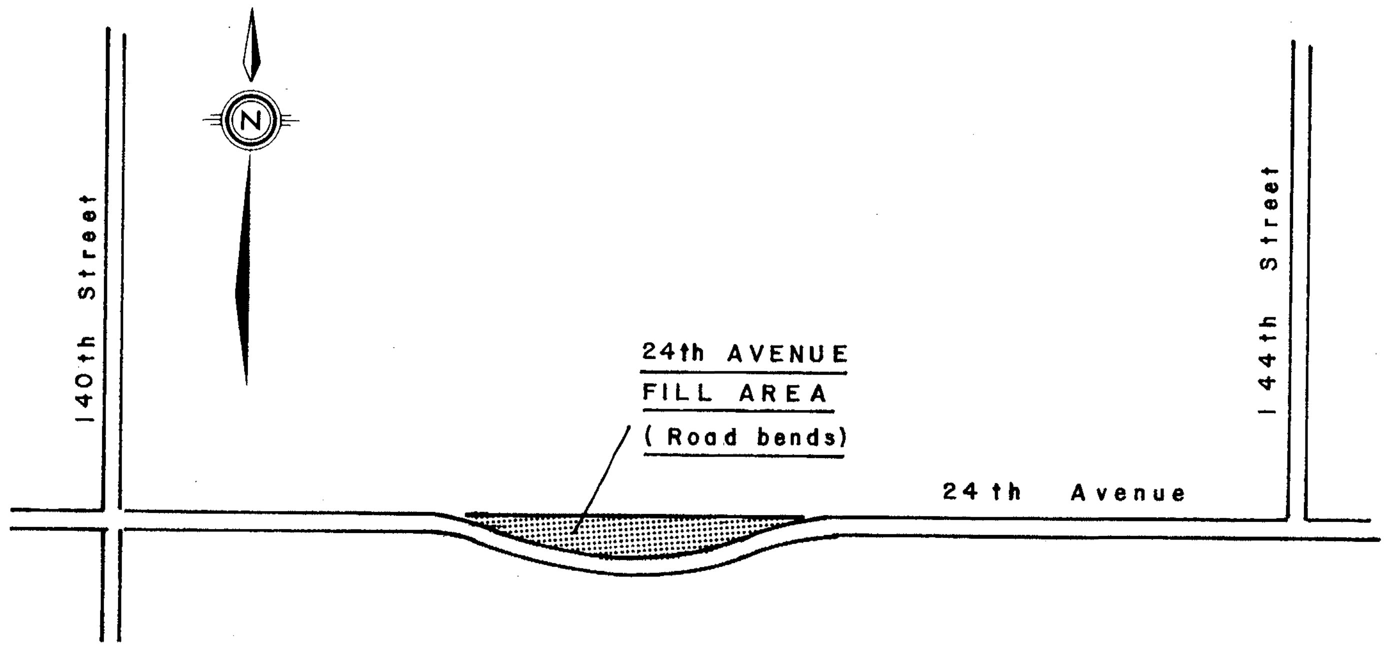


FIGURE E-12 SITE PLAN - 24TH AVENUE LANDFILL

APPENDIX F

LEACHATE TOXICITY

LARGE ACTIVE AND

WOOD WASTE LANDFILLS

TABLE F-1 ACUTE TOXICITY - LANDFILL LEACHATE

Landfill	Date and Collection Agency	96 hr LC (%)	(Species*, Loading)	LT at 100% (min.)	(Species, Loading)	Other	Comments
Burns Bog	02/02/78 (EPS)	7.5	(RT, 1)	42	(RT, 1)		-raw leachate
	03/03/78 (EPS)	6.5	(RT, .5)				-raw leachate
	26/02/78 (EPS)	18	(RT, .09)				-raw leachate
	26/09/77 (EPS)			33	(RT, 1)		-drainage ditch, SW corner of fill
	1975 (Corbett, 1975)	7	(RT, 1.2)				-raw leachate
Port Mann	04/05/76 (FMS)	12	(RT, .40)	21	(RT, .40)		-standing pool of leachate, 20% mortality in control
	29/11/76 (FMS)	42	(RT, .46)	1920	(RT, .46)		-leachate entering Fraser River
	09/03/79 (EPS)	50	(RT, .50)				
	21/03/79 (EPS)	38	(RT, .50)				
Braid St.	25/02/76 (FMS)	36.5	(RT,.35)			20 coho fry/10 litre, 100% mortality at 100 conc. in 48 hrs. 60% mortality at 10% conc. over 96 hr.	-leachate
	30/03/76 (FMS)	24	(RT, .50)	50	(RT, .50)		-leachate -10% mortality in control
Richmond	15/11/77 (EPS)			510	(RT, .40)		-#8 ditch
	15/11/77 (EPS)			240	(RT, .40)		-east ditch
	15/11/77 (EPS)				(RT, .40)	100% mortality at 100% conc. over 96 hr.	-#8 NR spring
	15/11/77 (EPS)			15	(RT, .40)		-north perimeter spring
	15/11/77 (EPS)			15	(RT, .40)		-north perimeter ditch 1 m downstream of spring

* Species: RT = Rainbow Trout
C = Coho

Loading: gm/l

TABLE F-1 ACUTE TOXICITY - LANDFILL LEACHATE (Continued)

Landfill	Date and Collection Agency	96 hr LC (%)	(Species*, Loading)	LT at 100% (min.)	(Species, Loading)	Other	Comments
Richmond (Cont'd.)	24/06/76 (EPS)	44	(RT, .56)				-#8 ditch
	08/09/76 (EPS)	29.5	(RT, .47)				-#8 ditch
	26/07/76 (EPS)	75	(RT, .50)				-#8 ditch
	26/07/76 (EPS)	30	(RT, .50)				-east ditch
	26/09/77 (EPS)				(RT, 1)	100% mortality at 100% conc. in 16 hr.	-north peri- meter spring
	24/10/76 (EPS)	42	(RT, .30)				-east ditch, SE gate
	24/10/77 (EPS)	42	(RT, .30)				-east ditch, Weir #2
	24/10/76 (EPS)	36	(RT, .30)				-east ditch, Weir #1
	Aug-Sept/75 (EVS, 1975)	X̄=49.6 range (35-95)	(RT, .5-1)				-#8 ditch, Stn. #15
	Aug-Sept/75 (EVS, 1975)	X̄=22.5 range (14-28)	(RT, .5-1)				-#8 ditch, Stn. #1
	01/01/78 (UBC)	4.2	(RT, .60)				-raw leachate, north peri- meter spring, sample stored 5 days at 2°C before assay. Strong H S smell.
	April to June/78 (B.C. Research)	30-90					

Species: RT - Rainbow Trout
Loading: gm/l

TABLE F-2 SUMMARY OF TOXICITY (96 Hours LC_{50}) DATA FROM WOOD WASTE LYSIMETER STUDIES (Cameron, 1975b) AND DFE FIELD COLLECTIONS

Location	Date	Fish Loading Density (g/l)	pH	Toxicity 96-hour LC_{50}
Lysimeter (Cameron, 1975c)	1975	-	-	0.48 to 4.0%
School House Creek Area - Coquitlam	May 18/78	0.4	6.0	LT 16 hrs.
School House Creek Area - Coquitlam	Feb. 7/78	0.6	6.4	Non toxic
Mayfair Industrial Park	July 5/78	0.4	6.9	75%
Tilbury Area Delta	July 19/78	0.7	6.3	Non toxic
Sunbury Area Delta	July 19/78	0.7	5.7	75%
Sunbury Area Delta	July 19/78	0.7	4.5	13.5%
North Surrey	Feb. 27/76	-	6.1	Non toxic
North Surrey	Mar. 30/76	-	6.3	32%
Near Scott Creek, Coquitlam	Apr. 6/77	-	6.6	Non toxic
Near Scott Creek, Coquitlam	Apr. 6/77	-	4.4	10%
South end of Queensborough Bridge	Feb. 26/76	-	5.4	Non toxic
South end of Queensborough Bridge	Mar. 30/76	-	6.1	Non toxic
Cowichan Bay	Jan. 1974	1.7	$\overline{X}$ = 6.7	$\overline{X}$ = 6.2%
Cowichan Bay	Feb. 1973	3.12	$\overline{X}$ = 6.8	$\overline{X}$ = 2.0%

APPENDIX G

EXCERPTS FROM

"THE AQUEOUS CHLORINATION OF ORGANIC COMPOUND: CHEMICAL REACTIVITY AND EFFECTS ON ENVIRONMENTAL QUALITY" (PIERCE 1978)

CHLORINE INTERACTION WITH NON-NITROGEN -CONTAINING ORGANIC COMPOUNDS

Activated aromatic compounds of low molecular weight, such as phenolic compounds, readily react with HOCl and hypochlorite in dilute aqueous solution. Four generalized reactions may occur: i) electrophilic aromatic substitution to form chlorophenols and related chlorinated derivatives; ii) oxychlorination (electrophilic aromatic addition and oxidation) to form chlorobenzoquinones and chlorinated muconic acid derivatives; iii) electrophilic addition to form chlorinated cyclohexene derivatives, such as chlorohydroxycyclohexanones, chlorohydroxycyclohexenones and chlorocyclohexadienones; and iv) intramolecular rearrangements subsequent to one or more of the above reactions to form chlorohydroxylactones and chlorohydroxycyclopentenes. Other low-molecular-weight aromatic compounds, such as substituted benzenes, biphenyls, naphthalene, fluoranthene an anthracene, may be chlorinated; usually one or two chlorine atoms are incorporated into the aromatic nucleus. The formation of chlorinated derivatives from these aromatic compounds requires relatively large doses of chlorine, extended reaction times, non-neutral pH values, and improper mixing to create localized high concentrations of reactants. Chlorohydrins are produced by the reaction of HOCl or hypochlorite with activated olefinic compounds. Chlorinated products are readily formed by the photolysis of relatively unreactive organic compounds (e.g. ethanol, n-butanol and benzoic acid) in the presence of HOCl. Chlorocatechols, chloroguaiacols, chlorobenzoquinones and chlorinated aliphatic compounds, such as chloromuconic acids and trichloromethane (chloroform), may be produced by the reaction of HOCl hypochloride or Cl_2 (aq) on the hydroxy and methoxy-substituted phenylpropane units of lignin. The interaction of chlorine dioxide and lignin or lignin-model compounds yields primarily non-chlorinated oxidation products, however some chlorinated phenolic benzoquinone derivatives have been isolated. Trihalomethanes may be produced by the interaction of HOCl with methyl ketones, polyhydroxybenzene constituents of humic material, organic compounds containing the methylene group flanked by carbonyl groups, alcoholic groups which may be oxidized to keto groups, m-oxysubstituted aromatic compounds and acetogenins. Of significance is the production of trichloromethane from compounds containing the pyrrole ring, such as typtophan, proline and chlorophyl. Increased trihalomethane levels in chlorinated drinking water have been associated with increased chlorophyl levels due to algal growth in raw water. Concentrations of trihalomethanes are relatively low when chlorine is added to water containing ammonia. Little information exists on the chemical interaction of monochloramine or other N-chlorinated compounds and organic compounds. Research on the reactions between chlorine species comprising combined available chlorine and organic compounds should be carried out, especially under photolytic conditions.

There is a paucity of information on the chemical reactivity of halogens other than chlorine towards organic compounds. If alternatives to chlorine are to be considered for disinfection and other purposes, more research must be carried out on the chemical speciation and reactivity of these alternatives in the presence of organic precursors.

CHLORINE INTERACTION WITH NITROGEN-CONTAINING ORGANIC COMPOUNDS

Little information is available on the chemical interactions of nitrogen-containing organic compounds, such as amines, a-amino acids, N-heterocyclic compounds, etc., and aqueous chlorine. Cyanogen chloride (CNCl) may be formed during the destruction of cyanide wastes by alkaline chlorination. Primary and secondary amines, amino acids and proteins are converted to N-chloro derivatives in the presence of aqueous chlorine. Tertiary amino compounds undergo oxidative cleavage, resulting in the production of N-chlorinated secondary amines and carboxylic acids. Amino acids undergo N-chlorination, oxidative deamination and decarboxylation. Nitriles, aldehydes and benzoquinones, some of which may be chlorinated, are produced by the reaction of HOCl or hypochlorite and the appropriate a-amino acid. N-heterocyclic compounds are N-chlorinated and may also undergo chlorine substitution at sites within the molecule. Pyrimidines, such as uracil, cytosine and thymine, yield chlorinated derivatives in the presence of HOCl or hypochlorite. The formation of chlorinated derivatives of purines in dilute aqueous chlorine solutions has not been demonstrated, although chloropurines have been identified in chlorinated water and wastes. Chlorinated nucleic acid fragments, such as 5-chlorouracil and 5-chlorocytosine, are formed from the reaction of HOCl or hypochlorite and nucleic acids.

APPENDIX H

VARIANCE IN BIOLOGICAL SENSITIVITY TO BIOASSAY

Sinley et al (1974), reported a higher tolerance to zinc by eyed eggs than by juvenile rainbow trout in hard and softwater, and that fish eggs exposed to zinc are more resistant than those not previously exposed to zinc as eggs. Russo et al (1974), report that rainbow trout, two gram trout and sac fry, exhibited greater tolerance to nitrite than the larger fish (235 g maximum) for the same exposure period. Smith and Williams (1974) reported a similar result with rainbow trout fingerlings (4.5 g) having a higher tolerance than yearlings (100 g). Thurston et al (1978) reported in unpublished data that rainbow trout parr are more tolerant to ammonia than are the juveniles or adults.

Burkhalter and Kaya (1977) demonstrated that the growth and development of rainbow trout sac fry are inhibited by long-term exposures to concentrations of ammonia as low as 0.05 mg/l (undissociated ammonia) as a suggested maximum safe continuous exposure level, but 0.005 ppm is probably safer for salmonids.

Schreck and Lorz (1978) reported that exposure of juvenile coho salmon to copper produced a marked, dose-dependent serum cortisol elevation, but this was not found with exposure to cadmium. The salmon exposed to copper had depressed serum chloride levels and reduced survival when challenged with salt water, but this was not found with cadmium.