LOWER FRASER RIVER/ESTUARY DISSOLVED OXYGEN DYNAMICS

by

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ABSTRACT

This investigation into the nature of dissolved oxygen dynamics in the lower Fraser River/Estuary has made use of the application of two mathematical water quality models - a tidally averaged dissolved oxygen model and a tidally varying dissolved oxygen model. The tidally averaged model analyzes the inter-tidal behaviour of the river/estuary, giving estimates of steady-state dissolved oxygen response. The tidally varying model, on the other hand, analyzes conditions within the tidal cycle, thereby describing the "real-time", intra-tidal behaviour of the river/estuary. Both dissolved oxygen models are one-dimensional and make the assumption that the only operative dissolved oxygen source/sink processes are deoxygenation due to the oxidation of discharged organics and reoxygenation due to atmospheric reaeration.

The present high dissolved oxygen levels in the lower Fraser preclude the accurate calibration of the dissolved oxygen models. However, an analysis of model sensitivities is presented, in lieu of verification, to document model responses.

Dissolved oxygen predictions made using the unverified models indicate that the assimilative capacity of the lower Fraser River/ Estuary is considerable, mainly because of the large freshwater inflows which afford extensive dilution as well as rapid flushing. The "critical period" is likely to be in late summer when the combined effects of water temperature and freshwater flows result in the lowest dissolved oxygen levels. Future water quality impairment in the main

channels of the lower Fraser, at least insofar as dissolved oxygen is concerned, is considered by this study to be unlikely, providing that existing pollution control policies are adhered to.

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INTRODUCTION

> "no man stands beside the Fraser River without sensing the precarious hold of his species upon the earth... here it is thrust upon you with a special clarity. In this grisly trench, bored out of solid rock through unimaginable time by the scour of brown water, the long history of lifeless matter, the pitifully brief record of life, the mere moment of man's existence, are suddenly legible. And here, in this prodigal waste of energy, nature's war on all...is naked, brutal, and ceaseless."

So begins Bruce Hutchinson's book, The Fraser, describing the Fraser River, its impact on man, and its effect on his environment. He continues to expound the importance of the river as "one of the basic political and economic facts of America...little understood by governments and seldom mentioned in history books".

Being mindful of the fact that the two decade-old description is an over dramatization by a native British Columbian journalist does not lessen the importance of the Fraser River and the role that it has played in British Columbia, its historical, social and economic development. Rail routes along the Fraser passage through the formidable Coast range completed around the turn of the century tied British Columbia into the rest of the nation and ultimately led to the development of the port city of Vancouver, the main port and urban centre on the west coast. A third rail route following the upper lengths of the Fraser corridor connects Vancouver to the northern parts of the province. Vancouver is thus a major node in a transportation network with two railway lines stretching over the entire nation and the third out to the hinterland of the province. It is ironic to see that the development of the urban centres and industry initiated through use of the river as a transportation corridor now have the potential to destroy that same river through their use of it as a sink for their wastes.

Domestic wastes from most of metropolitan Vancouver, representing a population in excess of one million are discharged into the lower Fraser River and its estuary, as are wastes from many other smaller urban centres scattered throughout the lower Fraser Valley. In combined total, the lower Fraser receives the domestic wastes of 1,100,000 persons, which represents about 50 percent of the population of British Columbia. In addition to the domestic waste inputs, large quantities of industrial wastewaters enter the river from numerous industries located on the river foreshore which, for the most part, is zoned for industrial development. Rapid growth of population in the lower mainland, which has occurred in recent years, along with associated expansion in industrial and commercial development is expected to continue. Projections have been made, indicating that by the year 2000, the lower mainland population will be 2,400,000 [LMRPB, 1968] with the major portion being concentrated in and around metropolitan Vancouver.

In recent years, pollution in the lower Fraser River has been an issue of particular concern. Among other reasons which included the contemporary ethic of "environmental awareness", pollution has been a particular concern because of the threat posed by the degradation of water quality to the, as yet, intact Fraser River salmon fishery. This unique, natural resource, which has cultural as well as economic significance, has been estimated to have an annual value in the order of $75 million [Fisheries

Service, 1971].

Deterioration of water quality caused by such things as dissolved oxygen depletion due to the assimilation of organic waste discharges and the toxicological effects of toxic discharges has serious implications not only with respect to the salmon fishery but also with respect to other uses of the river which may be impaired by polluted conditions. As a result of the general concern over water quality in the lower Fraser River, considerable attention has been directed towards assessment of water quality conditions and the formulation of water quality management policies. An investigation by the provincial Pollution Control Branch [Goldie, 1967] is worth noting, particularly because it formed the basis of adopted provincial government policy regarding pollution control on the Fraser River below the town of Hope [PCB, 1968]. The primary objective of this policy was "the maintenance of the lower Fraser River as a multi-purpose water resource for the people of the province for all time". More specifically, this objective was "to maintain the river free of harmful pollution and toxic substances in areas where the river is not so polluted...and...to bring about an improvement in the state of the river in areas where pollution has already occurred".

The findings of the Goldie report [1967] on waste disposal in the lower Fraser were, in summary, that dissolved oxygen levels in the main stem Fraser were high and thus the Fraser could be regarded as a "clean stream in terms of BOD" but that bacterial contamination was "undesirably high...and...continues to increase in some areas". In recognition that the lower Fraser's capacity to assimilate wastewater was not sufficient "to accept without danger of impairment the wastes of the foreseeable valley population and attendant industries" and to offer "adequate protection against impairment and excessive bacterial contamination", the report recommended as a general rule that "all sewage discharges to the lower Fraser River should first receive primary treatment and chlorination". This recommendation was, in large part, accepted by the Pollution Control Board as a principal requirement in its policy statement [PCB, 1968] to fulfill the general objectives for the lower Fraser.

The results of more recent research investigations into water quality in the lower Fraser [BCRC, 1971; Benedict et al., 1973; and Hall et al., 1974], in addition to having more completely defined the nature of the water quality conditions, have, in general, confirmed the findings of the Goldie report with regard to the dissolved oxygen levels and bacterial contamination. It is the former water quality parameter to which this thesis is addressed.

The basic objectives of the research described in this thesis were:

(i) to investigate the mechanics of dissolved oxygen dynamics in waterways;

(ii) to apply these concepts to the development of capabilities for predicting dissolved oxygen concentrations in the lower Fraser River;

(iii) to assess the validity and suitability of these predictive capabilities in their application to the Fraser River;

(iv) to assess, through use of the predictive capabilities, the probable impact of various waste discharge patterns on dissolved oxygen levels in the lower Fraser River.

This investigation into the development, assessment, and utilization of predictive capabilities for dissolved oxygen can be classed as a "model study" because it makes use of digital computer "models". The computer programs which form the basis of the dissolved oxygen "models" employ mathematical abstractions of the relevant processes as they seek to emulate the system they represent. Although this procedure of approximating a physical system by mathematical abstractions is often referred to as a "simulation" of the physical system it is stressed here that this is rarely if ever true. In fact the output of a "model" corresponds to the response of the real system only as the model assumptions represent reality and the output is as good as the weakest model assumption.

At its conception this research project was envisioned as being designed specifically to investigate the effects of sewage discharge from the proposed treatment facility at Annacis Island on the oxygen resources of the river, a "sister" project to the investigation undertaken by Rusch [1973] to study the effects of the proposed discharge on bacteria levels in the river. In the interim since project conception, during which the author has been employed with the Westwater Research Centre at U.B.C., and been associated with the development of a number of water quality models, the research aims of this project have undergone a degree of metamorphosis to become broader in scope. Rather than focusing on the analysis of a specific problem as originally intended this thesis has tackled the more general issue of understanding the nature of dissolved oxygen dynamics in the lower Fraser River. In doing this it has remained true to the original purpose of the research but more completely fulfilled the stated objectives. As well, it has hopefully offered some additional insights into the complex nature of the lower Fraser River and its considerable ability to assimilate organic wastes that may have gone undetected by the original project design.

The organization of thesis presentation can be thought of as having five distinct components - description, theory, application, results and conclusions. Chapter 1 is purely descriptive, presenting information on the details of the Fraser River relevant to dissolved oxygen model studies. These include run-off characteristics, water temperature characteristics and, for the lower Fraser, the effects of tidal influence and saltwater intrusion. Chapters 2 and 3 have a more theoretical basis. Chapter 2 presents some of the concepts, theories and mathematical formulations developed to describe dissolved oxygen and the factors which affect its dynamic behaviour in waterways. Chapter 3 discusses methods of applying the fundamental formulations for dissolved oxygen to rivers and estuaries in order to develop modeling capabilities. Chapters 4 and 5 are concerned, respectively, with the application of the models to the Fraser River and the results of the modeling exercise. Discussion in Chapter 4 is centred around the details of model application and the various assumptions implicit in each of the models. Specific results are presented in Chapter 5 to test the performance of the models. In addition, the results of various model runs are presented to enable a preliminary assessment of assimilative capacity and possible future dissolved oxygen levels in the lower Fraser River. Chapter 6 summarizes the investigation discussing and assessing the validity of the models, their utility and the results of their application. Chapter 7 draws conclusions and suggests ways of improving and strengthening the developed predictive capabilities.

CHAPTER 1

THE FRASER RIVER

1.1 THE FRASER RIVER

The Fraser River Basin, an area of some 90,000 square miles, is the largest river basin wholely within the province of British Columbia. Covering approximately one-quarter of the province, the drainage basin occupies the greater portion of the southern half of the province, draining the high central Interior Plateau which is flanked on the west by the Coast Mountains and by the paralleling Columbia and Rocky Mountains on the east (see Figure 1.1). The Interior Plateau and the rugged mountainous country surrounding it are characterized by their high elevation with more than 70% of the drainage basin being above 3,000 feet, and 10% being above 6,000 feet. [Fraser River Board, 1958].

The river itself rises in the Rocky Mountains at one of the most easterly points in the basin near the Yellowhead Pass, descends into the Rocky Mountain Trench, and flows northwesterly for some 250 miles before it turns south, crossing more than 400 miles of the Interior Plateau. At Lytton, the Fraser forms a spectacular canyon as it cuts through the formidable Coast Range. Continuing southward to Hope, the river breaks out of the canyon and turns westward, making its approach to the sea through 100 miles of the alluvial, deltaic Lower Fraser Valley. The "mighty" Fraser, by the time it enters the sea through the Straits of Georgia, has traversed a total length of approximately 850 miles.

Of the total Fraser River drainage area, approximately 52,000

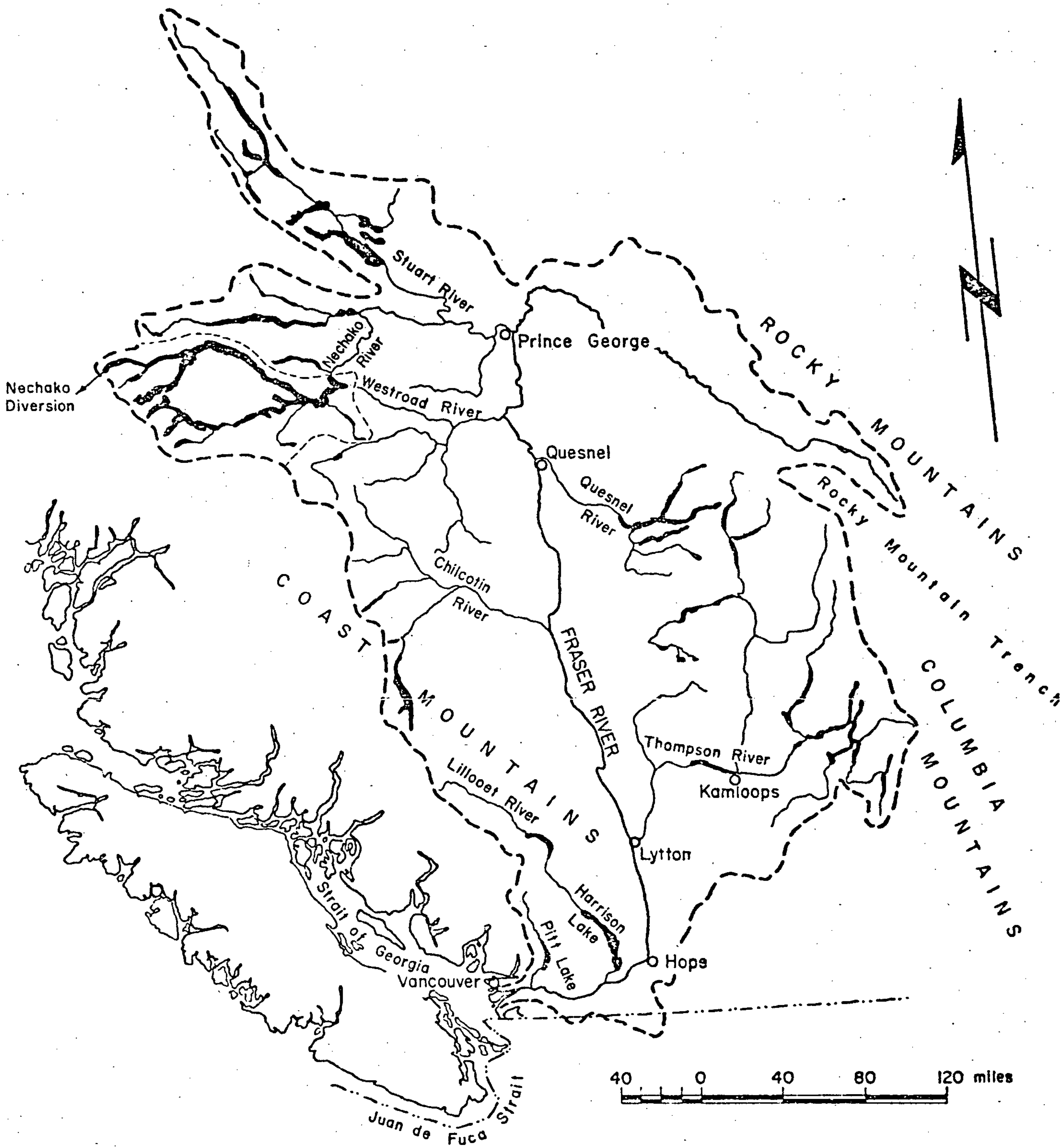


Figure 1.1
Fraser River Drainage Basin

square miles are contained in tributary sub-basins, the most important of which are shown in Figure 1.1. The Stuart, Nechako, Westroad and Chilcotin Rivers drain the westerly portions of the Interior Plateau and the easterly slopes of the Coast Mountains with the Quesnel and Thompson Rivers, on the east, draining the remaining portions of the plateau and the eastern slopes of the Columbia Mountains. Other than a development diverting 5,400 square miles of the Nechako sub-basin and a minor sub-basin development at Bridge River, the Fraser and its tributaries are not, as yet, dammed, although schemes for both flood control and hydro-electric power have been suggested. [Fraser River Board, 1958; B.C. Energy Board, 1972].

Climatic conditions vary considerably over the basin. Alpine maritime climate characterizes the lower portions of the drainage basin and as one proceeds up the basin climatic conditions change through dry and humid continental to alpine humid continental, characteristic of the head-waters and easterly areas. The extreme northerly portions of the basin border on alpine subarctic climatic regions. Annual precipitation also fluctuates markedly throughout the basin, ranging from rain forest precipitation levels of over 150 inches per year in the vicinity of Pitt Lake in the Lower Fraser Valley to arid and semi-arid levels of 5-10 inches per year in areas of the central plateau near Kamloops. Variability of vegetation is also extreme. Paralleling precipitation, it ranges from west coast rain forest to almost desert-like vegetation consisting of arid and semi-arid grassland in the interior. [Fraser River Board, 1958].

Temperature, along with precipitation, is a major climatic factor which affects the hydrology of the basin. The temperature at any point in the basin depends primarily on altitude, latitude and distance from the Pacific Ocean. Of particular significance is the fact that temperature over the whole of the Fraser River basin normally falls below freezing in the winter months. With the spring thaw, which usually begins in the south and spreads northward, precipitation which has been stored over winter in the form of snow and ice is released, causing freshet conditions which commonly result in flooding. Exceptional floods, as in 1948, are caused when abnormally high spring precipitation combines with rapid snowmelt. [Fraser River Board, 1958].

Throughout its length the main stem Fraser drops over 4,000 feet in elevation with a relatively uniform grade indicative of a maturing river, except in the headwaters where gradients are high, and in the Lower Fraser Valley where there is a sharp break in river profile. In the vicinity of Chilliwack, at the break of grade, the river deposits the major portion of its large sediment load, an estimated 25 million tons annually, [Pretious, 1972] which, as well as making the river turbid and brown in appearance, has led to the formation of the alluvial lower valley. From Chilliwack downstream to the sea, a distance of 55 miles, river grades are very low and as a result water slopes and river stage are affected by tides.

1.2 FRASER RIVER RUN-OFF CHARACTERISTICS

The runoff characteristics of the Fraser River basin are best described by the records of the gauging station at Hope, where the river flow is unaffected by tides, the river cross-section is stable and the period of record stretches back to 1912 [Water Survey of Canada, 1913 to 1970]. As well, this station integrates the effects of the entire basin, representing about 87% of the total drainage area and 80% of the estimated total runoff of the Fraser River to the sea [Fraser River Board, 1958].

An average annual hydrograph of Fraser River discharge at Hope is shown in Figure 1.2. Although its period of record is outdated, it shows the observed pattern of runoff and the influence of snowmelt on the runoff regime. Freshet typically begins near the end of April, peaks in mid-June and tails off to base flow conditions in late fall. Minimum annual low flows occur in the winter months, December to March. Low and high extreme flows recorded at Hope were 12,000 cubic feet per second (cfs) on January 8, 1916 and 536,000 cfs on May 31, 1948. An even larger flood occurred in 1894, with peak flow reaching an estimated 620,000 cfs.

The variation and distribution of mean monthly flows for the Fraser River at Hope over the period of record from 1913 to 1970 is shown in Figure 1.3. This summary information is based on flow analysis carried out by the Westwater Research Centre [Westwater, unpublished data]. For each month, median flows, along with five, ten and fifteen year return flows are reported. Discharge data in this form are useful not only as

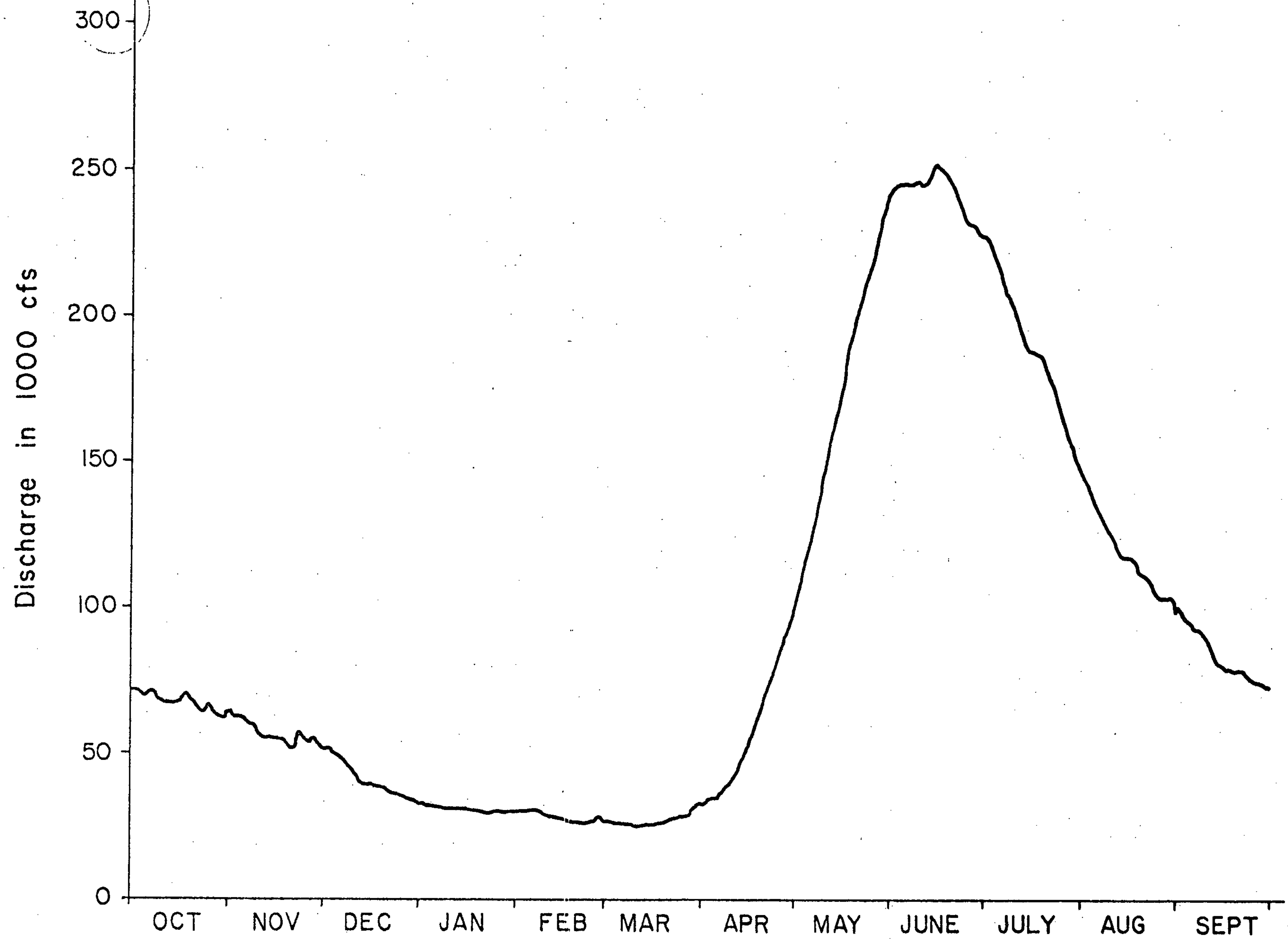


Figure 1.2

Average Annual Hydrograph for the Fraser River at Hope (1930-1960)

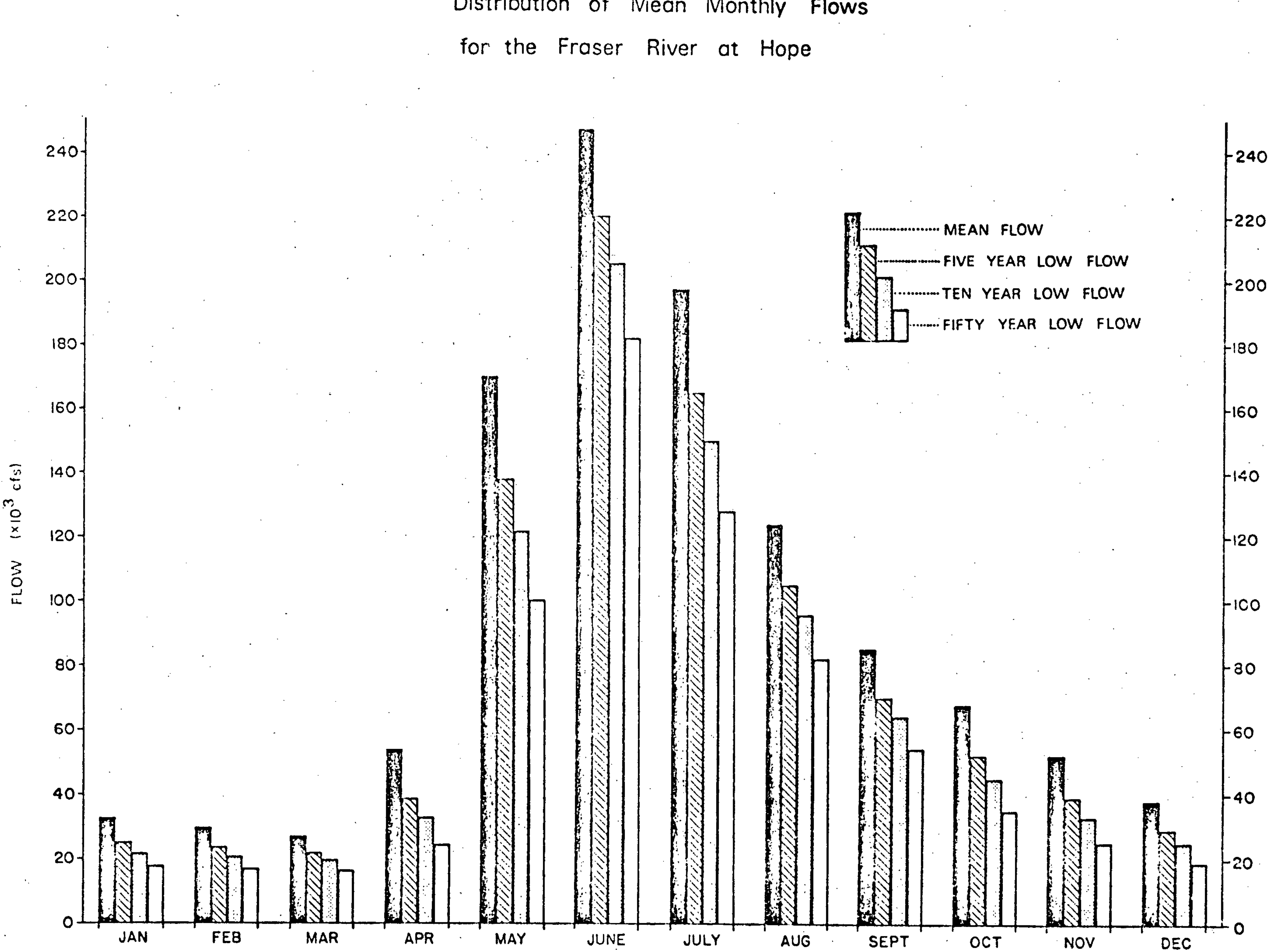


Figure 1.3
Distribution of Mean Monthly Flows for the Fraser River at Hope (1913-1970)

input information to the water quality models, as will be discussed in a subsequent section, but also, because the scope of the information is the broader monthly time base, it allows for a more tempered view of the range of Fraser River flows.

Results of an analysis of low flow conditions [Westwater, unpublished data] are shown in Figures 1.4 and 1.5. Figure 1.4, representing analysis of 7-day low flow data, is based on a recent publication by the Water Survey of Canada [WSC, 1974]. The 7-day low flow has been recommended by some researchers as the time period most suitable for study of water quality and the examination of water quality effects [McKee and Wolf, 1963]. The minimum yearly low flow distribution presented in Figure 1.5 has been included to round out the picture of low flow data for the Fraser.

1.3 FRASER RIVER WATER TEMPERATURE CHARACTERISTICS

The distribution of water temperature for each month at Hope is shown in Figures 1.6 and 1.7 and is based on mean monthly conditions for seven years of record as published by the Sediment Survey of Canada [1964-1970]. Distribution was assumed to be normal and it can be seen that the fit of the data points suggest that this is a reasonable assumption. Low temperatures occur during the winter months, with the lowest average temperatures being 1.0°C in January. There is considerable overlap in winter monthly mean temperatures between different months which reflects the influence of lower basin cold weather conditions, their severity and timing. Higher temperatures are observed in the summer period with August having the highest average temperature at 17.7°C.

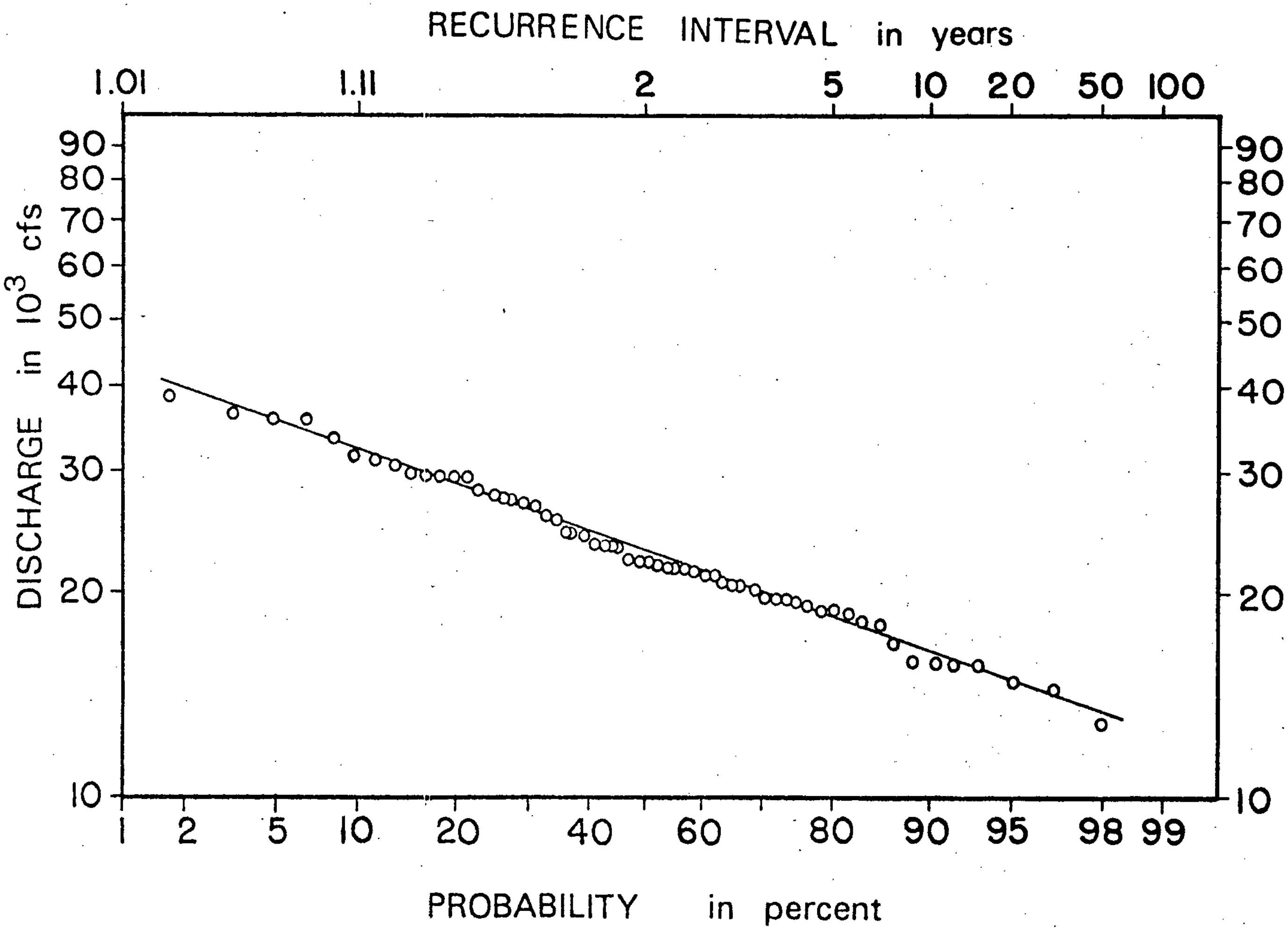


Figure 1.4

Distribution of 7-Day Low Flows for the Fraser River at Hope

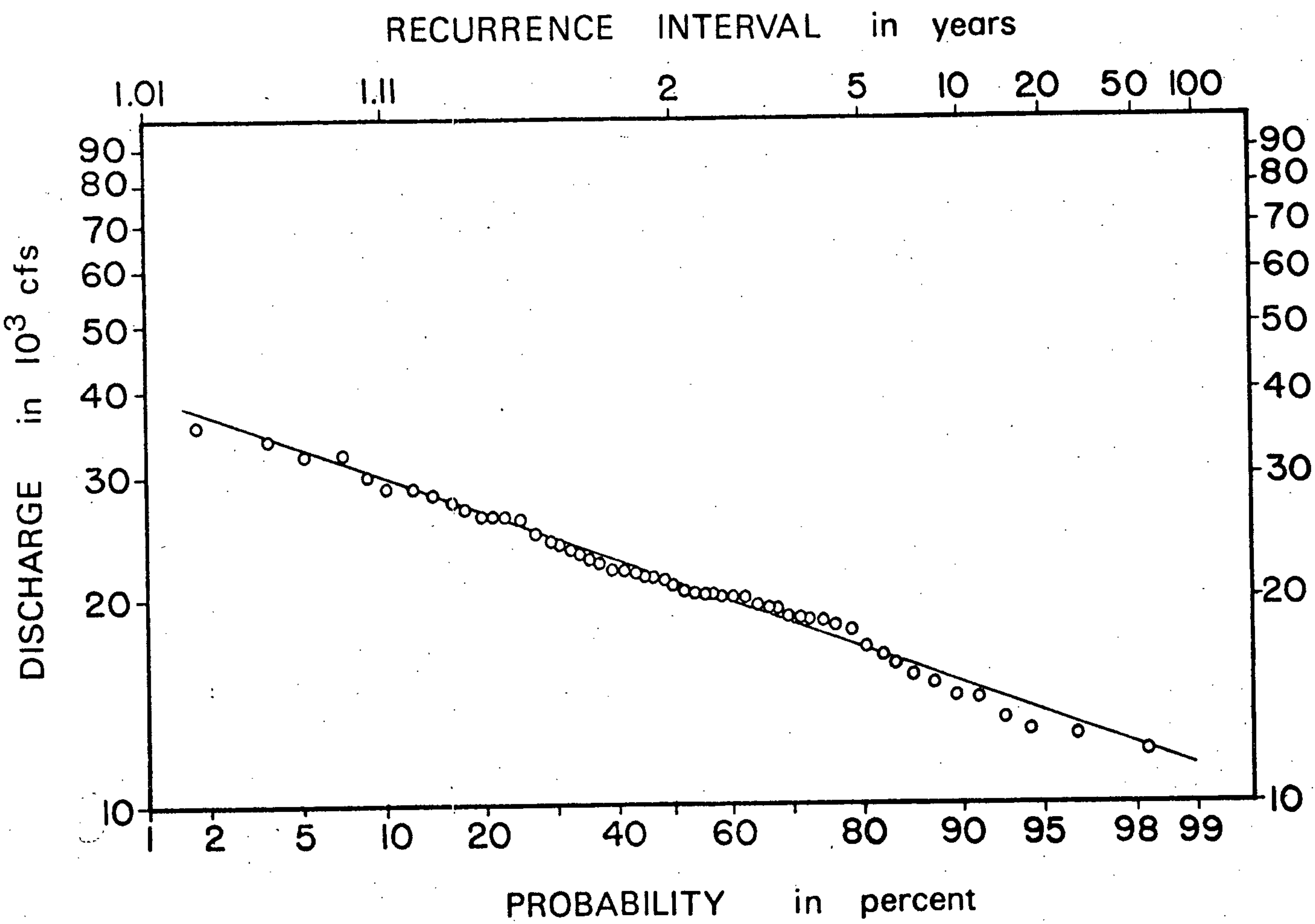


Figure 1.5

Distribution of Minimum Yearly Flows for the Fraser River at Hope

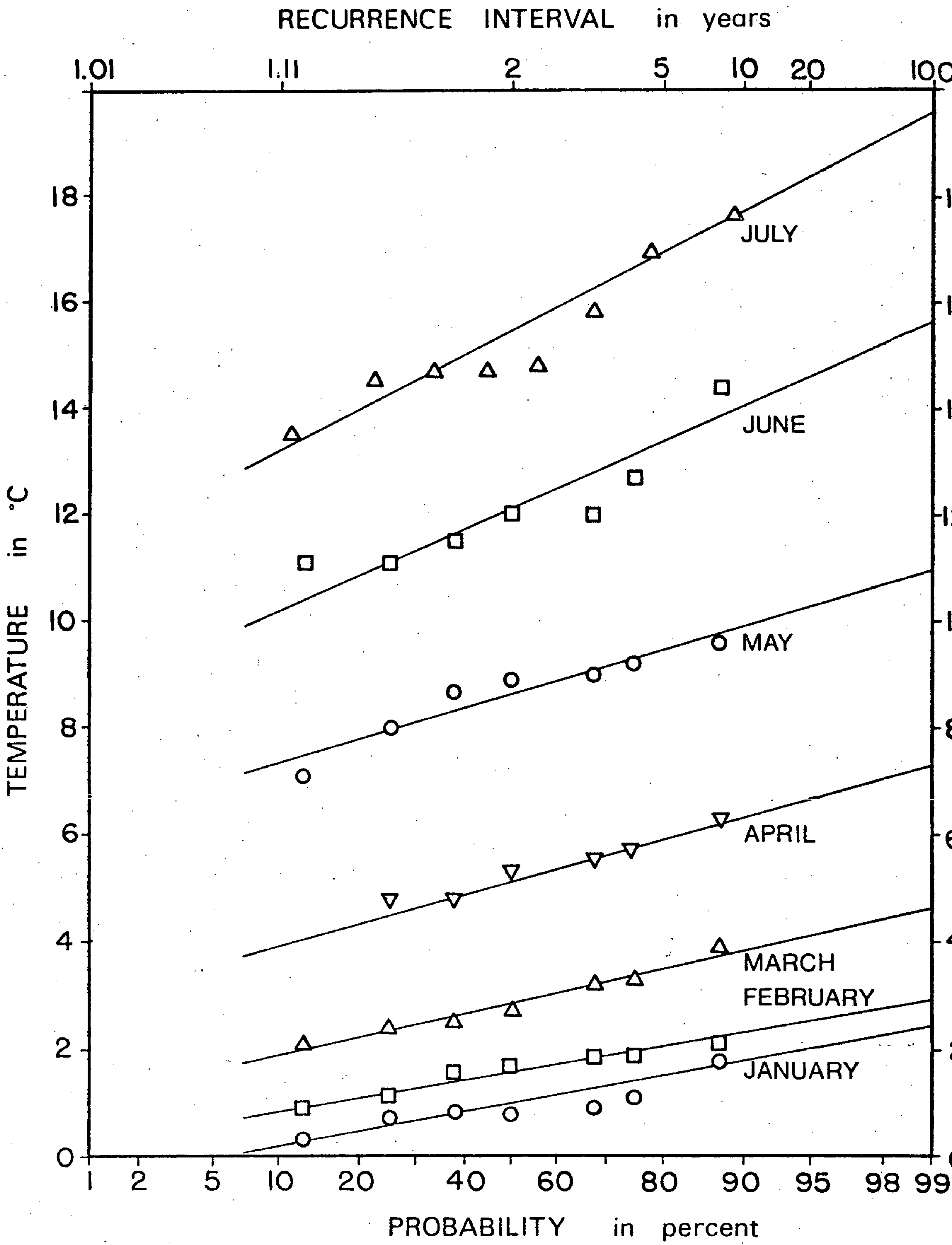


Figure 1.6

Distribution of Mean Monthly Fraser River Water Temperatures

January to July

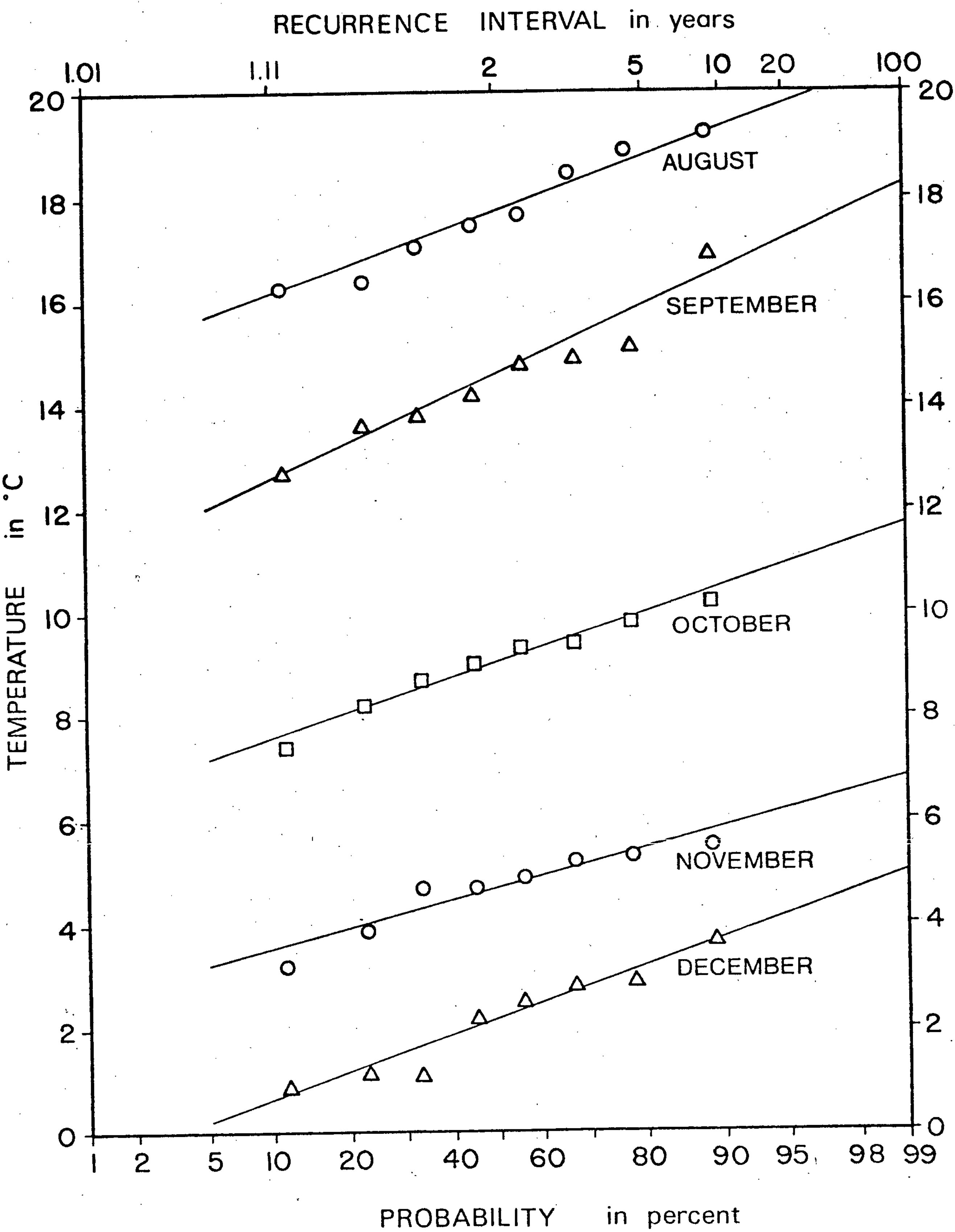


Figure 1.7

Distribution of Mean Monthly Fraser River Water Temperatures

August to December

1.4 THE LOWER FRASER RIVER

The lower Fraser River from Hope to the Strait of Georgia is shown in Figure 1.8. The drainage area of the lower river is approximately 6,000 square miles. Although this area represents only some six percent of the total catchment area, it can contribute significantly to the flows in the lower Fraser River, an estimated 15 percent during the freshet and as much as 50 percent during the winter months [Goldie, 1967].

Terminology used to describe the various stretches of the lower Fraser River is often confusing. The following brief description of the various channels of the river system delineates the nomenclature used throughout this paper [taken from Benedict et al., 1973].

The stretch of river from Hope to New Westminster is called the Main Stem. At New Westminster, the Fraser River branches into a major channel called the Main Arm, entering the Strait of Georgia at Steveston, and a minor channel known as the North Arm, which enters the Strait at Point Grey. In the North Arm, bifurcation caused by Sea Island results in the Middle Arm, which enters the Strait over Sturgeon Banks. A number of small islands and training walls in the Main Arm near Ladner result in the formation of side channels, the major ones being Ladner Reach and Sea Reach, which flow into Canoe Pass, the most southerly exit of Fraser River water. The Main Arm contributes 85 percent of the discharge that ultimately enters the Strait and the North Arm, Middle Arm and Canoe Pass contribute about 5 percent each [Goldie, 1967].

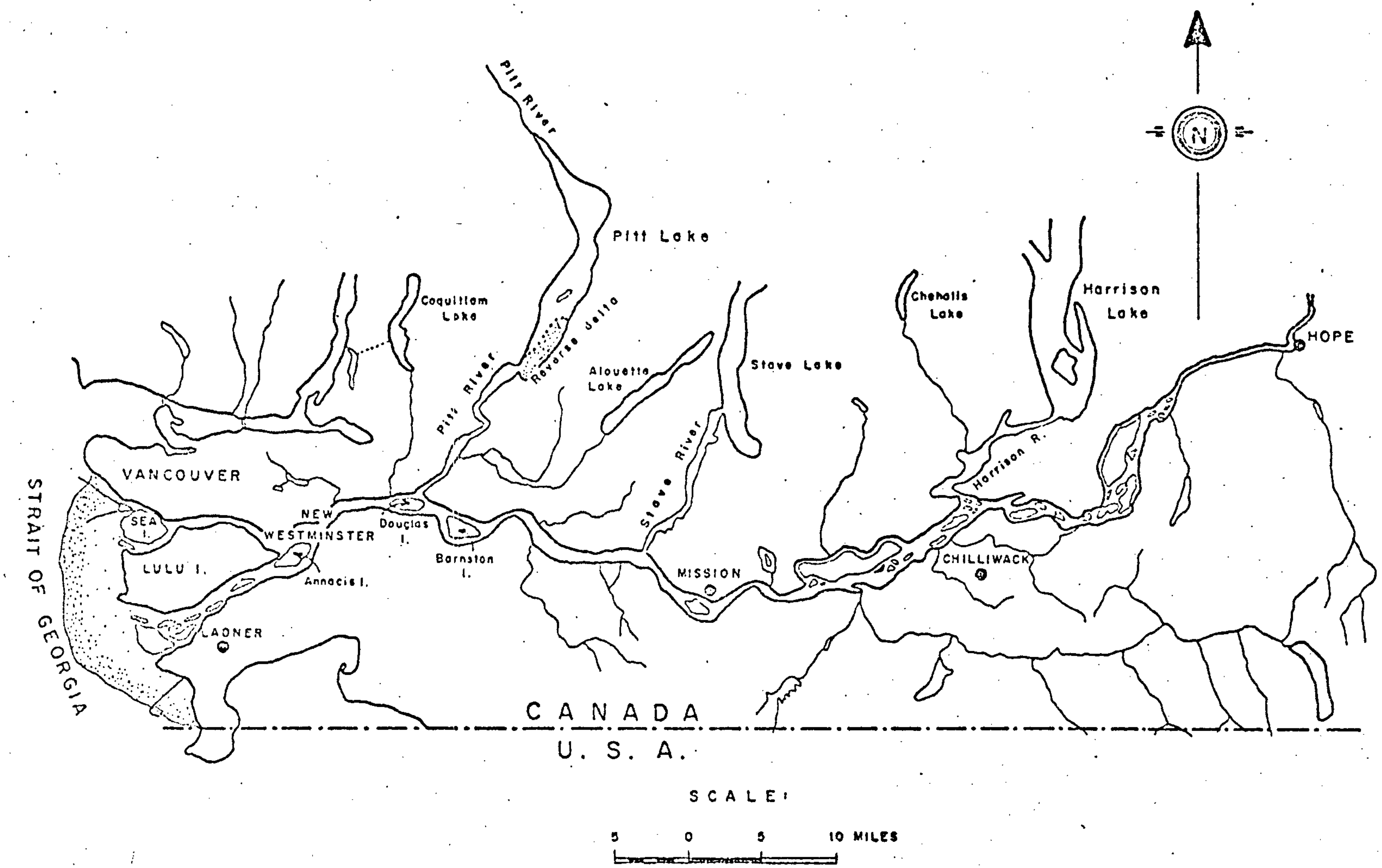


Figure 1.8

The Lower Fraser River

1.4.1 Tidal Effects. Tides in the Strait of Georgia, which is connected to the Pacific Ocean through Juan de Fuca Strait in the south and Johnstone Straits in the north (see Figure 1.1), are of the mixed type, characteristic of much of the coast of northwestern North America. That is, the tides alternate from spring to neap tides in a bi-weekly cycle resulting in diurnal inequalities most days of the cycle. The tidal range at Steveston for mean and large tides respectively is 10 feet and 15 feet. Typical tides are shown in Figure 1.9 for Point Atkinson, the nearest Hydrographic Services reference port [Canadian Hydrographic Services, 1974]. Low river slopes in the lower Fraser River, as previously mentioned, result in tidal action affecting water slopes and water surface elevation as far upstream as Chilliwack, some 55 miles from the mouth. On occasion, the combination of unusual tides and low river discharge has resulted in observed effects reaching even farther upstream to the vicinity of Rosedale, although this is exceptional [Baines, 1952]. The effects near the limit of tidal influence are restricted to minor changes in slope and elevation and, although discharge is affected, effects are not severe enough to cause current reversal. Current reversal has been reported, however, as far upstream as Mission [Baines, 1952].

Of additional importance in understanding the influence of the tidal effects on the hydraulic behaviour of the lower Fraser River is the recognition that the Pitt River-Pitt Lake system is a tidal storage area. Water surface elevations are observed to vary with the tides over the full extent of Pitt Lake, a surface area of some 25 square miles. This represents a large volume of water storage on the flood tide, which is later released on the ebb tide. There is a reverse delta at the entrance to Pitt Lake

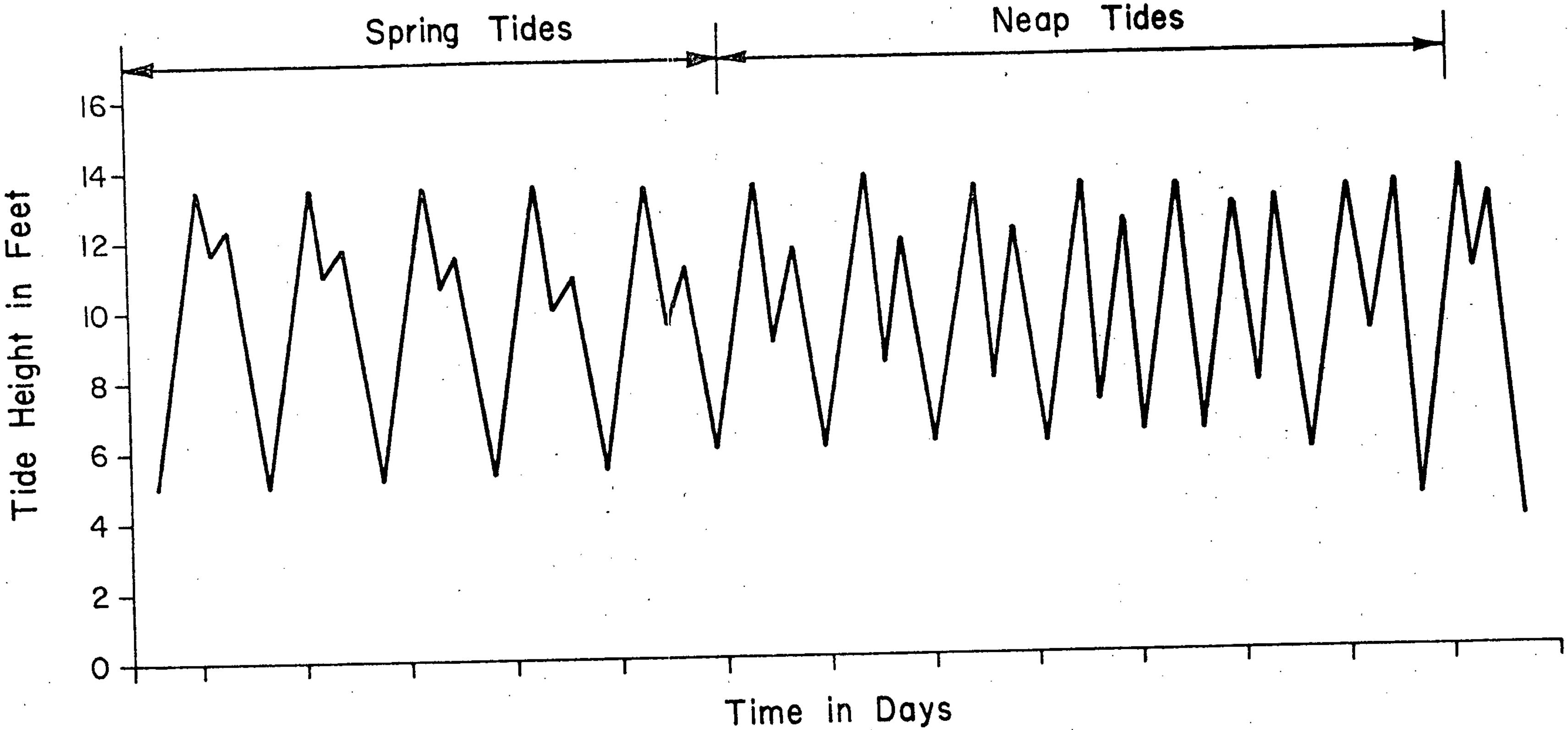


Figure 1.9

Typical Tides at Point Atkinson

which is evidence of the large volumes of water that enter the lake [Joy, 1974].

Low river flow-high tide conditions were the subject of a study by Baines [1952], referred to in the preceding paragraph. Cubature discharge calculations, based on one-half hourly river stage measurements for a day and a half, showed that instantaneous upstream tidal flows in the vicinity of Steveston reached nearly 140,000 cfs with downstream tidal flows reaching over 200,000 cfs for a freshwater discharge at Hope of 28,400 cfs (see Figure 1.10). The velocity variation over the same time period, also shown in Figure 1.10, was from 2.0 feet per second upstream to 3.8 feet per second downstream. This extreme variation in discharge and velocity makes one acutely aware of the complex nature of hydraulic behaviour in the lower Fraser River.

High river flow conditions during the freshet tend to dampen out tidal effects as the freshwater discharge predominates over tidal action. During these conditions there are no reversals of current and only minor river stage variation even in the seaward reaches of the river. Although flows at this time are still unsteady, the lower Fraser River behaves hydraulically in a manner more closely resembling that of a river.

Another consequence of tidal action is the increased complexity and unsteady nature of the physical processes of mixing and dispersion. Unlike the situation in a river where dispersion processes, although minimal, are more completely defined because insights and information are transferable from one river to another and the process can be reproduced

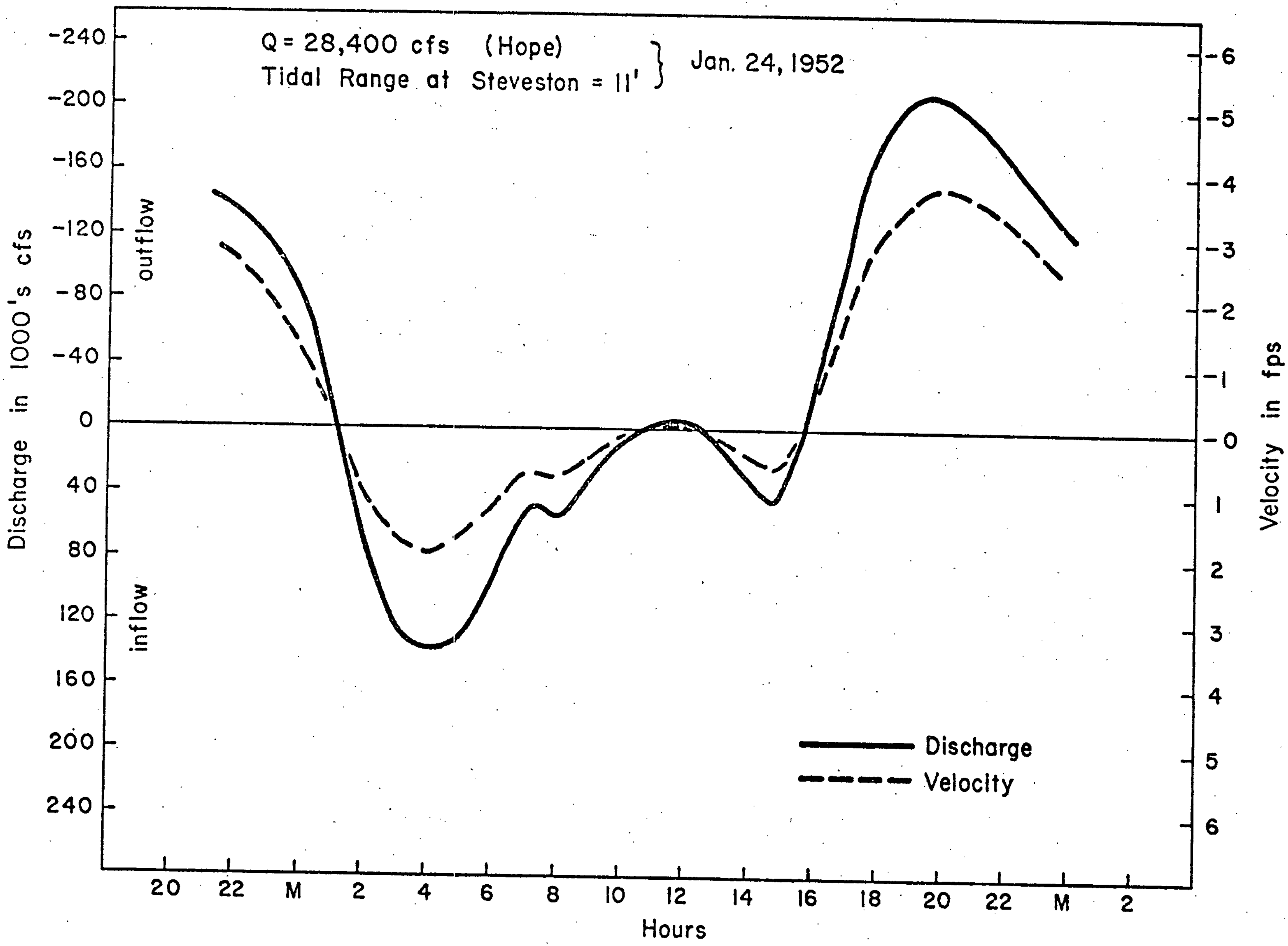


Figure 1.10

Discharge and Velocity Variation at Steveston Due to Tidal Influence in the laboratory, the phenomenon of tidal mixing in estuaries and tidal rivers is poorly understood. This is partly because it has received less attention, but mainly because each situation is unique and, as a result, the findings from other studies do not necessarily apply.

Mixing and dispersion processes on the lower Fraser River have been the object of a recent study by the Westwater Research Centre [Ward, unpublished data]. Dye tracer was released and tracked over a tidal cycle to determine rates - lateral, longitudinal and vertical mixing. The results of the dye studies indicated that, while vertical mixing normally occurs rapidly, the rates of lateral dispersion are much lower. Vertical mixing was calculated for unstratified flow to be complete within the first hour after dye injection, whereas the time required for transverse mixing to be completed was estimated to be more than one tidal cycle (or 25 hours). It was found, however, that when partially stratified conditions existed in the estuary due to the presence of the salt wedge, vertical mixing was inhibited.

1.4.2 Salinity Intrusion. Saltwater intrusion in the lower Fraser River has been the subject of much debate in recent years. Various spot measurements were made in the past [Waldichuk *et al.*, 1968] but, not until 1973, was any intensive research initiated to study this estuarine phenomenon. Field investigations during February-March, 1973 were undertaken to define the salinity patterns in the lower Fraser River [Hodgins, 1975]. In-situ continuous recording salinity and temperature measuring devices were installed at three fixed locations in the lower reaches of the river in an attempt to determine the nature and extent of the salt water intrusion. Results of this field investigation have been included in a doctoral thesis which describes a mathematical model of salt water wedge movement [Hodgins, 1975]. The salinity monitoring showed conclusively that a stratified salt water wedge was present in the lower reaches of the river during flood tide conditions. The tongue of the wedge, just after high tide, was estimated, through use of the saltwedge model, to extend nearly as far upriver as Annacis Island. With the succeeding strong ebb tide the wedge was washed out of the river to a point beyond Steveston. Recent salinity measurement in the river has confirmed that this is indeed the case [Ages & Hughes, 1975, and Westwater Research Centre, unpublished data]. Indications are that the toe of the wedge can extend upriver even as far as the eastern tip of Annacis Island.

During freshet, saltwater does not intrude upriver of Steveston due to the predominating effects of large river flows [Ward, unpublished data]. This situation then, with the saltwedge being washed out of the river during low freshwater discharge conditions and not entering the river at all during freshet discharge conditions, leads one to question the classification of the lower Fraser as an estuary. Since traditional estuary classification [Hansen and Rattray, 1966] implies that there be consistent presence of salinity, the system is perhaps more properly described as a tidal river, as it has been by some researchers [Callaway, 1971]. To account for the fact that salinity in the form of saltwater intrusion is present some of the time, the term "river/estuary" will be used in this paper.

CHAPTER 2

DISSOLVED OXYGEN DYNAMICS, A REVIEW

2.1 DISSOLVED OXYGEN

The dissolved oxygen resources of a waterway play a most important role in the maintenance of a healthy aquatic ecosystem. Dissolved oxygen, one of the most important indicators of water quality, is essential for support of a balanced aquatic habitat particularly as it affects the survival of fish life. To insure the survival of a healthy fishery, which will essentially guarantee protection of the entire aquatic community, concentrations of dissolved oxygen must generally be above 5 mg/l (milligrams per litre) although oxygen requirements may vary depending on the age and species of the fish, the temperature and composition of the water, and the presence of toxic substances [Klein, 1962].

More specific dissolved oxygen criteria for freshwater fish [FWPCA, 1972] are given as follows. For warm water game fish "DO concentrations should be above 5 mg/l ... (except) under extreme conditions ... the DO may range between 5 mg/l and 4 mg/l for short periods of time." More stringent requirements are recommended for cold water fishes such as trout and salmon especially in spawning areas where "DO levels must not be below 7 mg/l at any time." For good growth and the general well-being of these species "DO concentrations should not be below 6 mg/l ... (except) under extreme conditions they may range between 6 mg/l and 5 mg/l for short periods." In large streams which serve principally as migratory routes "DO levels may be as low as 5 mg/l for periods up to

6 hours but should never be below 4 mg/l at any time or place."

Aside from being a necessary requirement for fish survival, dissolved oxygen concentration is also a more general indicator of the degree of water pollution as it is linked with other water quality impairments such as the nuisance conditions created where concentrations are low. A classification of river quality based on dissolved oxygen content is shown in Table 2.1.

TABLE 2.1

CLASSIFICATION OF RIVER QUALITY BASED ON DISSOLVED OXYGEN CONTENT [KLEIN, 1959]

TYPE OF RIVER WATER	DISSOLVED OXYGEN % OF SATURATION
Good	Greater than 90
Fair	75 - 90
Doubtful	50 - 75
Badly polluted	Less than 50

The amount of oxygen dissolved in water from the atmosphere is dependent on temperature, barometric pressure and the amount of salts present in the water. Solubility of oxygen varies directly with barometric pressure and inversely with water temperature and salt content. Thus, high water temperatures result in low oxygen solubilities as does the presence of high salt concentration which, for example, characterizes sea water. The values generally used for the solubility of oxygen in fresh and salt water are those given by the American Public Health Association [Standard Methods 1971] and were calculated by Whipple and Whipple [1911] from gasometric determination carried out by Fox [1909]. More recent investigations at the Water Pollution Research Laboratory in England [Truesdale et al., 1955] have resulted in publication of what may be considered to be more correct values as determined by a modification of the standard Winkler method [Standard Methods, 1971]. These investigations have resulted in the following empirical equation representing the solubility of oxygen:

$$C_s = 14.161 - 0.3943T + 0.007714T^2 - 0.0000646T^3 \qquad (2.1)$$

where

C_s is the saturation concentration of oxygen in ppm;

and

T is the temperature in °C.

The effects of barometric pressure on solubility when conditions are different from standard atmospheric are also accounted for empirically by the following:

$$C_s' = C_s \frac{P}{760}$$

where

C_s' is the solubility of oxygen at P mm Hg pressure;

and

C_s is the solubility of oxygen at 760 mm Hg pressure

Saturation concentrations for oxygen dissolved in mixtures of freshwater and seawater cannot be reduced to an empirical equation but values have been tabulated [Standard Methods, 1971 and Truesdale et al., 1955].

2.2 OXYGEN DEMANDING WASTES

When a wastewater is discharged into a waterway, the biodegradable organics contained in that wastewater exert an oxygen demand on the dissolved oxygen resources of the stream or estuary. This fact was first recognized in Britain during the 19th century by way of the investigations of the Royal Commission on Sewage Disposal, which was appointed in 1898 "to report on methods for the treatment and disposal of sewage and trade wastes". The Commission published a series of ten reports over 17 years which described many aspects of sewage disposal, ranging from standards and tests of sewage and sewage effluents through contamination of shellfish and growth of weeds in tidal waters to a comprehensive treatise on methods available for purification and disposal of sewage and trade wastes. This "milestone" study laid the groundwork for implementation of remedial measures in Britain where by the end of the 19th century, the water pollution in some areas was so bad that "all fish life and other aquatic life, animal and vegetable had virtually disappeared" and "the scum in parts of the River Irwell was so thick and solid that birds walked on it without thinking". [Klein, 1962].

The Royal Commission studies formed the basis of what was undoubtedly the first major water pollution investigation and, as such, the findings have had long lasting and far reaching impact. In this respect, the 8th Report is particularly important. It dealt with the question of standards and tests applied to sewage and sewage effluents being discharged to receiving waterways. A test of purity for sewage effluents and river water, first recommended by the Royal Commission in this report, was the "dissolved oxgyen taken up in 5 days at 65°F" which became a standard wastewater and river water quality parameter. Later to be modified slightly and re-named, the Biochemical Oxygen Demand (BOD) test was used by the Royal Society in their report to classify rivers (see Table 2.2).

TABLE 2.2

ROYAL COMMISSION CLASSIFICATION OF RIVERS [KLEIN, 1959]

APPROXIMATE 5-DAY BOD @ 65°F (ppm)	CLASSIFICATION
1	Very clean
2	Clean
3	Fairly clean
5	Doubtful
10	Bad

The BOD test has retained its importance as a measure of wastewater and river water quality in spite of the fact that the test is subject to a number of sometimes serious errors. Its popularity, acceptance and widespread use as a water quality parameter is in part due to the endorsement it received because of its development by the prestigious Royal Commission but mainly because of its value as a direct measure of oxygen demand as it is a test aimed at reproducing the oxidation conditions of a natural waterway. Contemporary BOD testing is conducted under more carefully controlled conditions of nutrient enrichment, dilution water make up and bacterial seeding at an increased, fixed incubation temperature of 20°C, all of which are aimed at standardizing the BOD test in order to enhance reproducibility. These improvements in laboratory procedure have resulted in increased reproducibility with minimization of error. However, vagaries intrinsic to the BOD test remain.

In character, BOD is defined as "the amount of oxygen required by bacteria while stabilizing decomposable organic matter under aerobic conditions" [Sawyer & McCarty, 1967]. The oxygen required during the organic decomposition results from the activity of a group of microorganisms, namely the aerobic bacteria, which utilize the organics as a food source, deriving energy from the oxidation process. The biochemical reaction may be generally represented by a quantitative relationship which defines, on a theoretical basis, the amount of oxygen required to convert an amount of any given organic compound to its ultimate end products - carbon dioxide, water and ammonia [Sawyer & McCarty, 1967].

$$C_nH_aO_bN_c + (n + \frac{a}{4} - \frac{b}{2} - \frac{3}{4}c)O_2 \longrightarrow nCO_2 + (\frac{a}{2} - \frac{3}{2}c)\ H_2O + cNH_3 \quad (2.3)$$

The rate of oxidation of organic matter is governed to a major extent by two variables - bacterial population and temperature.

In the BOD test, control of the incubation temperature has been standardized. But, however much emphasis is placed on "seeding" the test sample with bacterial seed, it is unlikely that bacterial populations will be controlled. The root of this problem and the main drawback of the BOD test is that the bacteria, in many cases, take time to become acclimatized to a particular wastewater. The time required for the acclimatization of bacterial populations is not considered separately in the BOD test, but occurs over the first few hours (or days) of the 5-day test. Thus, the amount of oxygen consumed in the 5-day test period may not accurately reflect the long term demand exerted by a particular waste because of the time taken for the oxidation rate to reach its maximum. This is true, in particular, with wastes that contain exotic materials other than readily oxidizable organics as, say, are present in an industrial wastewater. Regardless of the inherent inaccuracies of the BOD test, it is still widely used as a measure of the strength of organic waste and has been related to the oxygen balance in a stream.

2.3 THE OXYGEN BALANCE

The discharge of organic wastes into a waterway deoxygenates the water. That is, the bacteria degrading the organic matter consume oxygen, thereby causing a depletion of dissolved oxygen in the receiving water. Simultaneous to the depletion of dissolved oxygen is the occurrence of another process of nature - atmospheric reaeration. During this process, adsorption of atmospheric oxygen (and other gases) by the water takes place in order to maintain the equilibrium between the dissolved gases and the atmospheric gases according to Dalton's law of partial pressures. The net result of these two counteracting forcing functions is that an oxygen balance is established. This forms the basis of natural self-purification in waterways. Figure 2.1 shows an idealized form of the balance for a single major waste discharge showing the zones of degradation and recovery of a polluted stream. Three cases are given: (a) slight pollution; (b) heavy pollution; (c) gross pollution. In each case shown in Figure 2.1, the curves are the net result of deoxygenation and reaeration occuring simultaneously with, in case (a), the oxygen demand being small enough so that the minimum dissolved oxygen deficit is small and tolerable; in case (b), the oxygen demand being sufficient to deplete the oxygen resources to a minimum; and in case (c), the oxygen demand being so large that it depletes the oxygen to such an extent that septic conditions prevail over a stretch of the waterway.

The oxygen resources of a waterway, in a constant state of dynamic equilibrium, are controlled by the kinetics of the processes of deoxygenation and reaeration.

2.4 DEOXYGENATION

When measuring the oxygen demand of a wastewater over a long period of time, whether with a respirometer or in a series of BOD bottles, the oxygen consumed is observed to vary in a manner similar to that shown in Figure 2.2. The total oxygen demand is the net result of two separate and independent oxidation processes - carbonaceous oxidation and nitrification. During the first 10 to 15 days, the carbonaceous oxygen demand, sometime called the first stage oxygen demand (FSOD), accounts for most of the total demand and is the result of the oxidation of carbonaceous organic matter. During the subsequent period, nitrification

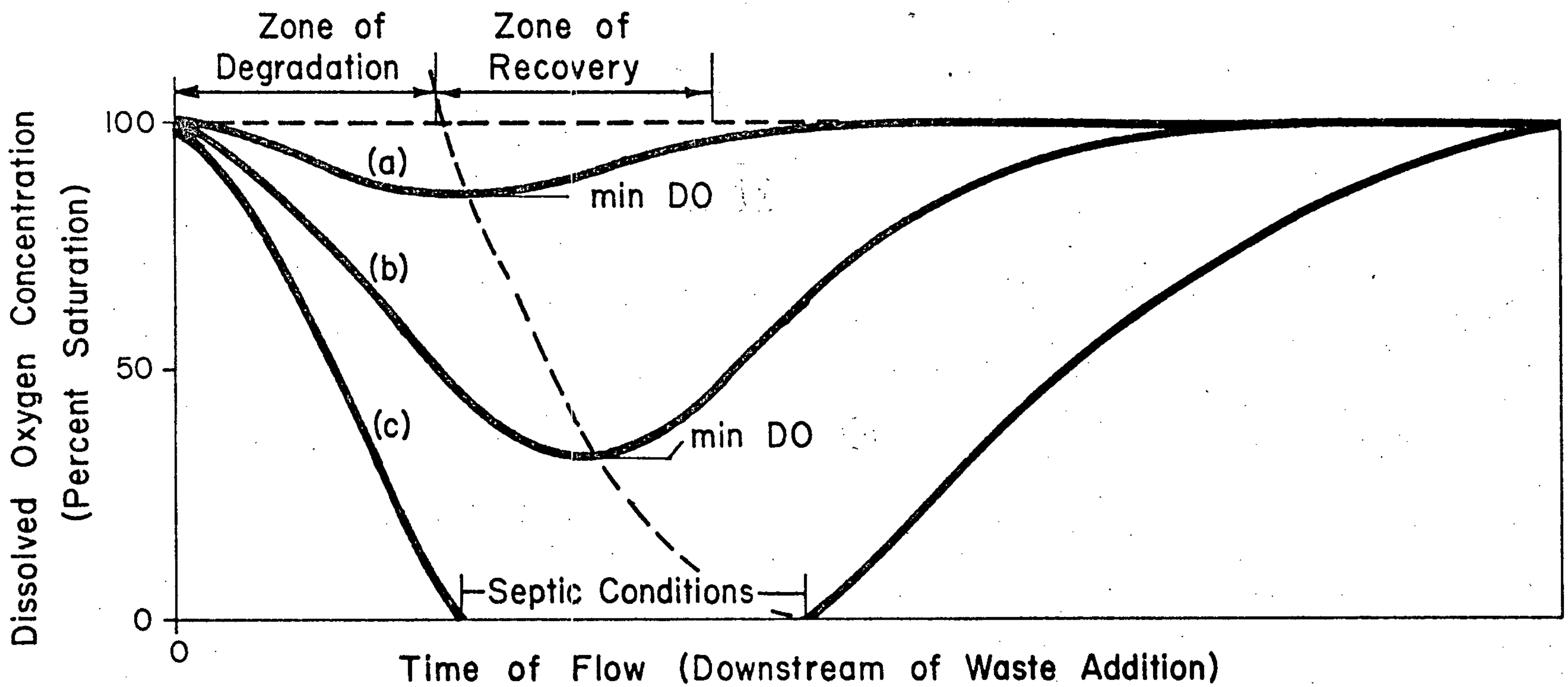


Figure 2.1

Oxygen Sag Curve

Degradation and Recovery of Stream Shown For:
(a) Slight; (b) Moderate; (c) Heavy Pollution

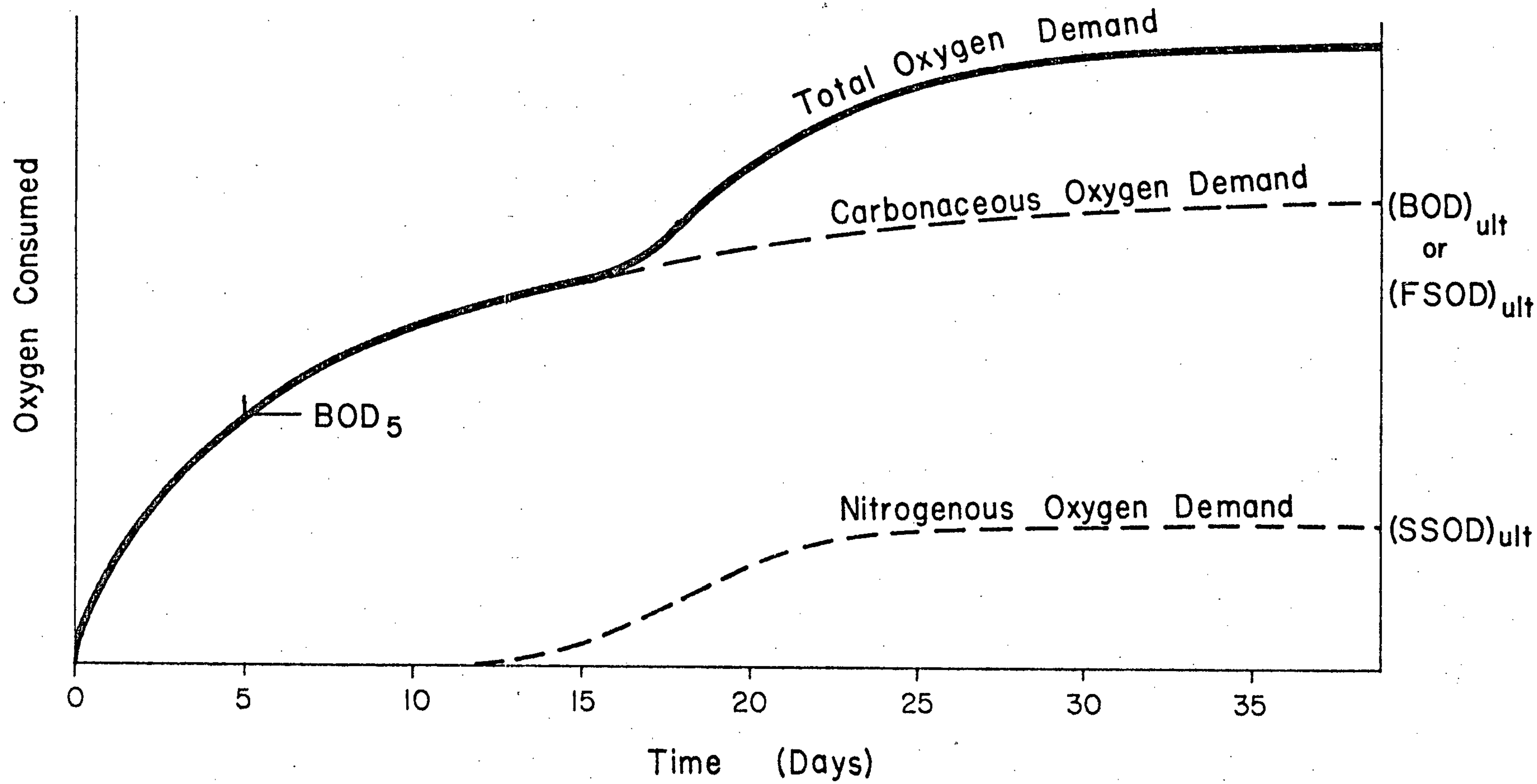


Figure 2.2

Oxygen Uptake of a Wastewater

occurs although, in some instances, if the wastewater contains significant quantities of ammonia, it can take place simultaneous to carbonaceous oxidation. In this process, the oxidation of ammonia ultimately to nitrate, by nitrifying bacteria, exerts an additional oxygen demand, often referred to as the second stage oxygen demand (SSOD), which results in an increased total oxygen uptake.

2.4.1 Carbonaceous Oxidation. The classical approach to expressing mathematically the variation of BOD with time taken by pioneers in the field [Theriault, 1927 and Phelps, 1944] made the assumption that the oxidation was a mono-molecular or "first order" reaction. That is, the rate of uptake of oxygen was assumed to be proportional to the amount of oxidizable organic matter remaining at any time, expressed mathematically as:

$$\frac{dL}{dt} = -K_1 L \tag{2.4}$$

where

L represents the ultimate FSOD at any time t;

and

K_1 is the rate constant for the reaction, sometimes referred to as the de-oxygenation coefficient.

The integrated form of this equation is:

$$L = L_o e^{-K_1 t} \tag{2.5}$$

where

L_o is the initial value of L at t = 0.

A more convenient form of equation (2.5) is given by:

$$y = L_o\ (1 - e^{-K_1 t}) = L_o(1 - 10^{k_1 t}) \tag{2.6}$$

where

y is the BOD at any time t;

and

L_o is the total or ultimate FSOD.

(It should be noted that, as is conventional, K_1 is the rate constant to the base e and k_1 is the rate constant to the base 10.)

The rate constant, k_1, from the classical formulation of BOD as a continuous first-order reaction is, in fact, not a constant but a variable, the magnitude of which is governed by a number of important factors, including temperature, the nature of the organic substrate and the ability of the organisms present to utilize the substrate. Foremost, as was discovered by the pioneers [Streeter and Phelps, 1925], increasing temperatures increased the de-oxygenation constant, the value roughly doubling for a temperature increase of 15°C. The temperature effect is generally represented by the following relationship, derived from van't Hoff's law:

$$(k_1)_T = (k_1)_{20}\ \theta^{(T-20)} \tag{2.7}$$

where

$(k_1)_T$ is the value of k_1 at any temperature T°C;

$(k_1)_{20}$ is the value at 20°C;

and

θ is a temperature coefficient.

Streeter and Phelps found θ to be 1.047 based on their early studies, but more recent research [Schroepfer et al., 1960] has shown the original temperature coefficient to be too low, resulting in inaccuracies at low temperatures. Schroepfer [1960] suggests θ values of 1.056 for the temperature range 20-30°C and 1.135 for the temperature range 4-20°C.

Another factor affecting the rate constant is the ability of the organisms to utilize the organic matter. It has been theorized [Eckenfelder, 1970] that BOD reactions and, in fact, all aerobic reactions, occur in two separate and distinct phases. During the first phase, which is usually complete in 18-36 hours, the organic matter present in the wastewater is utilized by the microrganisms for energy and growth (the "synthesis" phase). When the organics originally present in the wastewater are removed, the organisms present continue to use oxygen for auto-oxidation and endogenous metabolism of their cellular mass (the "endogenous respiration" phase). Only when the cell mass is completely oxidized to non-biodegradable cellular residue, usually taking more than 20 days, is the oxidation complete. The main distinguishing difference between the two phases is the different rates of reaction. During the first, or assimilation phase, the reaction rate is some 10 to 20 times faster than the rate of endogenous oxidation. The two-phase theory of BOD reaction sheds a new light on the meaning of the BOD rate constant, k_1. However, according to Eckenfedler [1970], the "overall average reaction rate" over the entire oxidation is comparable to the classical form of rate constant that was developed by Streeter and Phelps.

Another factor affecting the deoxygenation constant is the nature of the organic substrate undergoing oxidation. With raw sewage, for example, average reaction rate constants will tend to be high reflecting the ease with which the waste can be degraded. After treatment which removes much of the organic matter, and in rivers, where only low concentrations of organics are present, the rate constants are significantly lower. This can be explained in terms of the "two-phase" theory that, in these cases, the oxidation is mainly due to endogenous respiration which results in the lower rate constants. The variation in average BOD rate constants for a range of substances is shown in Table 2.3.

TABLE 2.3

AVERAGE BOD RATE CONSTANTS @ 20°C [ECKENFELDER, 1970]

SUBSTANCE	$(k_1)_{20}$
Untreated wastewater	0.15-0.28
High rate filters and anaerobic contact effluent	0.12-0.22
High degree biotreatment effluent	0.06-0.10
Rivers with low pollution	0.04-0.08

A comparison of a plot of equation (2.6) with the observed oxygen uptake of organic matter will, surprisingly enough, most often show a relatively good "fit" for the first 8 to 10 days of oxidation after which time the BOD curve diverges radically from the course it would be expected to follow as a unimolecular reaction [Sawyer & McCarty, 1967]. The main reason for the divergence from the theory after the initial period of oxidation is that the effects of nitrification become significant.

2.4.2 Nitrification. The nitrogenous stage of biochemical degradation, as in the BOD test, includes conversion of organic nitrogen to ammonia and the subsequent oxidation of ammonia ultimately to nitrate. Organic nitrogen is hydrolyzed to ammonia, under aerobic or anaerobic conditions, without the utilization of oxygen. The ammonia is successively oxidized by nitrifying bacteria through nitrite to nitrate by the organisms nitrosomcnas and nitrobacter, respectively. [Wezernak, 1968].

In the BOD test of raw wastewater there is generally a pronounced lag of 10-15 days between the beginning of carbonaceous oxidation and the nitrification step. This lag is less for treated samples, likely because of acclimatization of the nitrifying bacteria during the sewage treatment process, and is in the order of one or two days for highly treated samples. In streams, the two stages frequently proceed simultaneously, although there may be lags in the nitrification stage in highly polluted streams or in those with low dissolved oxygen. [Manhattan College, 1972].

There have been examples reported [Courchaine, 1963] where high nitrification rates have had significant effects on the oxygen balance of streams. This was attributed to the combination of a high degree of waste treatment and a high temperature receiving water both of which favour nitrification. It was found that the high receiving water temperatures (25-30°C) resulted in the natural proliferation of an over-abundant population of nitrifying bacteria. This, in turn, resulted in an almost immediate uptake of oxygen by these bacteria when a wastewater containing significant nitrogen was introduced. In general, however, except in high temperature receiving streams, nitrification is not of much significance when compared to the effects of carbonaceous demand on stream dissolved oxygen resources. At temperatures below 12 ± 2°C, nitrification is usually not significant [Manhattan College, 1972].

2.5 REAERATION

The deoxygenation of a waterway, due to the degradation of organics, is counterbalanced by the natural process of atmospheric re-aeration. Atmospheric oxygen is absorbed readily by the oxygen deficient water to maintain the balance of partial pressures between the atmosphere and the water which, in the limit, results in the saturation of water with dissolved oxygen. The driving force for the rate of oxygenation is the dissolved oxygen deficit, which is simply the difference between the saturation concentration and the existing concentration.

Mathematically, this may be expressed as:

$$\frac{dD}{dt} = -K_2 D \tag{2.8}$$

where

D is the dissolved oxygen deficit at any time t;

and

K_2 is the rate constant, often refered to as the reaeration coefficient, in form, similar to the deoxygenation constant.

Integration of Equation (2.8) results in:

$$D = D_\circ e^{-K_2 t} = D_\circ \, 10^{-k_2 t} \tag{2.9}$$

where

$D_\circ$ is the initial deficit at t = 0.

Many factors are known to affect the value of the reaeration coefficient, k_2. Natural mixing in waterways has been found to be particularly important as it effects the rate of surface renewal at the gas-liquid interface where the oxygen transfer takes place. Early research efforts [Streeter and Phelps, 1925] concluded that the influence of the hydraulic and physical characteristics of a stream on the reaeration coefficient could be described empirically as follows:

$$k_2 = aU^nH^{-m} \tag{2.10}$$

where

U is the velocity;

H is the water depth;

and

a, m and n are empirical constants dependent on the hydraulic conditions and on the slope and roughness of the stream bed.

Many research investigations since have been devoted to the determination of stream reaeration rates [Dobbins and O'Connor , 1956; Churchill, et al., 1962; Dobbins, 1964; Isaacs and Gaudy, 1966; and Langbein and Durum, 1967; to name but a few]. A number of these investigations concentrated on accurately measuring the reaeration rate and then determining the empirical constants to the original Streeter-Phelps formulation (Equation (2.10)). Other research efforts attempted a more theoretical approach to oxygen transfer investigating thin film theories and the effects of surface renewal. From this latter work, there appears to be a semi-theoretical basis to the empirical formulation as a number of the theoretical approaches have resulted in the development of relationship similar to Equation (2.10). Some of the values for the empirical constants given in the literature are shown in Table 2.4.

TABLE 2.4

SUMMARY OF CONSTANTS FOR THE REAERATION EQUATION

(For U in feet per second and H in feet)

a	m	n	REFERENCE
9.4	0.67	1.85	Owens *et al.*, 1964
5.5	0.50	1.50	Dobbins & O'Connor, 1956
5.0	0.97	1.67	Churchill *et al.*, 1962
3.7	1.00	1.50	Isaacs & Gaudy, 1966
3.3	1.00	1.33	Langbein & Durum, 1967

Of the range of empirical formulae, the one generally considered to be most reasonable for the estimation of reaeration rates over a wide range of depth and velocity conditions [Thomann, 1971] is that of Dobbins & O'Connor [1956] given by:

$$k_2 = 5.5U^{0.5} H^{-1.50} \tag{2.11}$$

In addition to the physical and hydraulic factors, reaeration rates are also affected by winds, waves and chemical agents, such as surfactants. The effects of surfactants are minimal in natural waterways because, even if they are present, concentrations are usually so low that the oxygen transfer is not inhibited. Wind and wave effects, however, can at times be significant as they result in increased surface renewal due to wind-induced shear stresses and breaking waves. They are usually accounted for empirically by an adjustment of reaeration rates according to wind velocities. A relationship which has been suggested in a recent study of dissolved oxygen dynamics in the Sacremento-San Joaquin Delta [DFWPS, 1972], considered to be additive to the reaeration coefficient previously discussed, is given by:

$$\Delta k_2 = \frac{(Cw)}{H} W \qquad (2.12)$$

where

W is the wind velocity in mph;

and

C_w is the wind correction factor ranging in value from 0 to 0.4.

Temperature has also been found to affect reaeration rates. Temperature compensation is made by the use of a temperature correction factor, θ, in a manner similar to that discussed earlier for BOD decay rates. The temperature correction factor has been found to vary from 1.017 to 1.044 [Dobbins, 1964] and most commonly is chosen to be 1.024.

2.6 OTHER SOURCES AND SINKS OF OXYGEN

As well as the sources and sinks of oxygen previously mentioned, namely reaeration and biochemical oxidation, there are other source/sink processes which are, at times, important factors in the oxygen balance. Additional demands are often exerted on the oxygen resources of a waterway by the decomposition of bottom deposits of organic matter, the respiration of aquatic plants and the immediate chemical oxygen demand of a wastewater.

On the other side of the ledger, oxygen may be added to a stream by the process of photosynthesis. As well, the removal of BOD due to settling, which occurs in a slow moving stream, is a positive factor; however, the amount of oxygen demand removed by this process will usually add to the benthic demand.

2.7 THE STREETER-PHELPS FORMULATION OF THE OXYGEN BALANCE IN A STREAM

The Streeter-Phelps formulation of the oxygen balance in a stream, cited several times above, can now be spelled out fully. Their investigation into pollution in the Ohio River in the 1920's applied the concepts of deoxygenation and reaeration to define the natural self-purification of waterways. The formulation of the oxygen balance in a stream, developed as a result of their studies, assumed that the bio-chemical oxygen demand of a wastewater and reaeration from the atmosphere were the only two processes which determined the net rate of change of oxygen deficit. The "classical" Streeter-Phelps formulation, sometimes referred to as the "oxygen-sag curve", is given by:

$$\frac{dD}{dt} = K_1 L - K_2 D \qquad (2.13)$$

where

all parameters are as previously defined.

Integration of Equation (2.13) results in:

$$D = \frac{K_1 L_o}{K_2 - K_1} \left[e^{-K_1 t} - e^{-K_2 t} \right] + D_o e^{-K_2 t} \tag{2.14}$$

where

D is the dissolved oxygen deficit at any time t;

with

L_o and D_o being respectively the initial ultimate FSOD and the initial DO deficit.

Thus, knowing the initial stream BOD and DO deficit and the appropriate deoxygenation and reaeration coefficients, the DO deficit may be calculated for any time t. The resultant plot of DO deficit versus time is a quantitative solution to the oxygen-sag curves shown in Figure 2.1 and discussed in Section 2.3.

The simplified Streeter-Phelps equation can be expanded to include the effects on the oxygen balance of some of the other oxygen source/sink parameters mentioned in Section 2.3.5, specifically, the settling out of BOD, the addition of BOD to overlying water by bottom deposits, and photosynthetic respiration. The solution to this more general form of the oxygen sag curve which accounts for the effects of these additional parameters [Camp, 1965] is given by:

$$D = \frac{K_1}{K_2 - K_1 - K_3} \left[L_o - \frac{p}{(K_1 + K_3)} \right] \left[e^{-(K_1 + K_3) t} - e^{-K_2 t} \right] + \frac{K_1}{K_2} \left[\frac{p}{(K_1 + K_3)} - \frac{a}{K_1} \right] \left[1 - e^{-K_2 t} \right] + D_o e^{-K_2 t} \tag{2.15}$$

where

K_3 is the rate of BOD loss due to settling per day;

p is the rate of BOD addition from bottom in ppm per day;

a is the rate of photosynthetic oxygen addition in ppm per day;

and

all other parameters are as previously defined.

CHAPTER 3

DISSOLVED OXYGEN MODELS

The mathematical formulation of the oxygen balance, discussed in the previous chapter, forms the basis of the dissolved oxygen "model". A "model" is, in the strictest sense, a formalism - the mathematical expression of a relevant physical process, or processes, in this case the natural processes of the deoxygenation and reaeration in water. The term "model", often used loosely, has numerous meanings. In the most general sense, it can mean a very non-specific conceptual idealization of a problem or process, as in the economists' use of "economic models". It can also be used to mean a simple analog of a real system, examples of which are the physical models used in engineering and hydraulics. Another use of the term "model", which combines the strict definition of formalism and the general definition of conceptualized idealism, refers to a mathematical formulation together with a technique which allows for a solution of the variables of concern. This usage is implicit in the term "digital model" which in some cases refers to a specific computer program used in the solution. It is also the most appropriate definition which can be used to describe the dissolved oxygen models which will be discussed in this chapter.

3.1 STREAM AND RIVER MODELS

The basic Streeter-Phelps formulation of the oxygen balance in streams and rivers describes a time dependent functional relationship between the basic oxygen source and sink processes for a single waste discharge. The application of this "classical" theory to streams and rivers in order to solve for oxygen-sag is quite straightforward when the following assumptions are made: (i) cross-sectional mixing is rapid; (ii) river flows are steady, and (iii) "plug flow" (lack of longitudinal mixing) exists. These assumptions are reasonable for most rivers where, in general, transverse and vertical mixing rates are high, resulting in relatively rapid cross-sectional mixing, and longitudinal mixing rates are low, resulting in minimal longitudinal dispersion. As for the steady nature of river and stream flow, although the flow may not be steady over the entire stream length, it can be considered to be steady at least over given stretches of the river.

The assumption of steady river flow allows simple transposal of reference frames from the time-scales used in the dissolved oxygen (DO) deficit equation (Equations 2.14 & 2.15) to distance-scales. By continuity, average velocities can be obtained and they may be related to distance as Velocity (V) = Distance (x) x Times (t). Thus by simple substitution of V/x for t in the oxygen-sag solution, the DO deficit is expressed as a function of x giving a solution for the spatial distribution of DO in a stream or river downstream of a single waste outfall. In the event that there is more than one outfall, a DO distribution can be calculated for each waste input and the solutions superimposed, the total DO deficit being the sum of the individual deficits. Also, if conditions of temperature and river flow change over stretches of the river, affecting changes in both of the reaction rate constants and the velocity, the stretch of river may be divided up into smaller reaches over which conditions are constant. This manner of segmentation can be made to coincide, where possible, with waste input locations so that both changes in river conditions and additional waste inputs can be handled in one step.

3.2 ESTUARY MODELS

The application of the "classical" dissolved oxygen model to estuaries is not as simple and straightforward as it is in the case of rivers and streams. This is primarily because of the complex, non-uniform nature of flow patterns in estuaries which exist as a result of tidal influence. Although the natural physical and biochemical processes involved are similar, the unsteady, oscillatory flows in estuaries present problems when one attempts to apply a dissolved oxygen model because of the difficulties encountered in obtaining the time-histories of different parcels of water in an estuary. Unlike rivers, where for a given set of river flows there will be a discrete set of river velocities for every location along the stream length, velocities in an estuary vary with time as well as location, the temporal variations being due to changes in water surface elevations throughout the estuary during the tidal cycle (see Figure 1.10 for typical variations in the lower Fraser River). The unsteady nature of velocities in estuaries, which are seen to vary "discontinuously" (in the sense of direction) makes it impossible to easily transpose between time and distance reference frames. Thus, the basic Streeter-Phelps formulations for DO distributions, which depend on a simple time-distance transposal, become difficult to implement.

Another facet of estuarine hydraulics which differs from river hydraulics and has important implications for the application of dissolved oxygen models is increased dispersion and mixing in estuaries caused by the oscillatory movement of the water mass. Significant vertical and lateral velocity gradients, together with turbulent diffusion, tend to erode concentration peaks from a slug discharge, spreading and redistributing the dissolved substance around the line of mean advective advance. This mechanism of longitudinal mixing, which is termed longitudinal dispersion, is many times more significant in estuaries than in rivers where its effect can be ignored in favour of a "plug flow" assumption. In estuaries, however, it must be accounted for; in the case of estuaries with zero fresh water inflow, dispersion due to tidal mixing is the only mechanism of material transport.

As we have seen, the non-simple nature of estuarine hydraulics with dispersion effects as well as spatial and temporal variation in velocity, prohibits the direct application of the basic Streeter-Phelps dissolved oxygen model. In order to develop a dissolved oxygen model which can be applied to estuaries, it is necessary to go back to basic mass transport principles. This more sophisticated approach will ultimately utilize all the basic concepts introduced in the previous chapter and result in a more generally applicable dissolved oxygen model.

3.3 THE ONE-DIMENSIONAL MASS TRANSPORT EQUATION

The basic one-dimensional equation for describing the mass transport of any dissolved substance in an unsteady non-uniform flow is obtained by making a mass balance over an elemental cross-sectional slice of the flow field of the waterway. Mass is transported through the slice by the processes of advection and dispersion, and these processes, together with any source or sinks of the substance within the slice, determine the concentration. The generalized form of the one-dimensional mass balance equation is given by:

$$\frac{\partial s}{\partial t} = -u\frac{\partial s}{\partial x} + \frac{E}{A}\frac{\partial}{\partial x}\left[A\,\frac{\partial s}{\partial x}\right] \pm \sum_{i=1}^{n} S_i \tag{3.1}$$

where

s is the concentration of any dissolved substance;

u is the longitudinal velocity;

E is the longitudinal dispersion coefficient;

A is the cross-sectional area;

S_i is the rate of production per unit volume due to the i^{th} source-sink process, $i = 1,\ldots,n$;

x is the longitudinal distance;

and

t is the time.

The three terms on the right hand side of the equation are the advective, dispersive and source-sink terms, respectively.

Since Equation (3.1) is one-dimensional, all the variables and parameters are cross-sectionally averaged values. In an estuary, because of tidal effects, the parameters of the equation u, E and A, defined by the cross-sectional geometry and hydraulics for the particular estuary, can vary over both space(x) and time(t). Thus, in the solution, it is necessary to account in some manner for the temporal as well as spatial variation of these parameters.

There are two types of solutions which can be applied to the one-dimensional mass transport equation - the "steady-state" or "tidally averaged" solution and the "tidally varying" solution. The steady-state solution averages out the effects of the tide over the tidal cycle assigning parameters their "mean tidal" value. Because of this averaging of tidal conditions, this solution is, in some ways, a temporal abstraction of the processes involved but, as we shall see, is nonetheless very useful. The second type, the tidally varying solution, accounts for the temporal variation in the parameters by obtaining their values independently through solution of the hydrodynamic equations which describe the hydraulics of the estuary. The tidally varying solution to the mass transport equation, although it allows for greater temporal resolution and the observation of "real time" effects, demands a much more sophisticated approach and greater expense of effort in development. The two types of solutions will now be discussed.

3.4 THE STEADY STATE SOLUTIONS TO THE MASS TRANSPORT EQUATION

This method of solving the one-dimensional mass transport equation, referred to as the "tidally averaged" approach, assigns parameters their mean values over a tidal cycle. This does not alter the form of the mass transport equation but merely changes the interpretations of the variables and parameters. For example, the velocity term(u) becomes the tidally averaged velocity(U), which is determined by the fresh water discharge through the mean tidal cross-sectional area. The longitudinal dispersion coefficient is replaced by what is called the tidal dispersion coefficient(E).

The assumption of steady-state also implies that any derivatives with respect to time are zero. This means that concentrations will be assigned their mean tidal value and will be seen to vary only with distance along the estuary. The incorporation of the steady-state assumption reduces Equation (3.1) still in the general case to:

$$0 = -U\frac{ds}{dx} + \frac{E}{A}\frac{d}{dx}\left[A\frac{ds}{dx}\right] \pm \sum_{i=1}^{n} S_i \tag{3.2}$$

where

U is the tidally averaged velocity (=Q/A);

E is the tidally averaged dispersion coefficient;

Q is the fresh water discharge;

A is the tidally averaged cross-sectional area;

and

all other parameters and variables are as previously defined.

Applying Equation (3.2) to dissolved oxygen in estuaries is relatively straight forward if it is assumed that the only source/sink processes are atmospheric reaeration and biochemical oxygen demand, both of which are assumed to be first order rate processes. The resultant steady-state distribution for dissolved oxygen is given by:

$$0 = -U\frac{dc}{dx} + \frac{E}{A}\frac{d}{dx}\left[A\frac{dc}{dx}\right] + K_2(c_s - c) - K_1L \tag{3.3}$$

where

c is the dissolved oxygen concentration;

c_s is the saturation concentration of dissolved oxygen;

L is the BOD concentration remaining at any point x;

K_2 is the reaeration rate coefficient;

K_1 is the BOD decay rate coefficient;

and

all other parameters and variables are as previously defined.

Since the BOD term appears in Equation (3.3), it is necessary to develop an equation for BOD distribution in an estuary before proceeding with the DO solution. Applying the general steady-state mass transport equation to BOD, assuming point source waste addition with first order BOD removal, results in:

$$0 = -U \frac{dL}{dx} + \frac{E}{A} \frac{d}{dx}\left[A \frac{dL}{dx}\right] - K_r L \tag{3.4}$$

where

L is the BOD concentration remaining at any point x;

K_r is the BOD removal rate coefficient;

and

all other parameters are as previously defined.

It should be noted that the BOD removal rate (K_r) is the total removal coefficient which can account for removal of BOD due to settling, as well as oxidation, it being assumed that settling is approximated by first order kinetics. If BOD removal is accomplished only by oxidation, then $K_r = K_1$.

We now have equations to describe, under steady-state conditions, the distributions of DO and BOD in an estuary. Since the DO response is determined in part by input from the BOD solution, Equations (3.3) and (3.4) are said to be a "coupled" pair of equations and the solution must proceed accordingly. Note that if E is set equal to zero in Equations (3.3) and (3.4), which would be the case for a river or stream, the resultant solution of the coupled BOD/DO equation would, under the similar assumptions, be comparable with that derived by way of the Streeter-Phelps formulation.

There are two basic solution techniques for the steady-state coupled BOD/DO system. The first technique is an analytical one which offers continuous solutions to the equations for various estuarial and boundary condition configurations. It can be credited largely to the work of D.J. O'Connor [O'Connor, 1965] who has provided solutions for a number of special cases. The advantage of the continuous solution approach is that it is designed to handle long river/estuary stretches in a simple, straightforward manner. The second approach, developed by Robert Thoman [Thoman, 1965, 1971], uses finite difference techniques to solve the coupled system of equations. Although this method is more flexible in that it is not restricted by estuary geometry, waste inputs or boundary condition configurations, it is often imposing as a solution method because of the large number of linear equations which must be solved by digital computer matrix inversion techniques. The two approaches to solution of the steady-state equations will now be discussed.

3.4.1 The Continuous Solution Approach. In order to obtain continuous solutions by analytically solving the BOD/DO equations, it is first necessary to make assumptions regarding estuary areal configuration. O'Connor [1965] offers a number of solutions which apply to estuaries with constant as well as variable cross-section providing that the variation can be expressed in terms of longitudinal distance by linear, power or exponential expressions. An examination of the cross-sectional geometry of the main stem Fraser (see Figure 4.4) revealed that the cross-sectional area increases in the seaward direction. From the results of a linear regression analysis, run to determine the relationship between area and distance, it was found that a linear expression relating cross-sectional area with longitudinal distance gave a good "fit" yielding values for the coefficient of determination (r^2) which varied from 0.62 to 0.69 depending on river stage. Thus O'Connor's analytical solutions for estuaries with cross-sectional area increasing linearly in the seaward direction were appropriate and could be applied to the lower Fraser system.

To insure that the details of the working solutions offered by O'Connor were correct, the solutions were re-derived, by way of proof, from the basic BOD and DO mass transport equations. This verification procedure, which will not be presented, proved that the analytical solutions were correct. The solution for BOD distributions upstream and downstream of a single waste discharge at $x = x_o$ are given by:

$$\text{for } x<x_o: \quad L_1(x)=\left[\frac{Wx_o}{A_oE}\right]\left[\frac{x}{x_o}\right]^v\left[K_v(qx_o)I_v(qx)\right] \tag{3.5a}$$

$$\text{for } x>x_o: \quad L_2(x)=\left[\frac{Wx_o}{A_oE}\right]\left[\frac{x}{x_o}\right]^v\left[I_v(qx_o)K_v(qx)\right] \tag{3.5b}$$

where

$$v = \frac{x_oQ}{2A_oE} \quad ;$$

$$q = \sqrt{\frac{K_r}{E}} \quad ;$$

$L_1(x)$ is the BOD distribution as a function of distance upstream of the outfall location;

$L_2(x)$ is the BOD distribution as a function of distance downstream of the outfall location;

W is the constant rate of waste addition;

A_o is the cross-sectional area at the outfall location;

Q is the freshwater inflow to the estuary;

E is the tidal dispersion coefficient;

K_r is the BOD removal rate coefficient;

I_v and K_v are modified Bessel functions of the first and second kind of order v;

and

x is longitudinal distance along the estuary.

The corresponding DO deficit distribution for the regions upstream and downstream of a single outfall at $x = x_o$ are given by:

$$\text{for } x<x_o:\ D_1(x)=\left[\frac{K_r}{K_2-K_r}\right]\left[\frac{Wx_o}{A_oE}\right]\left[\frac{x}{x_o}\right]^v\left[K_v(x_oq)I_v(xq)-K_v(x_op)I_v(xp)\right] \tag{3.6a}$$

$$\text{for } x>x_o:\ D_2(x)=\left[\frac{K_r}{K_2-K_r}\right]\left[\frac{Wx_o}{A_oE}\right]\left[\frac{x}{x_o}\right]^v\left[I_v(x_oq)K_v(xq)-I_v(x_op)I_v(xp)\right] \tag{3.6b}$$

where

$$p = \sqrt{\frac{K_2}{E}}\ ;$$

$D_1(x)$ is the DO deficit distribution as a function of distance upstream of the outfall location;

$D_2(x)$ is the DO deficit distribution as a function of distance downstream of the outfall location;

K_2 is the reaeration rate coefficient;

and

all other parameters and variables are as previously defined.

These solutions for BOD and DO distribution in an estuary may at first seem somewhat overbearing because they contain a relatively uncommon mathematical expression, the modified Bessel function. However, inspection of the behaviour of modified Bessel functions reveals that they are similar to exponentials, with the modified Bessel functions of the first kind, $I_v(x)$, behaving in a manner analagous to ae^{bx}. Functions of the second kind, $K_v(x)$, behave similarly to ae^{-bx} except in the region of the origin where, as x approaches zero, $K_v(x)$ asymptotically approaches infinity.

Equations (3.5) and (3.6) give the BOD and DO deficit distributions in an estuary due to a single waste source. If more than one source is present, the distributions due to each may be calculated and, by applying the principle of superposition as is done in rivers, summed to give the total distribution of BOD and DO deficit.

When applying the continuous solutions to an estuary it is necessary to assume that conditions are constant over the entire estuary. Thus it is not possible by this method to account for any variation over the length of the estuary in such things as water temperature or in the values of parameters describing the dissolved oxygen source/sink and dispersion processes.

3.4.2 <u>The Finite Section Approach</u>. The finite section approach in essence replaces the derivatives in the mass transport equation with finite difference approximations. In this method of solution, first applied by Thomann [Thomann 1965, 1971] to the Delaware Estuary, the estuary is divided into a number of segments (or boxes) with each segment assumed to be completely mixed. Assuming that the advective and dispersive transport processes and waste discharges to the estuary are steady in time, a materials balance can be written around each finite section of the estuary. Writing the mass balance is equivalent to replacing the derivatives in the mass transport equation by their finite difference approximations. The mass balance over segment i for a dissolved substance such as BOD undergoing first-order decay gives [Thomann, 1971]:

$$Q[\alpha_{i-1,i}c_{i-1} + (1-\alpha_{i-1,i})c_i] - Q[\alpha_{i,i+1}c_i + (1-\alpha_{i,i+1})c_{i+1}]$$
$$+ E'_{i-1,i}(c_{i-1}-c_i) + E'_{i,i+1}(c_{i+1}-c_i) - K_i c_i V_i + W_i = 0 \tag{3.7}$$

where

c_i is the concentration in segment i;

V_i is the volume of segment i;

W_i is the mass of waste substance discharged into segment i per tidal cycle;

K_i is the decay coefficient for segment i;

Q is the tidally averaged discharge through the estuary (the freshwater discharge);

$\alpha_{i,i+1}$ is the tidal exchange coefficient between segments i and (i+1);

and

$E'_{i,i+1}$ is the "effective dispersive" transport between segments i and (i+1).

The subscript notation of the various terms is illustrated in Figure 3.1.

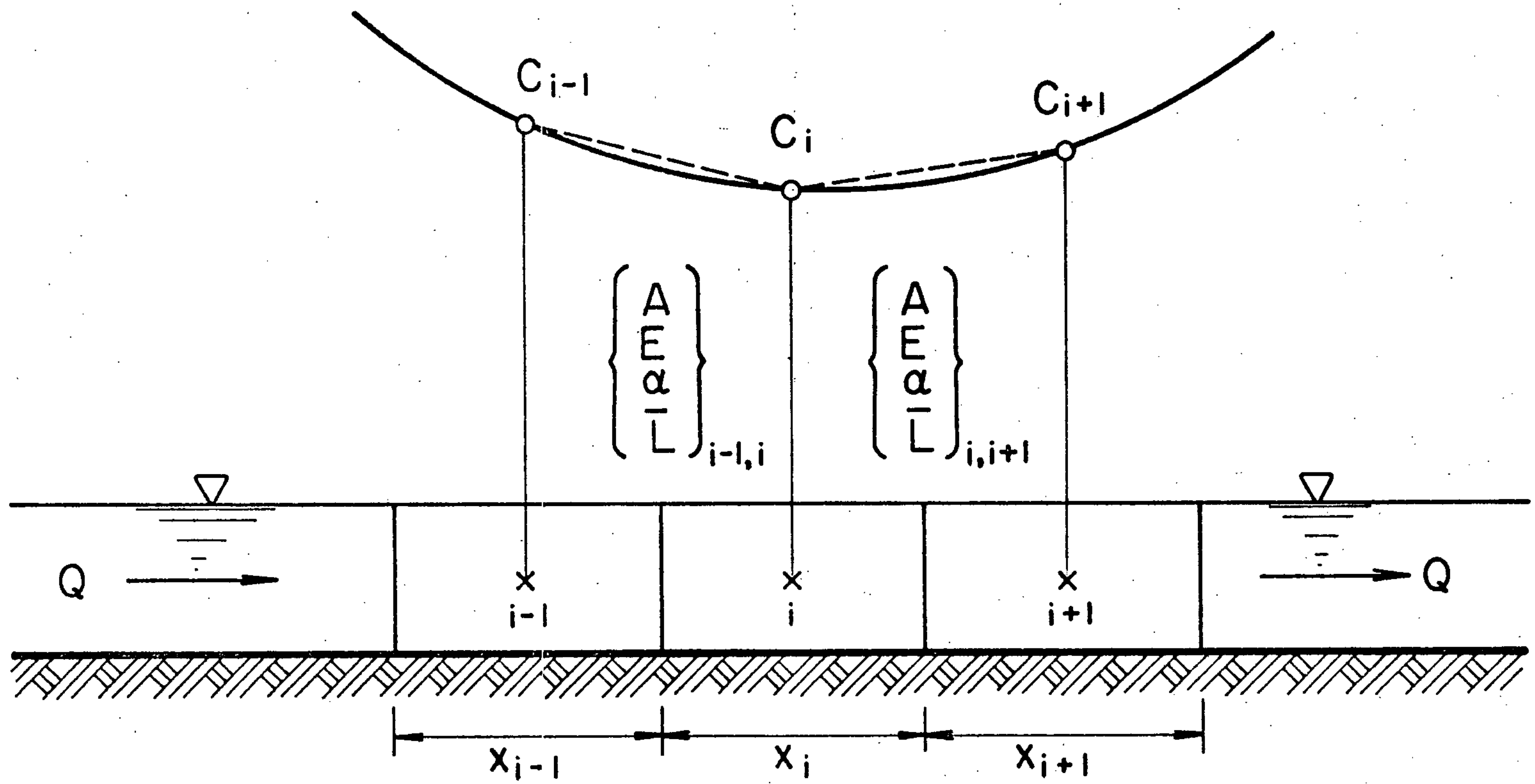


Figure 3.1

Subscript Notation of Tidally Averaged Model

The first two terms in Equation (3.7) are the tidally averaged advective transport into and out of segment i. The factor α is a weighting factor used to determine the concentration at the interface of two segments from the concentration within each segment. In tidal flows α is set equal to 0.5 to allow for the effects of flow reversal, whereas in a river flow situation α is set equal to 1.0 as the flow is always downstream. The next two terms of the equation represent the net dispersive transport of mass into segment i from the neighbouring segments. E' is given by:

$$E'_{i,i+1} = \frac{E_{i,i+1}\, A_{i,i+1}}{\bar{L}_{i,i+1}} \tag{3.8}$$

where

$E_{i,i+1}$ is the effective coefficient of dispersion over a tidal period at the interface of segments i and (i+1);

$A_{i,i+1}$ is the cross-sectional area (tidally averaged) of the interface between segments i and (i+1);

and

$\bar{L}_{i,i+1}$ is the average of the lengths of segments i and (i+1).

The final two terms of Equation (3.7) represent the effects of decay and waste discharge.

Equations similar to (3.7) can be written for each of n segments of an estuary to give a system of n simultaneous linear difference equations. These equations can be written in the general form as:

$$\sum_{i=1}^{n}\left[a_{i,i-1}c_{i-1} + a_{ii}c_i + a_{i,i+1}c_{i+1} = W_i\right] \tag{3.9}$$

where

$$a_{i,i-1} = -\alpha_{i-1,i}Q - E'_{i-1,i};$$

$$a_{ii} = Q\left[\alpha_{i,i+1} - (1-\alpha_{i-1,i})\right] + E'_{i-1,i} + E'_{i,i+1} + V_iK_i;$$

$$a_{i,i+1} = (1-\alpha_{i,i+1})Q - E_{i,i+1}.$$

Using matrix notation, the system of Equations (3.9) can be written

$$[A](c) = (W) \tag{3.10}$$

where [A] is a (nxn) tri-diagonal matrix and (c) and (W) are (nxl) vectors. The solution vector (c) is then obtained formally by inversion of the A matrix, i.e.

$$(c) = [A]^{-1}\ (W) \tag{3.11}$$

Thus, the problem of determining the steady-state, one-dimensional distribution of a waste material (such as BOD) in an estuary reduces to solving n simultaneous algebraic equations or inverting an (nxn) tri-diagonal matrix.

A finite difference approximation or mass balance around finite sections can also be applied to coupled systems. Suffice it to say here that a materials balance for DO can be written in a manner similar to that previously described for BOD. The resultant system of n simultaneous, linear equations for DO can be expressed in matrix forms coupled with the BOD system and solved by using matrix inversion techniques to obtain the steady-state, one-dimensional distribution of DO in an estuary [see Thomann, 1971 for details].

The application of the finite section approach to estuaries has the decided advantage of being more flexible than the continuous steady-state solution in that it is not restricted by assumptions regarding estuary geometry or other boundary condition configurations. Also, this method can easily be adapted to account for changes over the length of the estuary in such things as water temperature and the rate coefficients of the dissolved oxygen source/sink and dispersion processes. Since the finite section approach, by its main assumption, requires only that conditions be held constant within each segment, they may be allowed to vary from segment to segment throughout the estuary. Any number of waste discharges, each of which is assumed to be completely mixed into the segment adjacent to its location, can be handled simultaneously as the solution method is designed so that it interiorly accommodates the principle of superposition.

3.5 TIDALLY VARYING SOLUTIONS

Tidally varying solutions to the one-dimensional mass transport equation require, as input, information describing the spatial and temporal variations of the tidally varying parameters u, A and E. In order to obtain this information, it is first necessary to "model" the hydraulics of the particular estuary. This is done by applying the appropriate set of hydrodynamic equations (motion and continuity) to the water mass of the estuary and solving these equations throughout the tidal cycle. The resultant predictions of the space-time history of tidally varying parameters serve as input to the tidally varying solutions of the one-dimensional mass transport equation, and as such the hydrodynamic model is often referred to as a "sub-model" in the tidally varying scheme of solution. The hydrodynamic sub-model will now be discussed. It should be noted before proceeding that the inclusion, in this thesis, of the tidally varying approach to modeling was made possible because of an investigation into estuary modeling on the lower Fraser River carried out by C.S. Joy. His dissertation [Joy, 1974] and subsequent publication [Joy, 1975], to which the reader is referred, provide comprehensive coverage of details in the development of the hydrodynamic sub-model and the tidally varying solutions to the one-dimensional mass transport equations. The ensuing discussion which is but a brief summary of those details of the aforementioned research effort that are pertinent to this dissertation has borrowed heavily from Joy's publications.

3.5.1 The Hydrodynamic Sub-Model. The basic equations which describe the hydrodynamic behaviour of the water mass in an estuary, namely the equations of motion and continuity are, as applied to the Fraser River/Estuary, given by [Dronkers, 1969]:

$$\frac{\partial u}{\partial t} + u\frac{\partial u}{\partial x} = -g\,\frac{\partial h}{\partial x} - g\,\frac{|u|u}{C^2\bar{y}} \tag{3.12}$$

$$\frac{\partial}{\partial x}(Au) = -b\,\frac{\partial \bar{y}}{\partial t} \tag{3.13}$$

where

u is the mean longitudinal velocity;

h is the height of the water surface above an arbitrary level datum;

$\bar{y}$ is the mean cross-sectional water depth;

A is the cross-sectional area;

b is the cross-sectional width;

g is the local gravitational acceleration;

and

C is Chezy's friction factor.

Equations (3.12) and (3.13) are a coupled pair of partial differential equations with dependent variables u, A and either $\bar{y}$ or h (see Figure 3.2 for illustration of terms), independent variables x and t, and parameters C, b and g. Note that, as shown in Figure 3.2, in the hydrodynamic equations x increases in the upstream direction whereas in the mass transport equations x increases in the downstream direction. The main assumptions made in deriving the hydrodynamic equations were that the tidal storage width of the river/estuary is equal to the advective width, the Chezy formula adequately represents friction in the estuary and that the estuary hydraulics could be approximated by one-dimensional equations. These assumptions are, by and large, reasonable for the Fraser River/Estuary except perhaps in the lower reaches of the river where the presence of the salt water wedge causing a stratified flow field may affect both the frictional effects and the validity of the one-dimensional assumption.

Numerical solutions to the hydrodynamic equations can be obtained by using the fixed mesh, explicit finite difference method of Dronkers [1969]. By this method, which consists essentially of superimposing a grid of "fixed" stations along the estuary and replacing the derivatives in the hydrodynamic equation with finite difference approximations, solutions for the temporal and spatial variations of the tidally varying parameters u and A may be obtained for selected freshwater flows and tidal conditions. In this analysis hydraulic conditions are assumed

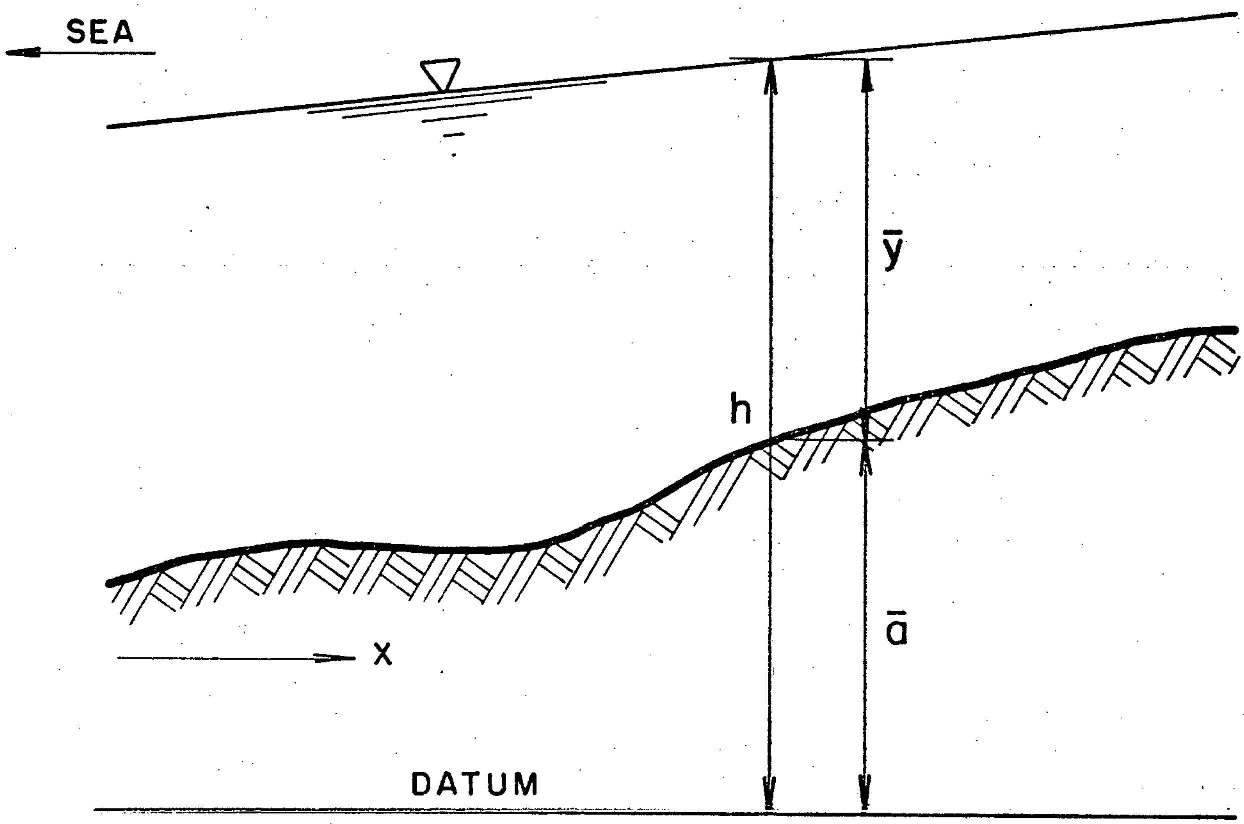


Figure 3.2

The Hydrodynamic Estuary

to be "quasi-steady" (i.e. repetitive) which simply requires that fresh-water flows be constant and tidal patterns be identical over the period of analysis. Since residence times in the lower Fraser are never more than four to six days, this assumption is reasonable.

The relative signs of grid spacing (Δx) and integration timed (Δt) in the solution scheme were selected according to requirements of stability in the numerical integration. It was found that a space grid at 5,000 foot intervals and an integration time of 90 seconds were satisfactory. Details of the network of stations chosen as a result of these criteria will be presented in Chapter 4 along with a discussion on the application to the Fraser River/Estuary of the hydrodynamic sub-model.

3.5.2 The Tidally Varying Model. Output from the hydrodynamic sub-model yields information on the space time history of the tidally varying parameters u and A. Before a tidally varying solution to the mass transport equation can be obtained, equivalent information must be derived for the coefficient of longitudinal dispersion (E). The time dependent behaviour of longitudinal dispersion, being a complex, poorly understood phenomenon influenced by such things as lateral and vertical velocity gradients, is difficult to describe. Since the models discussed in this thesis are one-dimensional, that is, they are not cognizant of any vertical and lateral variations, the tidally varying coefficient of longitudinal dispersion is replaced by the tidally averaged longitudinal dispersion coefficient. The tidally averaged coefficient is appropriate for use in this application especially because it is the form best suited to describing dispersion after cross-sectional mixing is complete. It must be stressed that this coefficient is <u>not</u> the same as the coefficient of tidal dispersion which is used in the steady-state solutions to the one-dimensional mass transport equations. The differences between these two coefficients will be discussed in Chapter 4.

Now that the spatial and temporal variations of all the tidally varying parameters can be accounted for, the tidally varying mass transport equation can be solved. The applicable solution method employs characteristic finite difference techniques [see Joy, 1974 for details]. The general one-dimensional equation (Equation 3.1) is transformed into its characteristic or Lagrangian form to give:

$$\frac{dx}{dt} = u \tag{3.14}$$

and

$$\frac{dc}{dt} = \frac{1}{A} \cdot \frac{\partial}{\partial x} \left[EA \cdot \frac{\partial c}{\partial x} \right] + \sum_{i=1}^{n} S_i \tag{3.15}$$

Equation (3.15) is then separated into its component dispersive and source/sink parts to give:

$$\frac{dc}{dt} = \frac{1}{A} \cdot \frac{\partial}{\partial x} \left[EA \cdot \frac{\partial c}{\partial x} \right] \tag{3.16}$$

and

$$\frac{dc}{dt} = \sum_{i=1}^{n} S_i \tag{3.17}$$

Finite difference approximations are used to replace the derivatives in Equations (3.14), (3.16) and (3.17) and numerical methods are used to obtain solutions to the tidally varying mass transport equation throughout the tidal cycle. By this solution technique, the mass transport equation is solved along the characteristic curves of the advective transport processes. The main advantage of this method is that it proceeds to a solution directly and accurately, eliminating such things as numerical dispersion. As well, by this technique each of the processes relevant to mass transport - advection, dispersion and source/sink - are handled independently.

Briefly, the mechanics of the solution are as follows. The advective transport process is simulated first by moving a grid of "points", each containing a concentration of dissolved substance for a time increment of one hour according to the velocities predicted by the hydrodynamic model. The concentration of each point is adjusted as it passes an effluent outfall. The second step in the solution involves a concentration adjustment for each point on the moving grid to account for dispersional effects during the time increment. Finally, the source/sink effects are accounted for by yet another readjustment of concentration. These three steps are repeated in sequence for the next time increment and so on. The solution thus passes through time, hourly readjusting concentrations on the grid of moving points. Moving points are added to and removed from the model estuary at its boundaries as they are needed. At the end of each hour, concentrations are extrapolated off the grid of moving points onto the fixed grid of stations used in the hydrodynamic model.

In order to use the tidally varying model to predict BOD and DO concentrations in the estuary, in addition to the tide and river flow information which is needed for the hydrodynamic sub-model, information is required describing the location and quantity of effluent discharges as well as the values of the rate constants which govern the source/sink reactions and the dispersion process. The effluent information is fed into the model as quasi-steady hourly discharge rates over a 25 hour tidal cycle at any of up to 40 different locations on the fixed grid.

CHAPTER 4

APPLICATION OF DISSOLVED OXYGEN MODELS TO THE FRASER RIVER/ESTUARY

The implementation of dissolved oxygen models consists of the following steps: making an abstraction of the physical-hydraulic system to fit the basic model formulations; making assumptions about the various processes involved; applying appropriate coefficients to each of these processes; entering various waste discharge patterns; and observing the predicted results. The application of the tidally averaged and tidally varying dissolved oxygen models to the Fraser River/Estuary will now be considered in light of the basic elements of model implementation.

4.1 THE MODEL RIVER/ESTUARY

The "model river/estuary" used in this study was developed by Joy [1974] in his investigations of estuarine hydraulic behaviour. It covers the lower Fraser River system from the Strait of Georgia to Chilliwack and includes the three principal channels - the Main Arm/Main Stem, the North Arm and the Pitt River system. The schematic layout of the model river/estuary is shown in Figure 4.1 and, as mentioned in Section 3.5.1, was first developed by Joy [1974] to be utilized within the hydrodynamic sub-model of the tidally varying model. This station arrangement is also used for the tidally averaged model.

As the model river/estuary extends to cover that portion of the lower Fraser River influenced by tidal effects, the upstream boundary was chosen to be in the vicinity of Chilliwack, the commonly accepted limit of

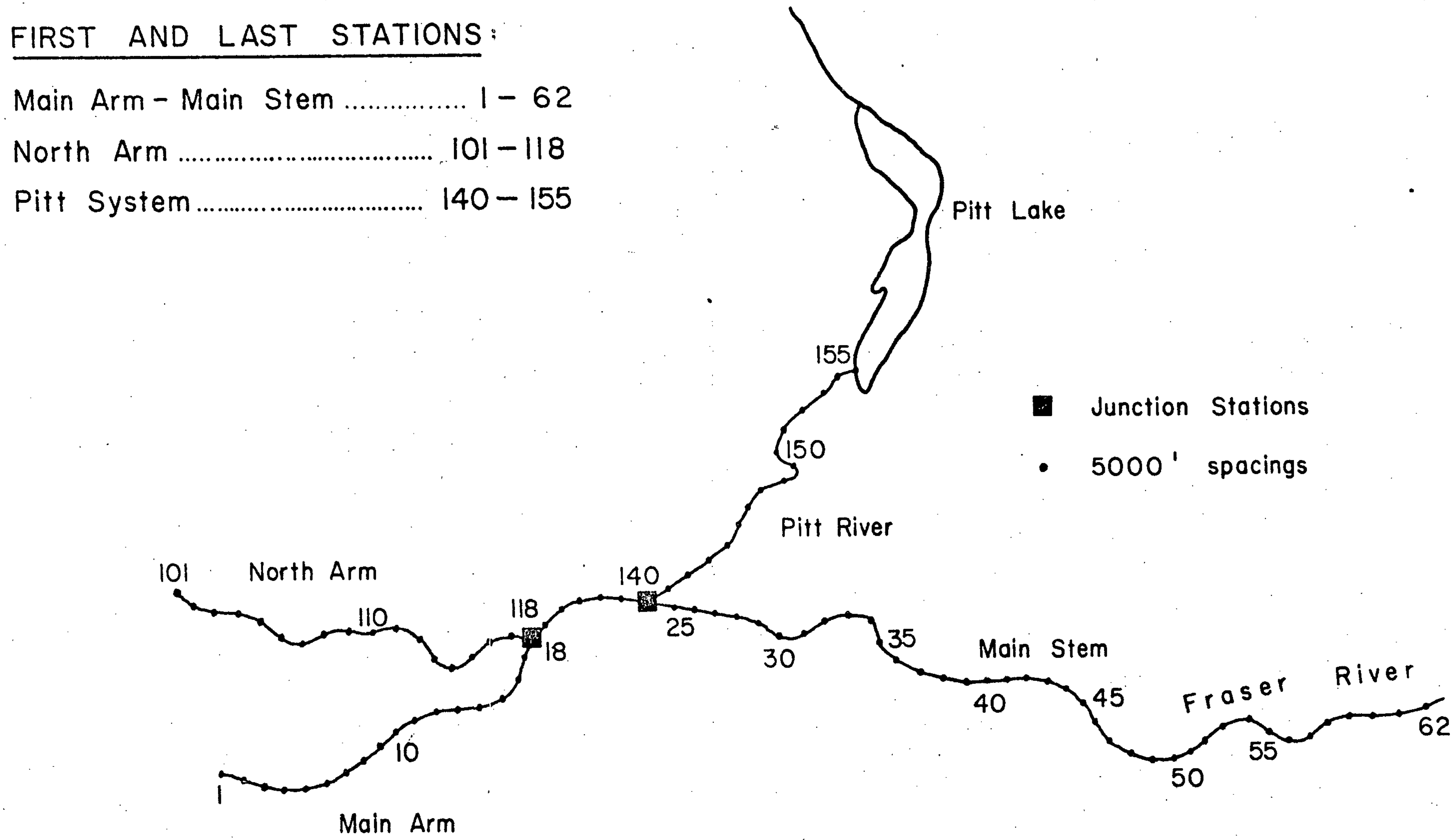


Figure 4.1

Numbering Scheme and Network of Stations Used in the Model River/Estuary tidal influence. Downstream model boundaries are the exits of the River to the Strait of Georgia at Steveston on the Main Arm and Point Grey on the North Arm. Other outlets of the lower Fraser River, the Middle Arm and Canoe Pass, have not been considered in the model. It should be noted, however, that their presence has been accounted for implicitly by areal adjustments of the Main Arm and North Arm exits. The Pitt River system was included within the bounds of the model because of its importance as a tidal storage area.

The numbering scheme and network of stations used in the model river/estuary are also shown in Figure 4.1. Segment length was arbitrarily chosen to be 5,000 feet. The Main Arm/Main Stem extends from Steveston to Chilliwack Mountain (station numbers 1 to 62); the North Arm, from Point Grey to New Westminster (station numbers 101 to 118) where it joins the Main Stem; and the Pitt River from the Main Stem junction to Pitt Lake (station numbers 140 to 155). In all cases only the main core of advective flow of the major channels has been considered.

Values of local low water depth, cross-sectional area, and river width for each station were obtained from hydrographic charts supplied by the Department of Public Works [DPW, 1970]. These parameters were adjusted to compensate for the presence of major side channels as was the case in the Main Arm and North Arm exits of the river and also to fit the advective flow core. Figures 4.2 and 4.3 show the variation of the gross and advective values of these parameters along each of the three channels considered by the models.

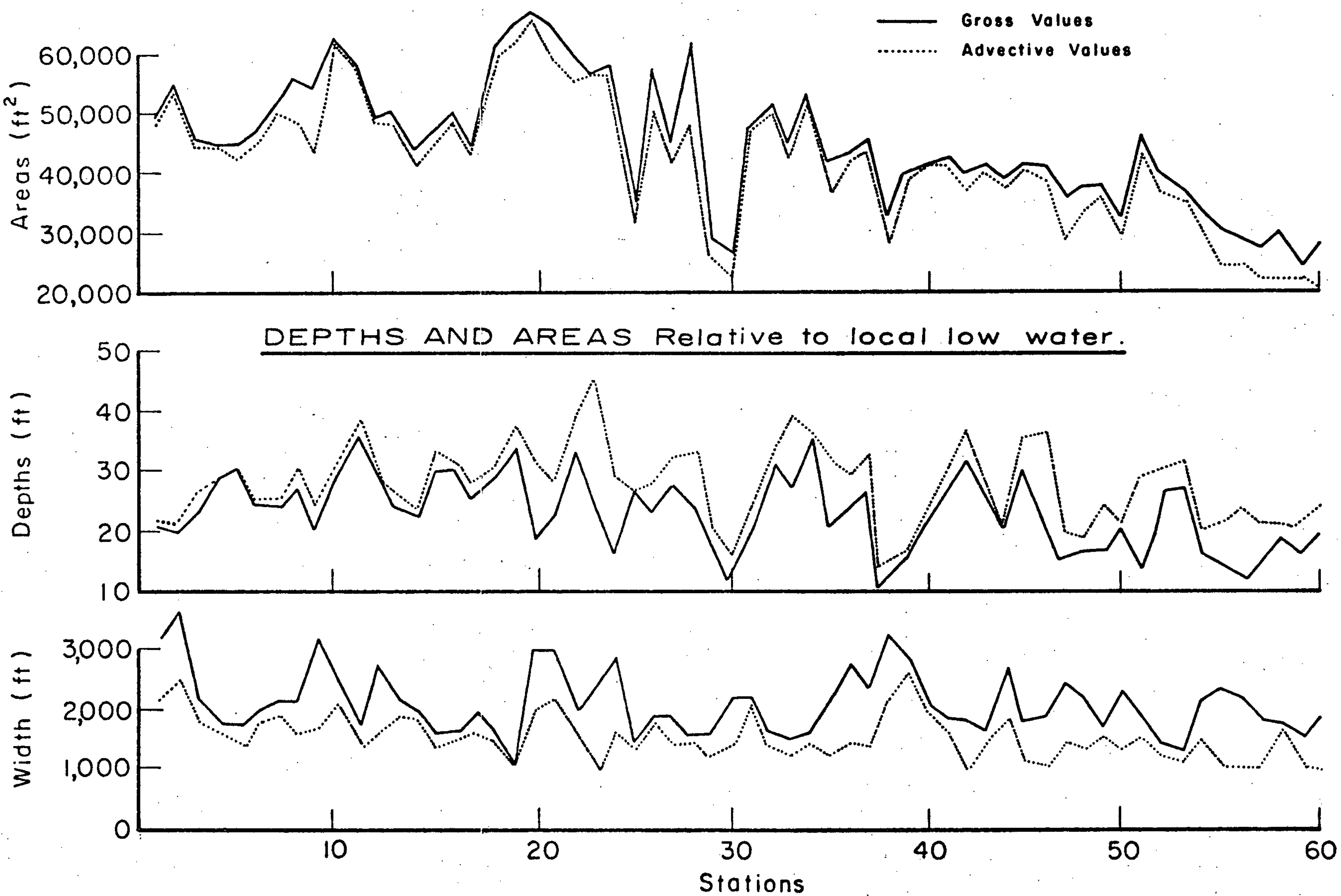


Figure 4.2

Variation of Cross-Sectional Parameters in the
Main Arm/Main Stem

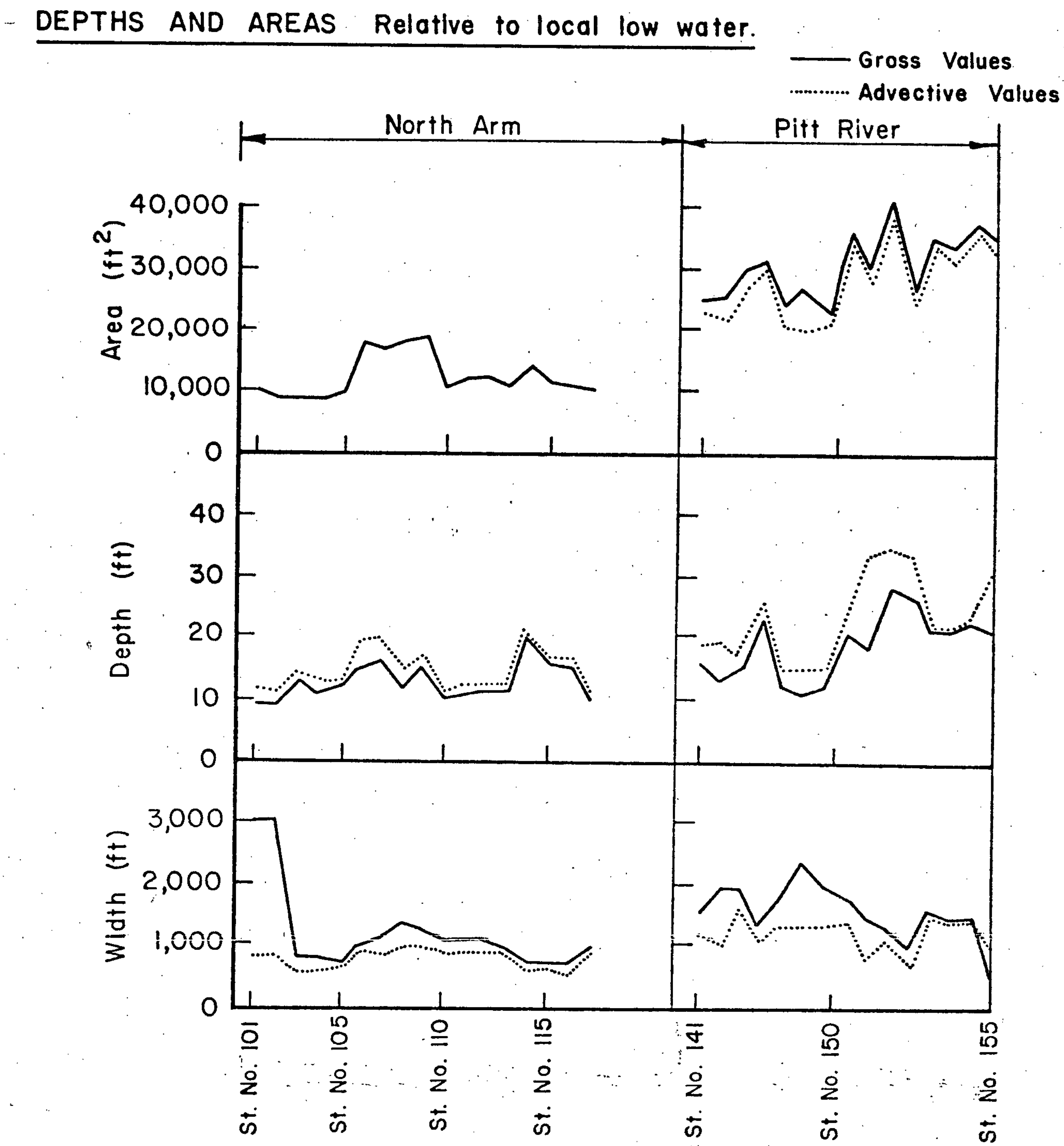


Figure 4.3

Variation of Cross-Sectional Parameters in the North Arm and Pitt River

4.2 IMPLEMENTATION OF THE MODELS

In Chapter 3 the general theory, formulations and solution of estuary dissolved oxygen models were discussed. The implementation of these models as they apply to the lower Fraser River/Estary is now considered.

4.2.1 Tidally Averaged Models. Two approaches to tidally averaged, steady-state modeling were discussed in Section 3.4: the continuous solution approach and the finite section approach. Of these, only the latter could be applied to the full extent of the lower Fraser River. The implementation of the continuous analytical solutions which offered advantages over the finite section approach both in terms of ease of development and the straightforward manner of solution was found to be impossible. The analytical solutions for estuarine BOD and DO distributions contained modified Bessel functions (see Section 3.4.1). When the application of these solutions to a steady-state Fraser River dissolved oxygen model was first considered, it was known that solutions of these functions were readily available in the form of package programs in the Computing Centre General Library [UBC Function, 1973]. The development of the model was initiated and it was not until the programming was completed that problems were encountered. Because of the relatively small changes in cross-sectional area with distance in the Fraser system, the arguments of the modified Bessel functions utilized in the solution proved to be much larger over most of the model extent than the limits allowed by the Bessel function package programs. As such, the values of the modified Bessel functions were indeterminate.

Although it should be possible to reprogram the modified Bessel function solutions to extend the limits of the arguments, this was well beyond the author's capabilities. Thus the continuous solution approach to the steady-state dissolved oxygen model was abandoned in favour of the finite section tidally averaged model.

The finite section solution approach to modeling estuarine mass transport (see Section 3.4.2) was successfully applied to the lower Fraser system using the segmented representation of the river/estuary shown in Figure 4.1. This station arrangement resulted in a total of 92 segments in the three branches of the river/estuary which required the solution of a system of 93 simultaneous linear equations. The solution for solving this large system of equations which involved matrix inversion procedures was complicated by the branched configuration of the river/estuary. The usual solution technique suggested by Thomann [1971] had to be modified slightly to allow for the coupling of the North Arm and Pitt River to the Main Stem. A procedure was developed which allowed for matrix solution to the three component system of equations without having to resort to techniques involving solution of individual matrices. It was achieved by using a single matrix with additional terms placed in appropriate locations to account for mass transport through the junction stations. The matrix A of Equation 3.11 as applied to the lower Fraser system, shown in Figure 4.4, in effect is partitioned into three separate blocks each of which represents a single channel. The blocks are uncoupled except at the junction stations where coupling is accomplished through the use of an additional term which differs from the main elements of the matrix in that it is not tri-diagonal.

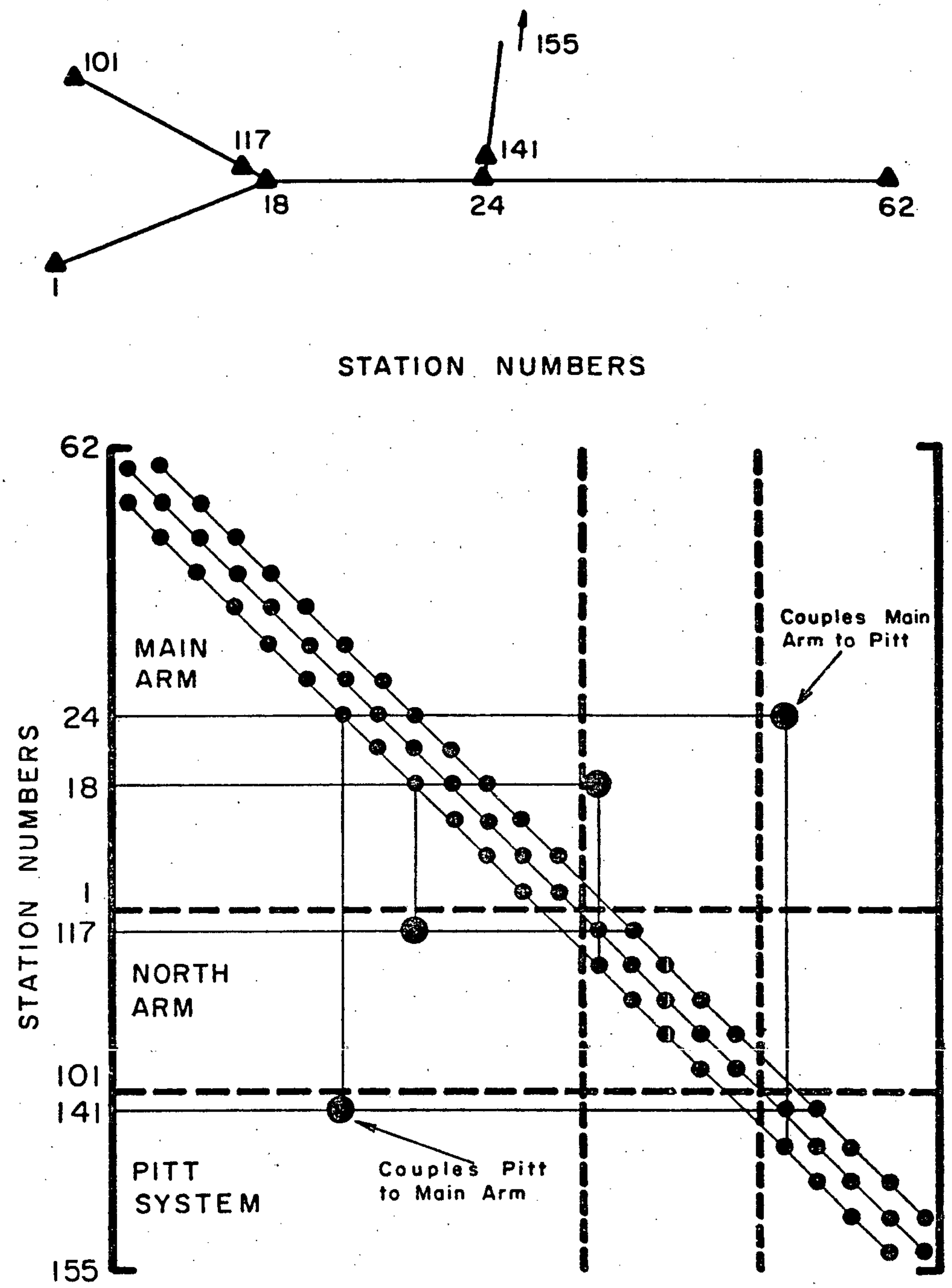


Figure 4.4

The Matrix [A] of the Tidally Averaged Model

The tidally averaged model was programmed to FORTRAN, making use of package program matrix inversion routines [UBC Matrix, 1973] for solution by digital computer.

4.2.2 The Tidally Varying Model. Joy [1974 and 1975], through use of a hydrodynamic sub-model, derived tidally varying solutions to the mass transport equations which he applied to the lower Fraser River/ Estuary to form the basic tidally varying model (see Section 3.5). As Joy's investigation dealt with conditions in the estuary caused by the discharge of conservative substances, all that remained was to implement the coupled BOD-DO system to the tidally varying model and apply the model to the purpose of this research, namely, the prediction of dissolved oxygen levels. The following is a brief summary of the implementation of the tidally varying mass transport model and its application to dissolved oxygen modeling.

The hydrodynamic sub-model, the operation of which is preliminary to the mass transport model, yields output in the form of half-hourly predicted values of velocity and cross-sectional area over a tidal cycle for any of the stations in the model estuary (see Figure 4.1) given the fresh-water discharge conditions at Chilliwack and tidal variation at Steveston. In applying the hydrodynamic model, a design tidal cycle specifying the selected freshwater discharge at Chilliwack and tidal conditions at Steveston is chosen along with assumed initial values of velocity and water surface elevations. Running the model for several tidal cycles results in convergence of these initial values of velocity and water surface elevation to the "true" values for the given conditions. During the solution, it is assumed that the river discharge and tidal conditions are quasi-steady over the period of analysis. The hydrodynamic model has been calibrated under both high tide-low flow and high tide-high flow conditions and found to adequately reproduce water surface elevations [Joy, 1974]. An accurate verification of predicted velocities has not been possible because of the lack of the necessary field measurements. However, based on personal observations and isolated field measurements, the predicted velocities appear to be reasonable. It should be noted that the hydrodynamic sub-model, which does not account for the presence of the salt wedge, may underestimate velocities during certain tidal conditions. This was pointed out in more recent research into hydraulic modeling of the Fraser River/Estuary [Hodgins, 1975].

Selected output from the hydrodynamic sub-model in the form of hourly values of velocity and cross-sectional areas along with independent estimates of dispersion is used as input to the tidally varying mass transport model. The model advects water parcels along the estuary re-adjusting hourly the concentrations of any dissolved substance, in this case BOD, to account for the effects of waste addition and dispersion. In order to use the tidally varying mass transport model to predict dissolved oxygen concentrations, analytical solutions to the basic Streeter-Phelps equation (see Section 2.7) are incorporated in the solution scheme to readjust dissolved oxygen concentrations hourly according to the BOD concentration and the values of the deoxygenation and reaeration rate coefficients. The resultant model output gives a time-history of dissolved oxygen concentration throughout the estuary. The hydrodynamic sub-model and the tidally varying mass transport model are programmed in high speed FORTRAN for solution by digital computer.

4.3 MODEL ASSUMPTIONS

Although the model assumptions were discussed in the chapter on dissolved oxygen model development, they will be briefly reviewed here in a comparative context.

4.3.1 General Assumptions. The basic and, perhaps, most restrictive assumption which applies to both the tidally averaged and tidally varying models is that of approximating the mass transport (and hydrodynamic) process by one dimensional equations. The one dimensional space assumption requires that all variables and parameters be assigned their cross-sectionally averaged values. Thus the models are not able to "see" variations over the width or depth of the river; changes are "seen" only in the longitudinal direction. By this assumption the saltwater wedge effects are ignored. As well, this assumption requires waste inputs to the models to be treated as being "completely mixed", either over the cross-section or, as is the case in the finite section steady-state model, within a segment.

In terms of time, the models view conditions differently. The tidally averaged models, with the assumption of "steady-state" conditions which in essence eliminates the time variable, are a temporal abstraction of the physical system. As all parameters are assigned their average tidal values, the models do not "see" "real time" effects such as current reversal which occur within the tidal cycle. The tidal averaging process may be thought of as a representation of the response of the estuary over a number of tidal cycles and, as such, models based on this approach are often referred to as "inter-tidal". The complex, oscillatory flow field in the estuary is replaced by a freshwater flow field and a tidal dispersion term which accounts for current reversal as well as tidal mixing. Thus, the real time effects caused by tidal action, although not "seen" by the tidally averaged models, are taken into consideration implicitly and the output from the models, which is in the form of tidally averaged concentrations, may be thought of as integrating over a tidal cycle the real response of the estuary.

The tidally varying model does not suffer from lack of temporal resolution. As real time effects caused by tidal action are accounted for in the hydrodynamic sub-model, this model attempts to more accurately represent the true nature of response in the estuary. By approximating conditions hourly, the tidally varying model can "see" changes that occur within the tidal cycle and as such is often referred to as a "real time" or "intra-tidal" model.

Because of the aforementioned differences in temporal resolution, waste discharge information is handled differently in each of the models. In the tidally averaged models, effluent discharges are fed in as average daily loadings. In the tidally varying models, however, in order to keep all process information compatible, waste discharge information is fed in as hourly loadings. As a result, it is possible to vary discharge rates within the tidal cycle, thus making practicable the investigation of short term loadings such as slug loads from storm water overflow or accidental spills.

4.3.2 Dissolved Oxygen Assumptions. In the application of dissolved oxygen models to the lower Fraser River it has been assumed that only two factors affect the oxygen balance: biochemical oxidation of organic matter and atmospheric reaeration. Other oxygen source/sink processes are assumed to be either inoperative or, if present, of little significance compared to the main processes, and as such have not been considered in the models. Photosynthetic oxygen production has not been considered to be important because of the high natural turbidity of Fraser River water which blocks the penetration of light. The effects of oxygen production by aquatic plants and weeds have been ignored because of the noticeable absence of aquatic plant growth on the riverbanks. As for sink processes, benthic oxygen demands have been assumed to be insignificant because the high tidal velocities in the lower Fraser generally prohibit settling of suspended material. This assumption also results in the exclusion of settling effects as a sink of BOD. Nitrification is another oxygen sink process which has not been included in the dissolved oxygen models. This process is usually associated only with high water temperatures and, as noted in Section 2.4.2, the effects of nitrification are considered to be insignificant at water temperatures below 12 ± 4°C. Also, unless wastes are highly treated, there generally is a lag of 5 - 10 days before its effects become prevalent. As water temperatures in the Fraser are low (see Figures 1.6 and 1.7) except during the summer months when residence time in the estuary is generally less than 2 days and because wastewaters receive the equivalent of or less than primary treatment, it is reasonable to assume that nitrification effects can be ignored. Atmospheric reaeration was assumed to be the only oxygen source process, that is, wind and surface wave effects, etc. were ignored.

4.4 MODEL COEFFICIENTS

4.4.1 Dissolved Oxygen Model Rate Coefficients. The selection and evaluation of model rate coefficients is the most crucial step in model application. As the coefficients to a large extent control model output response, their suitability ultimately determines the ability of the models to represent the physical system. Traditionally, the appropriate dissolved oxygen model coefficients for deoxygenation and reaeration are selected by a calibration procedure during which model response is "tuned" to fit the system response by adjustment of the coefficients. Usually the reaeration rate coefficients are calculated from prediction equations based on hydraulic considerations and then, with all inputs to the models equivalent to their counterparts in the physical system, the BOD decay rates are obtained during the model calibration, a procedure known as "verification".

Model verification requires, firstly, that there be a measurable dissolved oxygen response in the physical system and, secondly, that sufficient data be available to fully document this response. In the application of dissolved oxygen models to the lower Fraser River/Estuary, the calibration and verification procedures are extremely difficult. At present, Fraser River dissolved oxygen levels are generally at or near saturation, which means that there is little or no observable response in the oxygen dynamics of the system to present waste discharge patterns. In addition, during those isolated instances when substantial oxygen depletions have been measured, as was the case in June and July, 1970 [Fisheries Service, unpublished data, 1970], the available documentation of the estuary dissolved oxygen response is of insufficient spatial and temporal resolution to be of use in model calibration. As a result, it is not practicable to obtain the model coefficients by verification of Fraser River dissolved oxygen models. Consequently, the evaluation of model coefficients was carried out solely by estimation.

In this study, the reaeration rate coefficients are based on the predictive equation (Equation 2.11) of Dobbin and O'Connor [1956] using a temperature correction factor (θ) of 1.024. The appropriate form of velocity to be used in applying this formula to estuaries is, according to Thomann [1971], the mean tidal velocity which in the lower Fraser River ranges from 0.4 to 8.0 feet per second. Assuming an average depth of 25 feet, the reaeration coefficient (K_2) is found to vary from 0.07 to 0.30 per day.

In attempts to evaluate the rate coefficients of BOD decay appropriate for the lower Fraser, laboratory investigations of BOD response were conducted. These studies which were carried out in conjunction with the Westwater Research Centre as part of an investigation designed to evaluate rate coefficients and determine temperature and salinity effects on bacterial die-off, as well as BOD decay in Fraser River water, yielded inconclusive results [Westwater Research Centre, unpublished data]. Consequently, the rates chosen for use in this research had to be taken from the literature which described other investigations of dissolved oxygen dynamics. The range of BOD decay coefficients (K_1) used in the models and considered to be appropriate for rivers with low pollution such as the Fraser is 0.09 to 0.20 per day. A temperature correction factor (Θ) of 1.135 was used.

4.4.2 Dispersion Coefficients. Because the view taken of estuarine dispersion processes is different in each of the models, a brief discussion of dispersion coefficients is deemed necessary in order to resolve possible confusion which might exist over the use of the terms. In the tidally averaged models the tidal dispersion coefficient, by design, accounts for all tidal effects including upstream water movement and tidal mixing. Thus, it bears little resemblance to the term used to describe longitudinal dispersion in the tidally varying model as this coefficient is based on theories which describe "real" dispersion phenomena, the processes by which concentration peaks are eroded and mass is redistributed in an estuary due to the effects of turbulent diffusion and vertical and lateral velocity gradients. Since the tidal dispersion coefficient is an abstraction of true estuarine mixing phenomena, it has no real theoretical basis and, although several semi-theoretical formulations have been postulated, their use is questionable [Thomann, 1971]. As a result, tidal mixing coefficients are usually evaluated empirically through use of some observable tracer in the estuary such as salinity or chloride concentration. Values of the tidal dispersion coefficient listed by Thomann [1971] for various estuaries range from 1 to 20 square miles per day with a mean value of about 10 square miles per day. In the lower Fraser River/Estuary it is impracticable to establish empirical values of the tidal dispersion coefficient because of the unsteady stratified nature of salinity intrusion (see Section 1.4.2) and since it is not possible through use of presently available data to define a reliable steady-state salinity distribution. Therefore assumed values of this coefficient must be used. It should be noted, however, that as the lower Fraser River/Estuary is "freshwater dominated", as will become evident in subsequent discussion, the tidally averaged model response is relatively insensitive to assumptions regarding the values of tidal dispersion coefficient.

In the tidally varying model, the coefficient of longitudinal dispersion has been assumed to be zero, that is, all dispersive effects have been ignored. The reasons for doing this are twofold. Firstly, it has been established from the results of the recent dye tracer study that due to the inhibitory effect of stratified flow conditions, longitudinal dispersion coefficients in the lower Fraser River/Estuary are low [P. Ward, unpublished data]. Thus, ignoring the effects of longitudinal dispersion should not severely restrict the applicability of predicted concentration to approximate Fraser River/Estuary conditions. Secondly, as this assumption eliminates the dispersive effects of erosion and redistribution of concentration peaks, it will result in an exaggeration of concentration peaks, giving a clearer picture of intra-tidal response in the river/estuary.

4.5 WASTE LOADINGS

4.5.1 Present Waste Loads. Estimates of present waste loadings entering the lower Fraser River/Estuary are shown in Table 4.1 for the Main Arm/Main Stem and Table 4.2 for the North Arm. Average loading rates expressed in terms of pounds of BOD per day are tabulated according to location within the model river/estuary as specified by model segment number. These waste loading estimates are based on effluent discharge permit information supplied by the lower mainland district office of the Pollution Control Branch, New Westminster [PCB, unpublished data, 1973]. The present BOD loading to the lower Fraser totals some 250,000 pounds of BOD per day, of which approximately two-thirds is contributed by municipal sources and the remaining one-third by industrial sources.

4.5.2 Possible Future Waste Loads. An investigation of the possible impact of future waste discharge patterns on the dissolved oxygen dynamics of the lower Fraser River/Estuary requires assumptions of possible future waste loads. Instead of attempting to forecast the magnitudes and locations of future waste loads, hypothetical waste loads will be used in this study. These loadings will be arranged in a manner that, rather than typifying what might be expected as a future condition, will show what can be considered a severe future impact. Basically this involves locating the waste outfalls at positions in the river/estuary where the discharges will have an exaggerated effect.

4.6 MODEL OUTPUT

Since present organic discharges are absorbed by the lower Fraser without causing significant depletion of dissolved oxygen, it is not

TABLE 4.1

WASTE LOADINGS TO THE MAIN ARM/MAIN STEM, LOWER FRASER RIVER

MODEL SEGMENT	PRESENT BOD LOADING (LBS/DAY)
1	720
2	1,730
3	52,470
4	1,760
6	50
7	3,490
11	5,040
14	13,780
15	3,300
16	810
17	7,350
18	990
20	37,420
21	5,470
23	2,550
24	2,520
25	50
29	600
30	20
31	1,800
32	3,690
37	2,410
46	380
47	1,500
51	8,780

TABLE 4.2

WASTE LOADINGS TO THE NORTH ARM, LOWER FRASER RIVER

MODEL SEGMENT	PRESENT BOD LOADING (LBS/DAY)
106	10,250
107	1,040
108	4,450
109	3,120
110	11,690
111	120
112	4,950
114	1,000
115	10
116	4,850

possible at this time to validate dissolved oxygen model output. The lack of observable system response to which model response could be compared by way of verification prohibits accurate calibration of the dissolved oxygen models. As such, the results from the models, presented and discussed in Chapter 5, should be viewed with caution since unverified model output is at best considered to be only a series of scenarios representing sets of possible outcomes.

CHAPTER 5

DISSOLVED OXYGEN MODEL RESULTS

Since the traditional model calibration procedures are not applicable to the lower Fraser River/Estuary dissolved oxygen models, verification of the models is not presently possible. It follows therefore, that model predictions cannot be viewed with complete confidence. In order to alleviate those doubts associated with the mechanics of the model, a sensitivity analysis can be carried out to document model response over the expected range of input parameter variation for the full range of conceivable model coefficients. If, by this analysis, model response is found to be reasonable, the mechanics of the models can be accepted as being valid and the remaining doubts as to the accuracy and validity of model predictions lie in choosing the correct value for each model coefficient. The sensitivity analysis, as well as being a "check" of model validity, is also useful in that it brings to light some of the details peculiar to the nature of lower Fraser dissolved oxygen dynamics. Thus it is more than just a necessary preliminary to the ensuing discussion on dissolved oxygen model predictions, because as well as determining if the models are "well behaved", it also affords us with a focus upon which to base initial discussions on the assimilative capacity of the lower Fraser River/Estuary.

5.1 TIDALLY AVERAGED DISSOLVED OXYGEN MODEL RESPONSE

A study of model response characteristics through investigation of the sensitivity of model output to variations in input parameters and model coefficients is best carried out by holding all parameters and coefficients constant except for the particular element of concern, which is allowed to vary within the expected range of values. Repeating this procedure for each parameter and coefficient in turn, enables one to obtain a complete documentation of model response for the full range of anticipated input values. In analyzing the behaviour of the tidally averaged dissolved oxygen model, the effects of variation in six parameters were considered - freshwater flow, waste loading, dispersion, reaeration rate, deoxygenation rate and temperature. The values used in the analysis are specified in Table 5.1.

TABLE 5.1

PARAMETERS AND COEFFICIENTS USED IN SENSITIVITY ANALYSIS

Parameter	Constant Value	Range of Values
Freshwater flow (Q)	40,000 cfs	10,000 to 60,000
Waste loading (W)	1,000,000 lb	100,000 to 1,000,000
Dispersion Coefficient (E)	10 sq. miles/day	0 to 30
Deoxygenation Coefficient (K_1)	0.2/day	0.1 to 0.6
Reaeration Coefficient (K_2)	0.2/day	0.0 to 0.4
Temperature (T)	10.0°C	5.0 to 20.0

The response of the model over the Main Stem reach of the river with waste discharge location at Station 40 in the upstream portion of the model will be used throughout the analysis.

5.1.1 Effect of Freshwater Inflow Variation. The model response to flow variation is shown in Figure 5.1. The effect of increases in freshwater flow in the river/estuary can be seen to result in reduced BOD concentration and increased DO concentration as would be expected due to

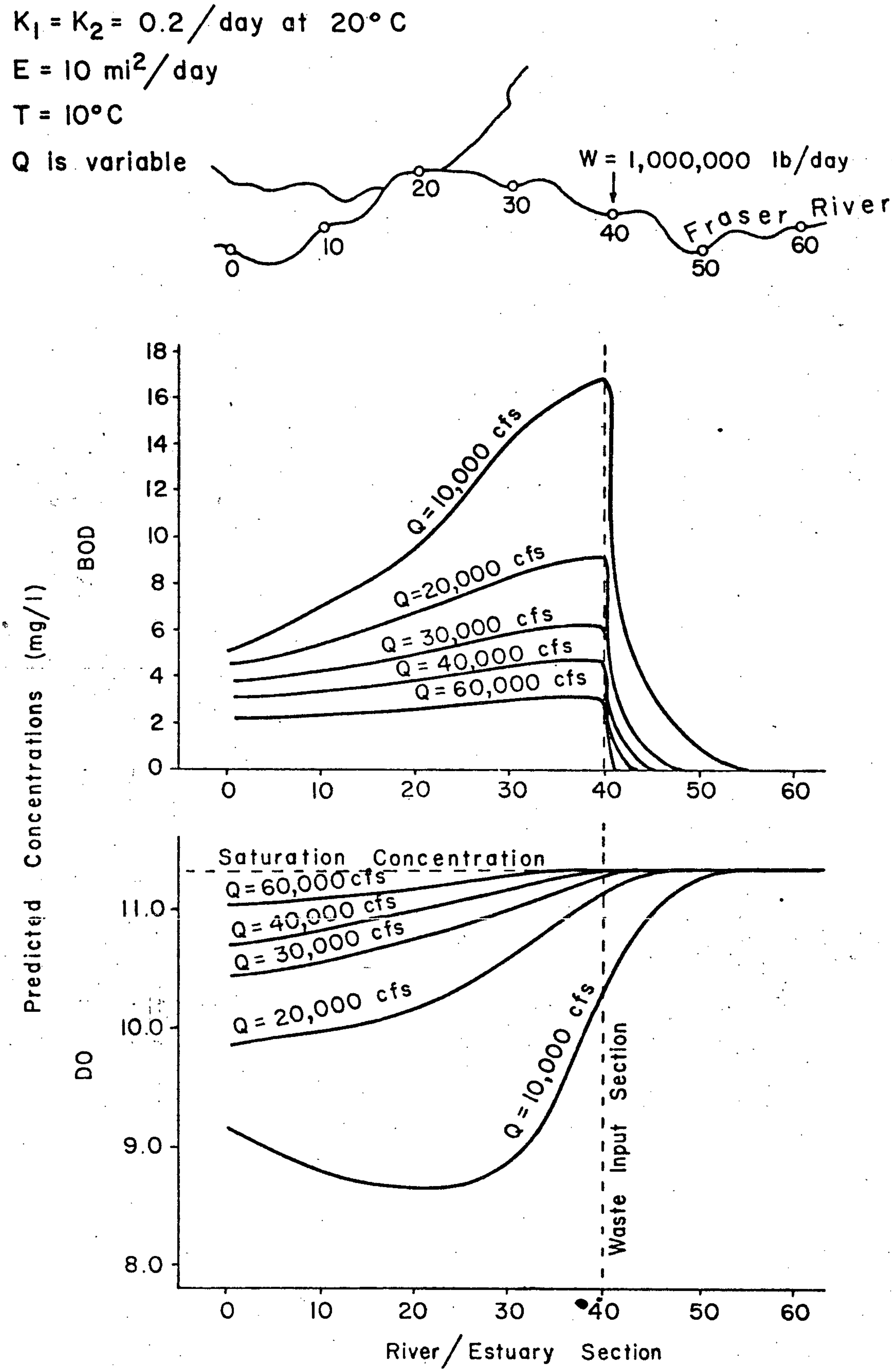


Figure 5.1

Effect of Freshwater Inflow on Model Response the increased dilution afforded by the higher flows. An effect of particular interest is observed for the lowest river flow (Q = 10,000 cfs). The DO distribution in this case, in contrast to the others, is seen to reach a clearly defined minimum in the vicinity of station 25 after which it begins to increase in the seaward direction - the characteristics of a typical oxygen-sag curve.

As this effect is not observed for the higher river flows it is evident that according to model response, it is only during extremely low flows which result in increased flushing times and high BOD concentrations that the condition of maximum DO deficit is reached within the channelized stretch of the river/estuary. At the higher river flows the bulk of the oxygen demand is presumably flushed out into the Strait of Georgia.

Also evident for decreasing river flows is the increasing dominance of tidal influence as evidenced by the increased upstream effects for both BOD and DO. It is noted that if the freshwater inflow to the estuary were zero, the steady state BOD and DO deficit concentrations would be distributed normally about the outfall location. The influence of freshwater inflow is seen to skew the respective distributions in the downstream direction with only the maximum BOD concentration still occurring at the outfall location as the point of maximum DO deficit is shifted downstream. It can be seen in Figure 5.1 that the BOD distribution is skewed significantly and that the point of maximum DO deficit for any except very low river flow is displaced out of the model river/estuary, thus indicating the predominating influence of freshwater inflow. Note that the minimum low flow for the Fraser River at Chilliwack is estimated to be 18,000 cfs [Westwater Research Centre, unpublished data].

5.1.2 Effect of Waste Loading Variation. The model response to various waste loadings is shown in Figure 5.2. Increases in BOD and DO deficit concentrations are found to be related linearly to increases in waste loading rates indicating that the principle of superposition is adhered to by the tidally averaged model. A slight increase in BOD concentration and a corresponding decrease in DO concentration is also observed upstream of the waste discharge location with this effect, due to tidal dispersion, again being found to be related linearly to increases in waste loadings.

5.1.3 Effect of Dispersion Coefficient Variation. In the tidally averaged model all tidal effects are accounted for by the tidal dispersion coefficient. It accounts for current reversal and the effects of upstream water movement as well as tidal mixing. The effect on model response for a range of values of this coefficient is shown in Figure 5.3. Included is the case of zero dispersion which converts the estuary model into a model applicable to rivers. Note that in this case the tidal exchange coefficient (α) must be set equal to 1.0. It is evident that variation of this coefficient has a rather limited effect on both BOD and DO distributions. This is again evidence of the predominance of freshwater inflow and its effects in the lower Fraser River/Estuary. The effects of varying this model coefficient , although minor, are observable. In the case of zero dispersion, no effects upstream of the outfall are evident indicating that flow is continually in the downstream direction. As the magnitude of the dispersion coefficient is increased, upstream effects become evident and are of increasing significance in terms both of magnitude of BOD and DO deficit concentrations and extent of their upstream influence as the coefficient

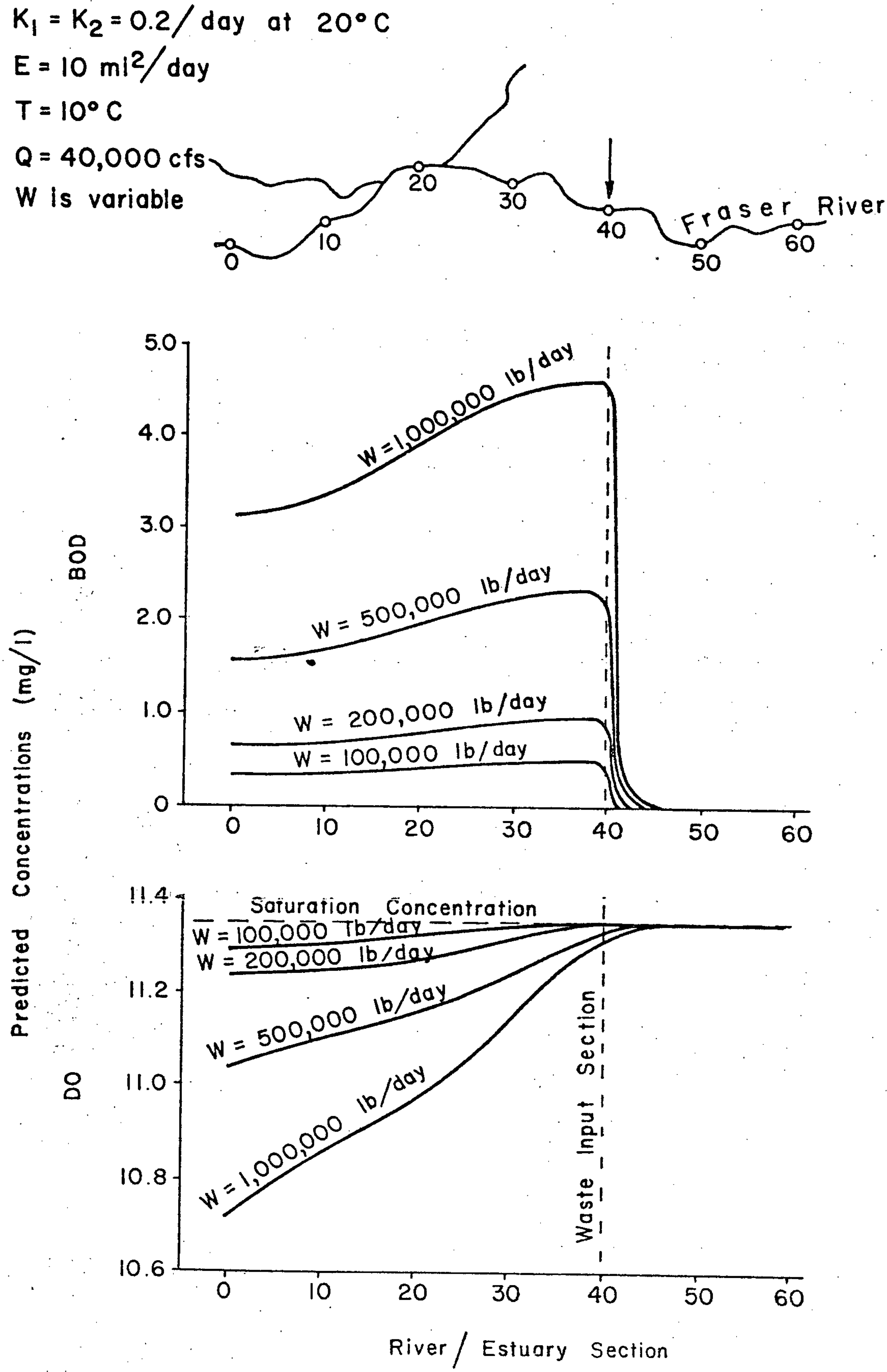


Figure 5.2
Effect of Waste Loading on Model Response

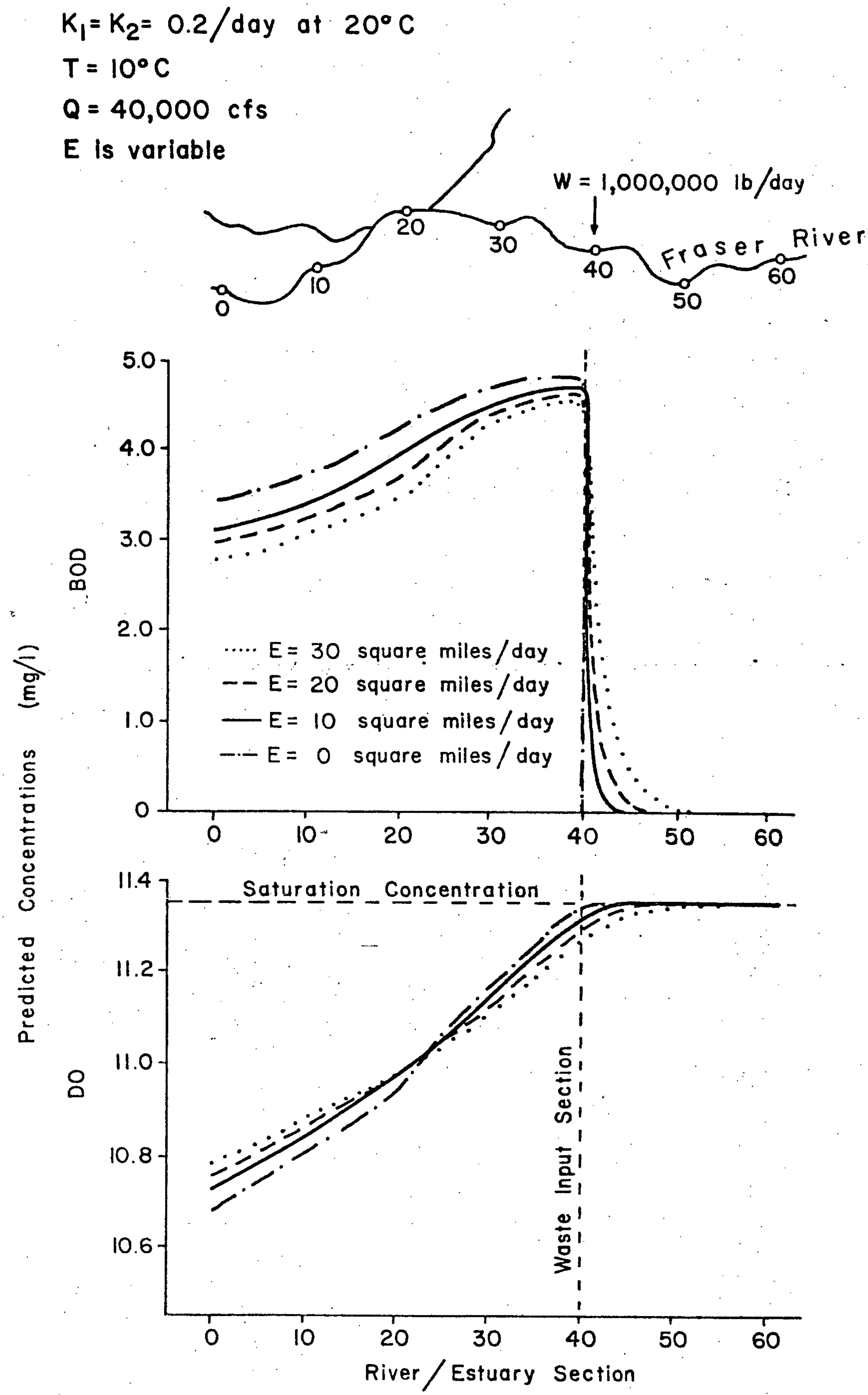


Figure 5.3

Effect of Dispersion on Model Response

approximates the effects of water movement in the river/estuary. In contrast to this, downstream of the outfall increases in the tidal dispersion coefficient have the opposite effect, that is, BOD concentrations are reduced as far upstream as the outfall and DO deficit concentrations are reduced in the downstream reaches. In this region the tidal dispersion coefficient operates in the manner of the "true" dispersion coefficient as it seeks to redistribute mass by minimizing concentration gradients.

5.1.4 Effect of Deoxygenation Rate Coefficient Variation. The effect of variation of deoxygenation rate coefficients on models can be seen to be evident only in the river/estuary stretch downstream of the waste discharge location (see Figure 5.4). Increases in the rate coefficient are observed to result in increased deoxygenation as evidenced by reduced oxygen levels and corresponding decreases in BOD concentrations as a greater portion of the oxygen demand is satisfied. This is typical of the expected response of this coefficient. Note that at the assumed water temperature of 10°C, the effective rates of deoxygenation will be somewhat lower than the magnitudes shown, which are the values at 20°C.

5.1.5 Effect of Reaeration Rate Coefficient Variation. Model response due to variation in the reaeration coefficient is shown in Figure 5.5 with the only effect being the decrease in DO concentration evident for decreases in reaeration rate. Again, as the water temperature is held at 10°C the effective rates of reaeration will be somewhat lower than their specified values which are for 20°C.

5.1.6 Effect of Water Temperature Variation. Water temperature effects on model response are shown in Figure 5.6. The obvious effect is seen as a lower saturation concentration for increasing temperature. As

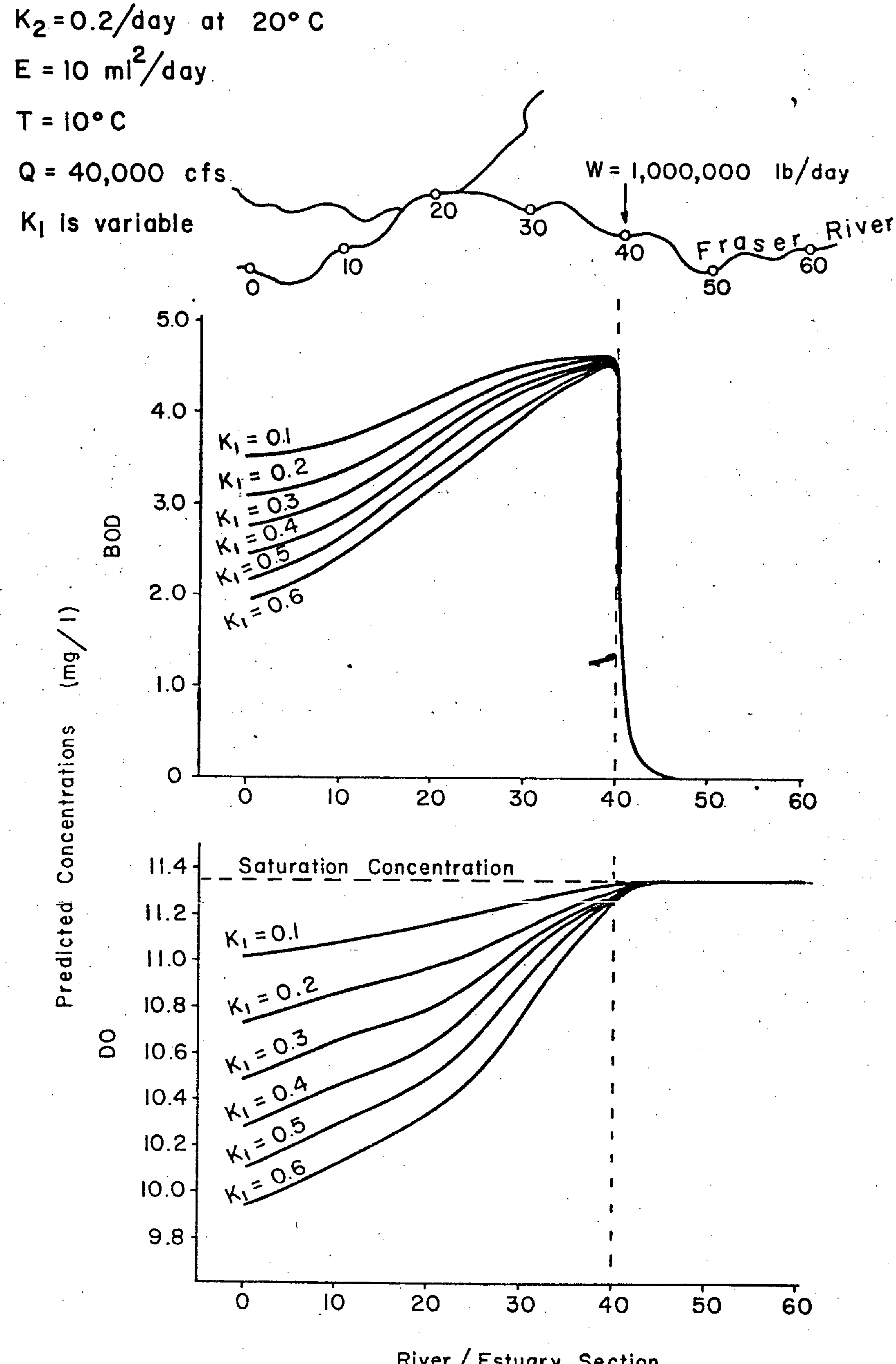


Figure 5.4

Effect of Deoxygenation Rate on Model Response

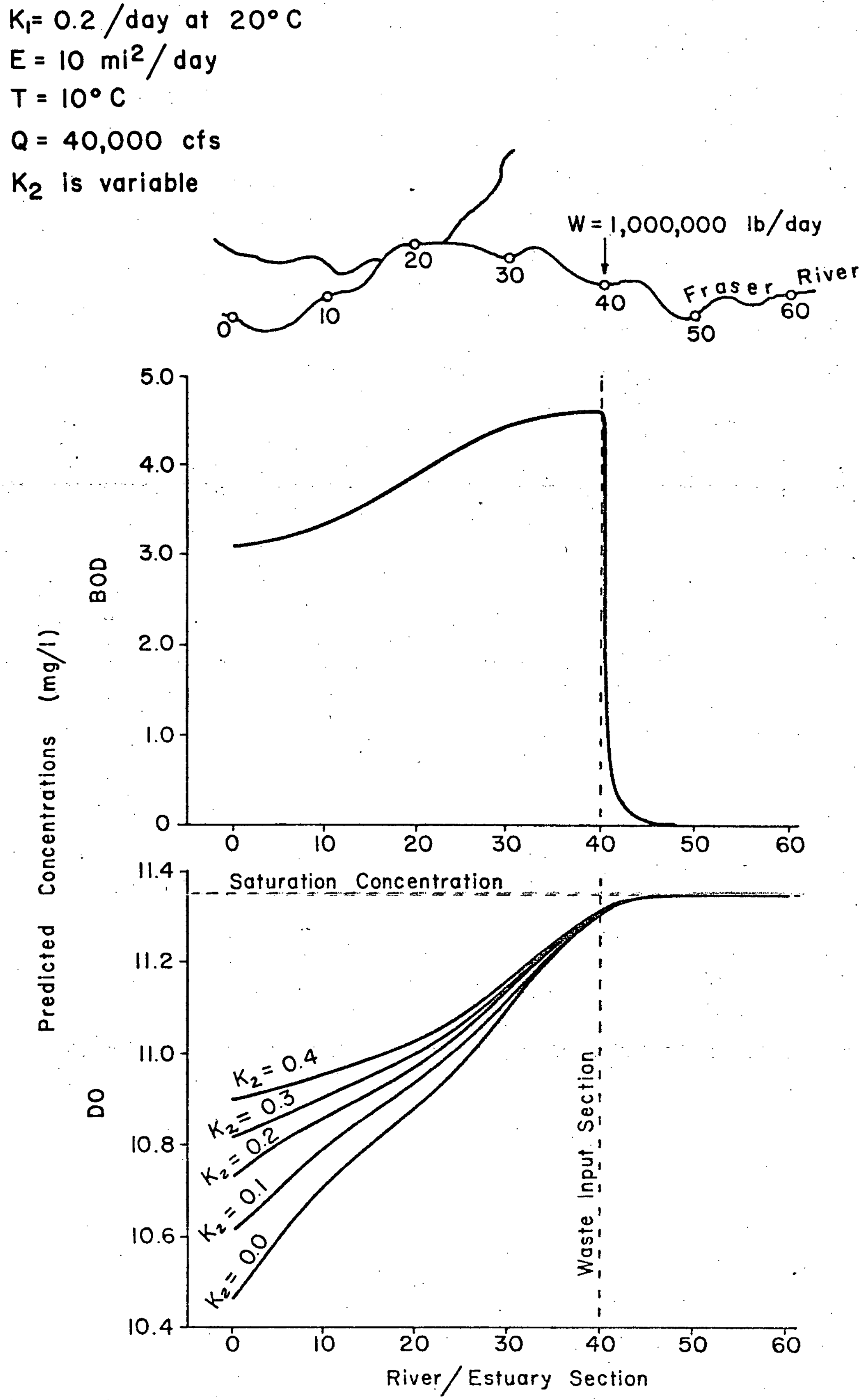


Figure 5.5

Effect of Reaeration Rate on Model Response

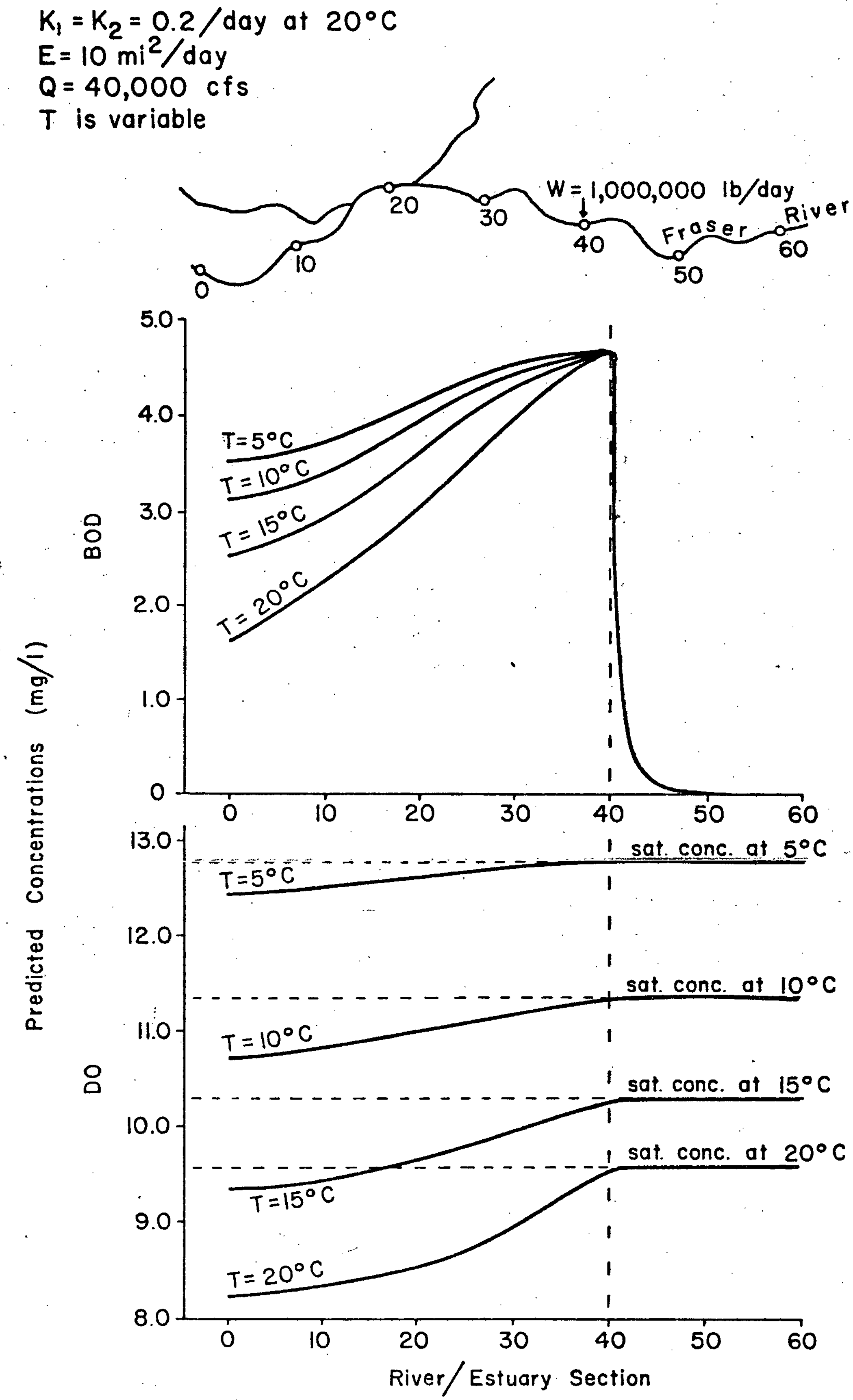


Figure 5.6

Effect of Temperature on Model Response

well, increased deoxygenation as evidenced by decreases in BOD concentration and increases in DO deficit with increasing temperature show the effect of temperature compensation (θ in this case is 1.135) on the deoxygenation rate coefficient ($K_t = K_{2o}\theta^{t-20}$). The corresponding temperature effect on reaeration rate, not observable directly, is less severe (θ being 1.024) and thus its effects are seen to be outweighed by increased deoxygenation.

5.1.7 Summary. The sensitivity analysis has shown that the tidally averaged dissolved oxygen model response is reasonable over the range of input parameters and model coefficients considered by the analysis, that is, model mechanics appear to be sound. Thus the model can be considered to be valid at least in the sense that model response is in the direction it should be. As well, the analysis has pointed out some interesting details regarding lower Fraser dissolved oxygen dynamics. Foremost, the predominating influence of freshwater inflow which minimizes the effects of tidal dispersion is seen to flush oxygen demand out of the river/estuary to be exerted in the Strait of Georgia. In addition, the beneficial effects of low water temperatures have become evident in their dual role of retarding biochemical oxidation and at the same time increase the dissolved oxygen saturation concentration.

5.2 TIDALLY VARYING DISSOLVED OXYGEN MODEL RESPONSE

The utility of a tidally varying model lies in its ability to describe the intra-tidal behaviour of modeled parameters. Thus, the tidally varying dissolved oxygen model affords us with an opportunity to more fully investigate the nature of dissolved oxygen resources in the lower Fraser River/Estuary in that it will allow an assessment to be made of intra-tidal dissolved oxygen response.

Before we can accept the results from this unverified model we must be assured that its manner of response is reasonable. Although it would be desirable to carry out a rigorous investigation of the sensitivities of model response as was done with the tidally averaged model, this is precluded by operational limitations inherent in the tidally varying dissolved oxygen model which result from the complex, multiple model solution format. The main limitation arises from the awkward, unwieldy nature of the solutions,which are costly, not only in terms of computing time, but also in terms of expense of effort because all sub-models must be reprogrammed each time an input parameter or model coefficient is altered. Thus an investigation of the complete sensitivities of this model's response over the required range of input parameter and model coefficient variation is highly impractical, if at all possible. This fact also points out a definite shortcoming of the tidally varying model, namely its rather sophisticated ability to describe the detailed behaviour of the river/estuary has severely restricted over-all model flexibility.

Even though a thorough sensitivity analysis is not possible, it will still be useful to investigate the behaviour of the tidally varying dissolved oxygen model predictions to see how they compare to results from the tidally averaged model. This comparative assessment will also serve as a suitable framework for pointing out some of the details of river/estuary behaviour that cannot be observed through use of the tidally averaged model. The following discussion will deal separately with each component of the multiple model tidally varying solution to illustrate all aspects of model output. As in its present state all components of the tidally varying model are essentially unverified, it will also be worthwhile to briefly assess the validity of the various levels of model output and to indicate how inaccuracies might ultimately have an effect on the validity of tidally varying dissolved oxygen predictions.

5.2.1 Hydrodynamic Sub-Model Output. Typical output from the tidally varying hydrodynamic sub-model is shown in Figure 5.7 for a freshwater inflow of 40,000 cfs at Chilliwack and tidal range of 10 feet at Steveston. Predicted velocities for three stations along the river/ estuary are shown according to their relation with tidal stage at Steveston. Note that the negative velocities indicate flow in the downstream direction. Current reversal is predicted to occur at all three stations. The timing of its occurence is seen to vary from three hours after local low water (LLW) at Station 2 to four and five hours after LLW at Stations 20 and 40, respectively. The design tidal configuration is also seen to result in an extended period of essentially slack water around hour 12 in the tidal cycle.

The advective transport of particles released at various times from Station 40 on the Main Arm/Main Stem of the river/ estuary according to hydrodynamic sub-model velocity predictions is shown in Figure 5.8. Particles 1 and 2 released at hours 8 and 14, respectively are seen to exit during the strong ebb at approximately 70 hours. Particle 3 released at hour 20 and particle 4 released at hour 26 are not seen to

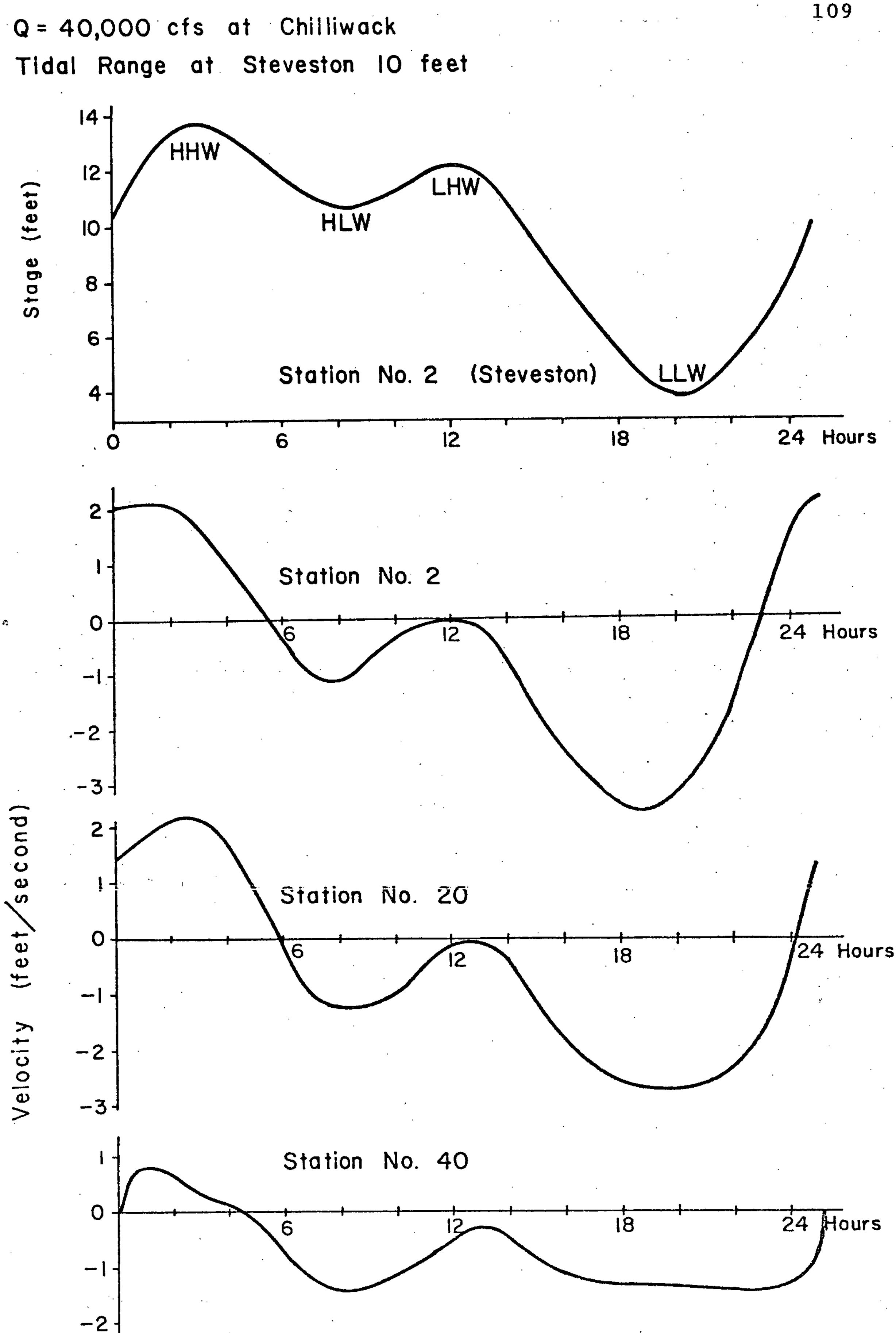


Figure 5.7
Predictions of Tidally Varying Velocity

Q = 40,000 cfs at Chilliwack

Tidal Range at Steveston 10 feet

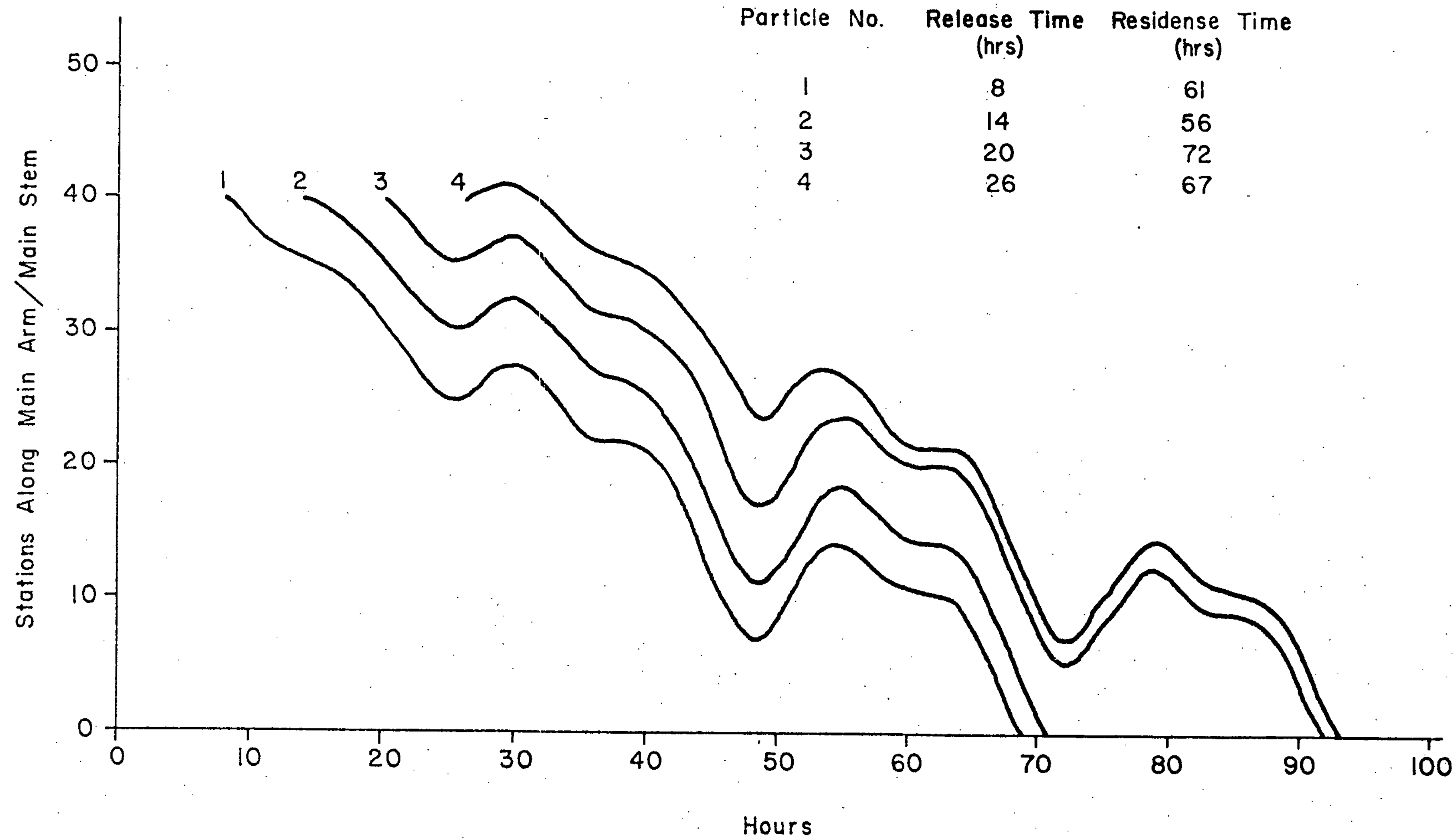


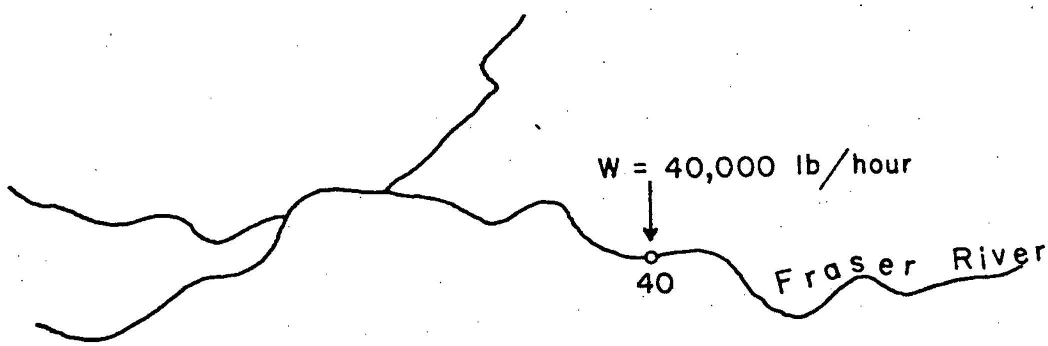
Figure 5.8

Predicted Trace of Particles Released at Various Times From Station 40 exit until the ebb of the following tidal cycle at 90 hours which results in a significant increase in residence times for these particles.

5.2.2 Tidally Varying Initial Effluent Concentrations. When an effluent is discharged into an estuary the oscillatory movement of the water mass results in a variable initial effluent concentration (see Figure 5.9). This is caused by the variations in magnitude and direction of tidal flows at the point of effluent discharge. In part, this is due to the phenomenon of "multiple dosing" which occurs when a parcel of water receives a slug of effluent as it first moves past the discharge point in the downstream direction during an ebb flow; another slug of effluent as it moves upstream past the outfall on the flood tide; and yet another slug of effluent as the water parcel moves downstream on the succeeding ebb tide. Thus flow reversal can result in a water parcel being "dosed" a number of times by the same effluent discharge. Also, any period of extended slack or slow moving water results in decreased effluent dilution which again causes increased effluent concentration. An examination of Figure 5.9 shows three concentration "spikes" in the predicted initial tidally varying BOD concentration profile (BOD_{tv}). The increased concentrations at hours zero and four result from the flow reversals which occur at those times (see Figure 5.7). The concentration "spike" observed at hour 12 is due to an extended period of slow moving water. This latter "spike" is seen to be the highest BOD concentration, being approximately six times higher than the tidally averaged concentration (BOD_{ta}). It was pointed out by Joy [1974] that, according to model predictions, peak tidally varying concentrations could be up to

Q = 40,000 cfs at Chilliwack

Tidal Range at Steveston 10 feet



Continuous Waste Discharge at Station 40

Predicted Initial BOD Concentrations

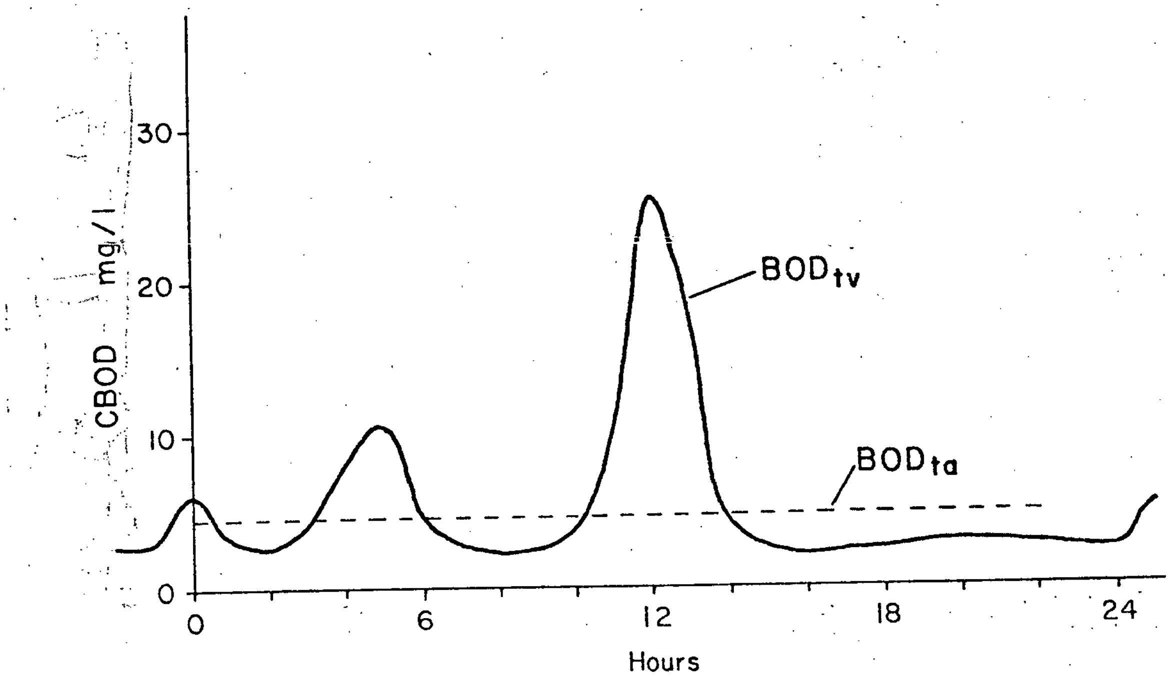


Figure 5.9

Predicted Initial Effluent Dilution

ten times higher than the tidally average values.

5.2.3 Intra-Tidal Dissolved Oxygen Response. To investigate the intra-tidal response of dissolved oxygen in a manner that allows for comparison to the tidally averaged river/estuary response, equivalent river/estuary and waste discharge conditions were chosen: waste discharge equal to 40,000 pounds of BOD per hour at Station 40; water temperature of 10°C; decay (K_1) and reaeration (K_2) coefficients equal to 0.2 per day at 20°C. The river/estuary response as predicted by the tidally varying dissolved oxygen model is shown in Figures 5.10 and 5.11 for downstream Stations 20 and 2, respectively. These figures illustrate the predicted variation in BOD (BOD_{tv}) and DO deficit (DO_{tv}) which, although shown to be continuous curves, are actually approximated by hourly values. The two minor effluent "spikes" that had been distinct in the predicted initial effluent dilution curve (see Figure 5.9) have "blended" together and result in the minor BOD peak at hour one in Figure 5.10. That is to say, the "spikes" arrive at Station 20 within the same hour and since the tidally varying model cannot "see" events within the hour it is unable to distinguish between them. It should be noted that the time scales in Figures 5.10 and 5.11 are arbitrary. The parcel of water which contained the major initial BOD "spike" is seen to have moved downstream completely past Station 20 at hour four and then part way back around hour 14. Thus one slug of water is responsible for the two major BOD peaks in Figure 5.10 and two slugs are responsible for the minor peak. Corresponding peaks of DO deficit are observed.

At Station 2 (see Figure 5.11) the same phenomenon is observed

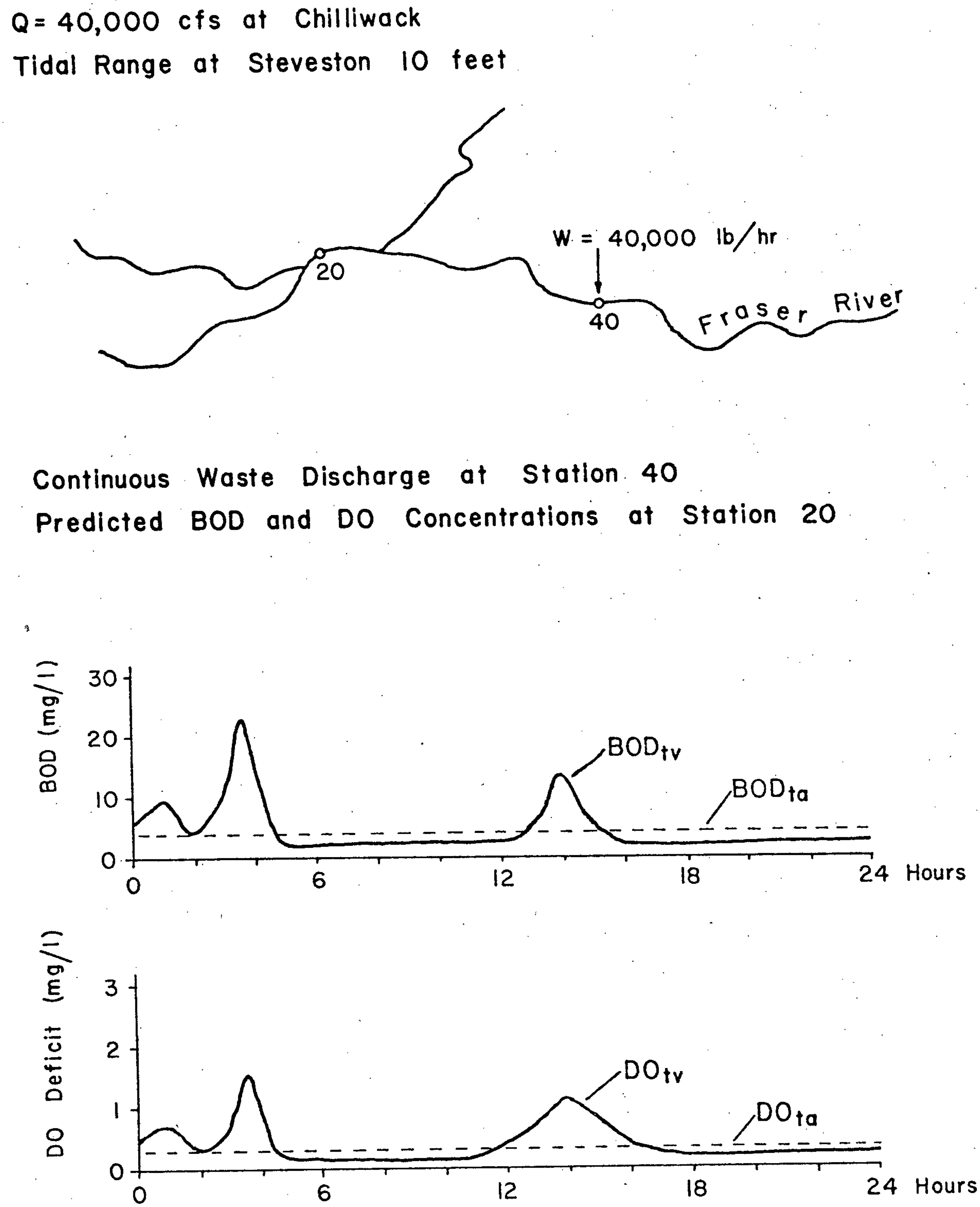


Figure 5.10

Tidally Varying Dissolved Oxygen Model Response: Station 20

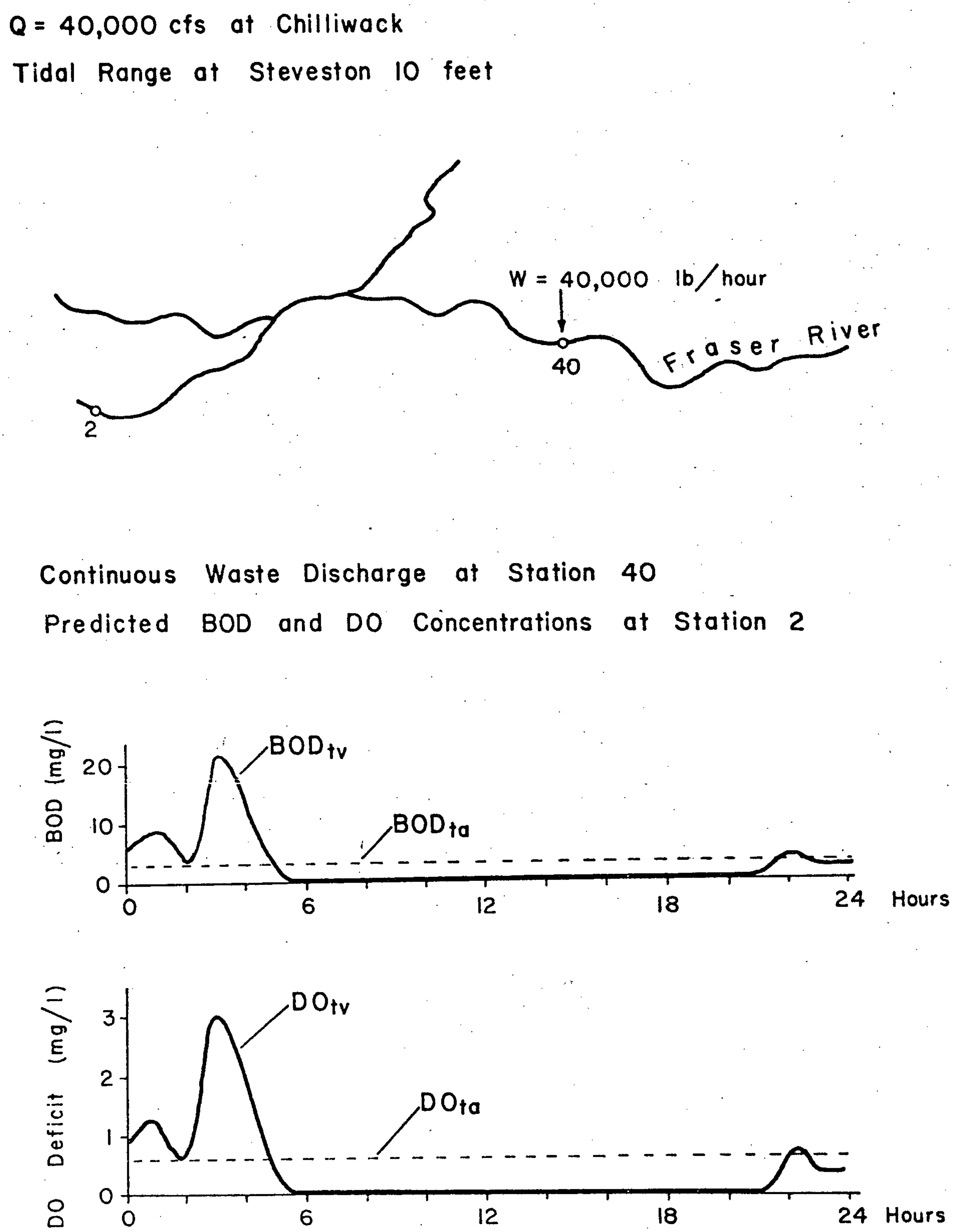


Figure 5.11

Tidally Varying Dissolved Oxygen Model Response:
Station 2

only in this case the major BOD peak is flushed out of the model river/ estuary at hour six. This is evident because the BOD and DO deficit concentrations are observed to be zero between hours six and twenty, which means that sea water which is assumed to be unpolluted has moved into the river/estuary.

A comparison of peak tidally varying DO deficit (DO_{tv}) to the tidally averaged deficit (DO_{ta}) reveals that at both stations the tidally varying deficit is significantly greater with the maximum ratio in each case being around six. The fact that the ratio of peak DO_{tv} to DO_{ta} is equal to the peak BOD_{tv} to BOD_{ta} ratio is to be expected because of the linear nature of the dissolved oxygen solutions. That is for equal initial deficits, constant and equal coefficients and roughly comparable resident times, the relative size of DO deficit concentrations as determined by the Streeter-Phelps oxygen sag equation (Equation 2.3) will be directly proportional to the relative magnitude of initial BOD concentrations. Note that the relative difference between tidally averaged and "actual" residence time of a water parcel in the river/estuary is minimal with the difference becoming proportionally of lesser significance as residence time increases.

This points out the importance of predicted initial effluent dilutions as their values ultimately determine what the tidally varying dissolved oxygen response will be. It should be noted that the above analysis was made neglecting longitudinal dispersion. Thus it represents an extreme case, in that, had dispersive effects been included, the effect would have been to erode the "spikes" thereby redistributing BOD, the net result being decreased maximum DO deficit concentrations. According to Joy [1974], who investigated the predicted dispersion of a slug load released in the upstream river/estuary reaches, there is a five-fold decrease in peak concentration within the first 24 hours after release (see Figure 5.12). Although recent field investigations have established that coefficients used by Joy were likely too large [P. Ward, unpublished data] the degree to which this affects the results is not easily determined. It is recognized that the effects of dispersion on effluent peaks produced by a steady discharge will be somewhat less than in the case of a slug load because of the reduced concentration gradients.

5.2.4 Validity of Tidally Varying Predictions. It was pointed out in Sections 4.2.2 and 5.2 that due to its sophisticated approach the tidally varying model must, of necessity, utilize a staged, multiple model solution format. The basic model, the hydrodynamic sub-model, predicts tidally varying velocities and water surface elevations throughout the river/estuary for given river discharge and tidal conditions. This information, in addition to data describing waste discharges, is used in turn as input to the general mass transport model which ultimately predicts the time-varying distribution of BOD and DO throughout the estuary. Within the framework of the general mass transport model the hydrodynamic information is used to perform two basic functions. Firstly, the velocity field predictions are used as the basis for routing particles released at various times and locations in the river/estuary, thereby simulating advective transport and defining a trace of the time-history of various water and/or effluent parcels. Secondly, the predicted velocity information in combination with the cross-sectional area, which is derived from water

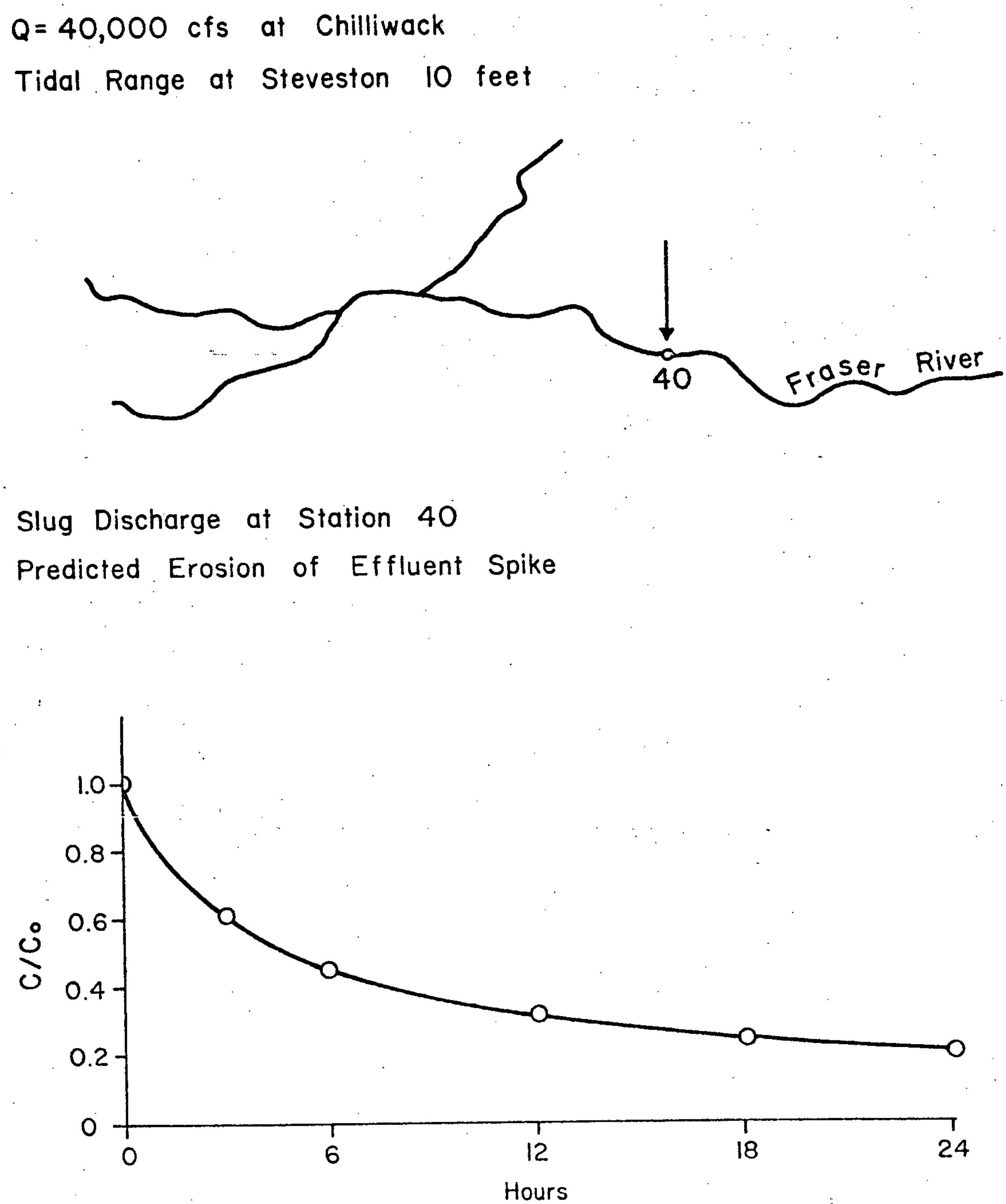


Figure 5.12

Dispersion of a Slug Load

surface elevation predictions, is used to calculate tidal flows which are used in turn to obtain estimates of initial effluent dilution. Thus the accuracy of predicted tidally varying dissolved oxygen response depends to a large extent on the validity of the hydrodynamic sub-model; specifically on the accuracy of tidally varying velocity predictions as they are used to determine residence times as well as initial effluent concentrations.

With regards to possible inaccuracies in the magnitudes of velocities as they might effect residence times, it appears that these could be substantial, particularly in the lower river/estuary stretches where flow stratification effects exist due to the intrusion of saltwater (see Section 1.4.2). Hodgins [1974] developed a modified hydrodynamic model which could account for the saltwater wedge effects. He found that the main effect of the saltwater layer was to increase the freshwater ebb velocities which resulted in sufficiently different rates of particle advection. A comparison of the advection paths as they are predicted by the stratified model and the barotropic model reveals that the effects of velocity underestimation are two-fold. Firstly there is a substantial error in predicted residence time. For example, consider particles released from Annacis Island (particles 3 and 4 in Figure 5.13). Particle 3 in the stratified model is seen to be flushed out of the river/estuary approximately nine hours faster than its counterpart particle 4 in the barotropic model, a reduction in residence time of over 30 percent. The second effect evident from the trace of particles 1 and 2 in Figure 5.12 is that the underestimation of velocities results in a greater degree of multiple "dosing", in this case, four separate dosings as compared with only two in the stratified model.

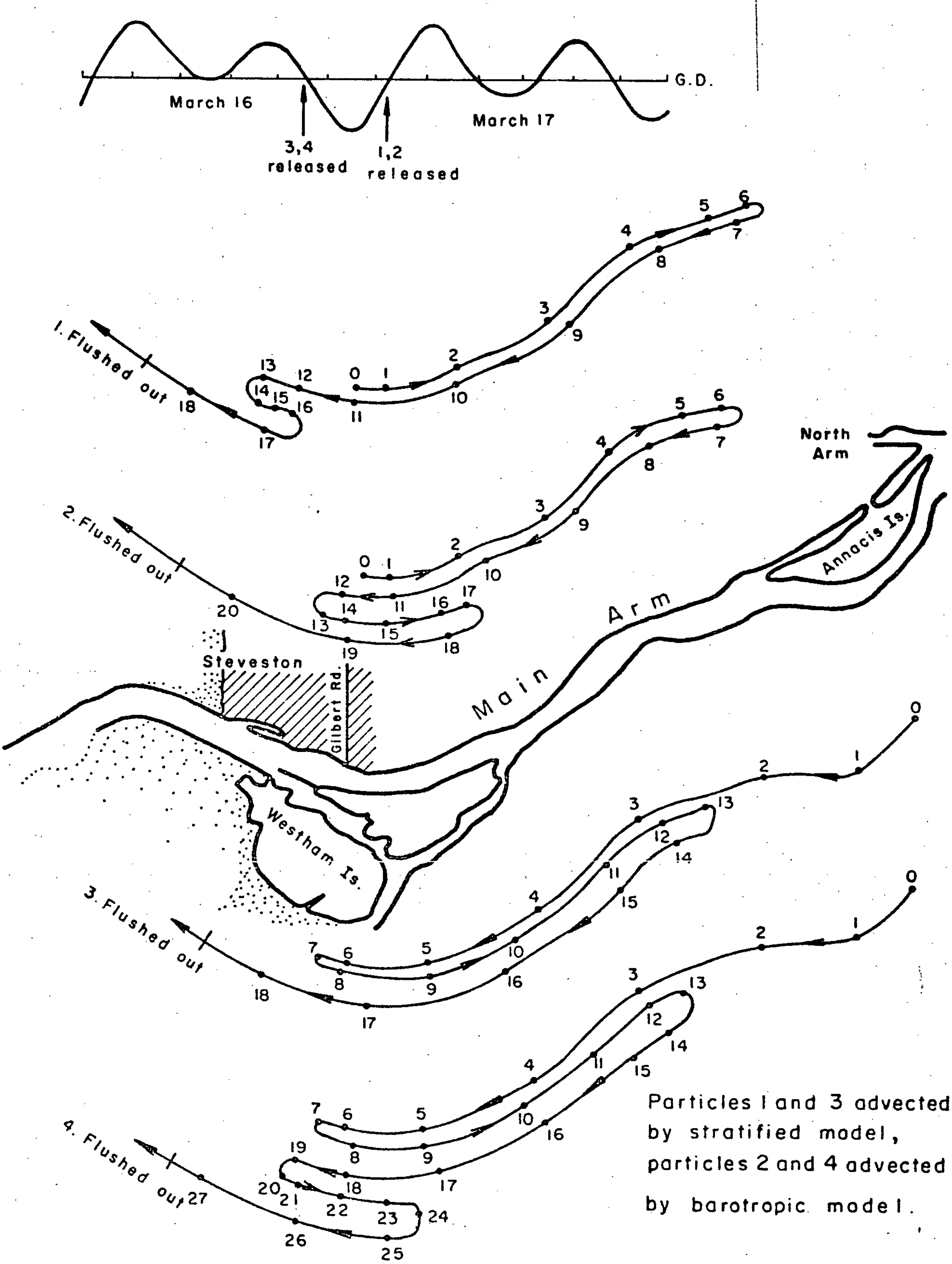


Figure 5.13

dvection Paths of Particles in Stratified Model and Barotropic Model

Consider now the effects of velocity errors on predicted inital effluent dilution. Of critical importance here are those velocity predictions which determine the major effluent spikes, namely predicted values around the slackwater periods. It is crucial that the predicted values be accurate not only in terms of magnitude but as the length of the slackwater period is important, they must also be accurate in timing. Before one can appreciate the significance of near-slack-tide velocity errors it is necessary to understand the method used in the tidally varying mass transport model to calculate initial effluent concentrations. Estimates of initial dilution are obtained by diluting the effluent mass discharged over one hour into the volume which flows by the discharge point during the same period. Although this method is acceptable when average tidal flows are non-zero, it is inappropriate when the net hourly flow approaches zero as in this case initial effluent concentrations become indeterminate and the initial dilution curve becomes discontinuous. To prevent the occurrence of this slackwater discontinuity, the net hourly flow has been constrained so that it can never reach zero. Thus in the event that the zero flow condition occurs, the value of initial instream waste concentration is determined by the arbitrarily chosen minimum flow rate.

This weakness, inherent in the tidally varying model, cannot easily be overcome. Possibly by increasing the temporal resolution of the model so that it would approximate river/estuary conditions using smaller time increments (i.e. in the order of minutes instead of one hour), the effect could be minimized. Although the zero tidal flow condition might still occur, because of the finer time increment, its impact in the simulation would be reduced as each time increment would then represent a smaller fraction of the tidal period and thus receive less weight in defining the tidally varying response.

Another important factor that influences the validity of the tidally varying predictions is the fact that longitudinal dispersion has been ignored. This was done purposefully to exaggerate the intra-tidal river/estuary response, however it may have drastically effected the validity of the predicted results. In particular, since the dispersion process is time dependent, it will exhibit its most severe effects on concentration spikes in effluent parcels which have extended residence times, precisely the same conditions that result in the most significant DO depletions.

In this study, although it has not been possible to fully determine the extent to which this effect might alter the validity of the tidally varying predictions, it is considered that longitudinal dispersion after any extended period (i.e. more than one tidal cycle) will result in a two to five fold reduction in any concentration spike.

5.2.5 <u>Summary</u>. In summary, the results from the tidally varying dissolved oxygen model show an increased DO deficit during portions of the tidal cycle. Although this effect is typical of estuary dissolved oxygen response it may not be an accurate representation of true river/estuary response. Concern has been expressed over the validity of the tidally varying predictions; firstly, because the mechanics of the solution method are sensitive to the conditions causing the increased deficits, namely the velocity predictions around slackwater, and secondly, because the assumption of zero dispersion may also drastically affect the conditions of maximum deficit. Because of the concern that the tidally varying model may not be entirely appropriate for use in describing the intra-tidal behaviour of dissolved oxygen parameters, it will not be used explicitly in the assessment of lower Fraser assimilative capacity. It will be used, however, to exemplify intra-tidal response thereby augmenting and tempering the following assessment of lower Fraser dissolved oxygen dynamics.

5.3 AN ANALYSIS OF LOWER FRASER RIVER/ESTUARY ASSIMILATIVE CAPACITY

The following prefactory comments are offered prior to making a preliminary assessment of lower Fraser River/Estuary assimilative capacity. Their purpose is to inform the reader as to the intent of the ensuing discussion and, at the same time, to offer the rationale behind it.

When it comes time to utilize the capabilities of a study such as this, a decision must be made regarding the mode of attack. Since it is possible through the use of mathematical models to investigate an inveritable infinitude of different input combinations and permutations, the decision must take into account the limits of practicality as well as the objective at hand. The most appropriate use to be made of the predictive capabilities for the purposes of this study is to bring to light some of the main features of lower Fraser oxygen dynamics. To do this the analysis has made use of specifically chosen hypothetical situations, thus retaining an air of generality in its approach. It falls far short of being completely definitive and, therefore, has deliberately refrained from making specific forecasts of future conditions. However, what it does offer is some indication of how the models have been useful in helping in obtaining an improved understanding of the nature of assimilative capacity in the lower Fraser River/Estuary.

The analysis is based mainly on results from the tidally averaged model. Sensitivity analysis has shown this model to be well behaved in its predicted response (see Section 5.1) and therefore it is considered to be more reliable in terms of the validity of its predictions. Some use will be made of tidally varying model results but because serious concerns have been expressed about its validity as applied in this investigation (see Section 5.2), its use will be restricted to exemplifying the expected effect of intra-tidal dissolved oxygen response; thereby qualifying to some extent the overall assessment of assimilative capacity. As the following analysis is based on unverified dissolved oxygen models, all conclusions drawn out of it must be considered to be tentative.

In the investigation of tidally averaged model response indications were that, according to model predictions, the dissolved oxygen dynamics of the lower Fraser River/Estuary were to a large extent governed by two factors: the influence of freshwater inflow and the effect of water temperature. As conditions in the lower Fraser are such that low flows occur during the period January to March when water temperatures are low, and conversely, that high temperatures occur during higher flow periods, it is not possible to easily define a "critical period" during which dissolved oxygen concentrations would be most seriously affected by waste water discharges. In attempts to establish this critical period and also to obtain some indication as to the effect on dissolved oxygen levels of a large waste loading, there being no observable response using actual waste loadings, a series of model runs was made using the tidally average DO model to stimulate conditions for each month of the year. A hypothetical waste load of 1,000,000 pounds of BOD per day (approximately four times the present total BOD load) was discharged at Station 40 in the model river/estuary, the location chosen to be in the upstream reaches so that a greater oxygen response would be observed within the model boundaries. Monthly mean low flows and high temperatures (see Sections 1.2 and 1.3) representing coincident events with ten and fifty year return periods were chosen for use in the analysis as these conditions would represent an extreme dissolved oxygen response in the river/estuary. It is recognized that the simultaneous occurrence of these two events is highly improbable. Assumed values for the dissolved oxygen model coefficients (see Section 4.4) were used.

The results of the simulation are shown in Figures 5.14 and 5.15, the former representing analysis using ten year return period conditions while the latter represents fifty year conditions. These "space-time" plots are a useful means of presenting large quantities of data in summary form. It should be noted that they are not true space-time plots because the simulation, in this instance, has made use of mean monthly conditions. Since these diagrams represent expected deficit concentrations per 1,000,000 pounds of BOD discharged at Station 40, they can also be thought of as "unit response diagrams". For example, if the waste load at Station 40 were to be doubled, the deficit concentrations throughout the river/estuary would be found to double according to the linearity of

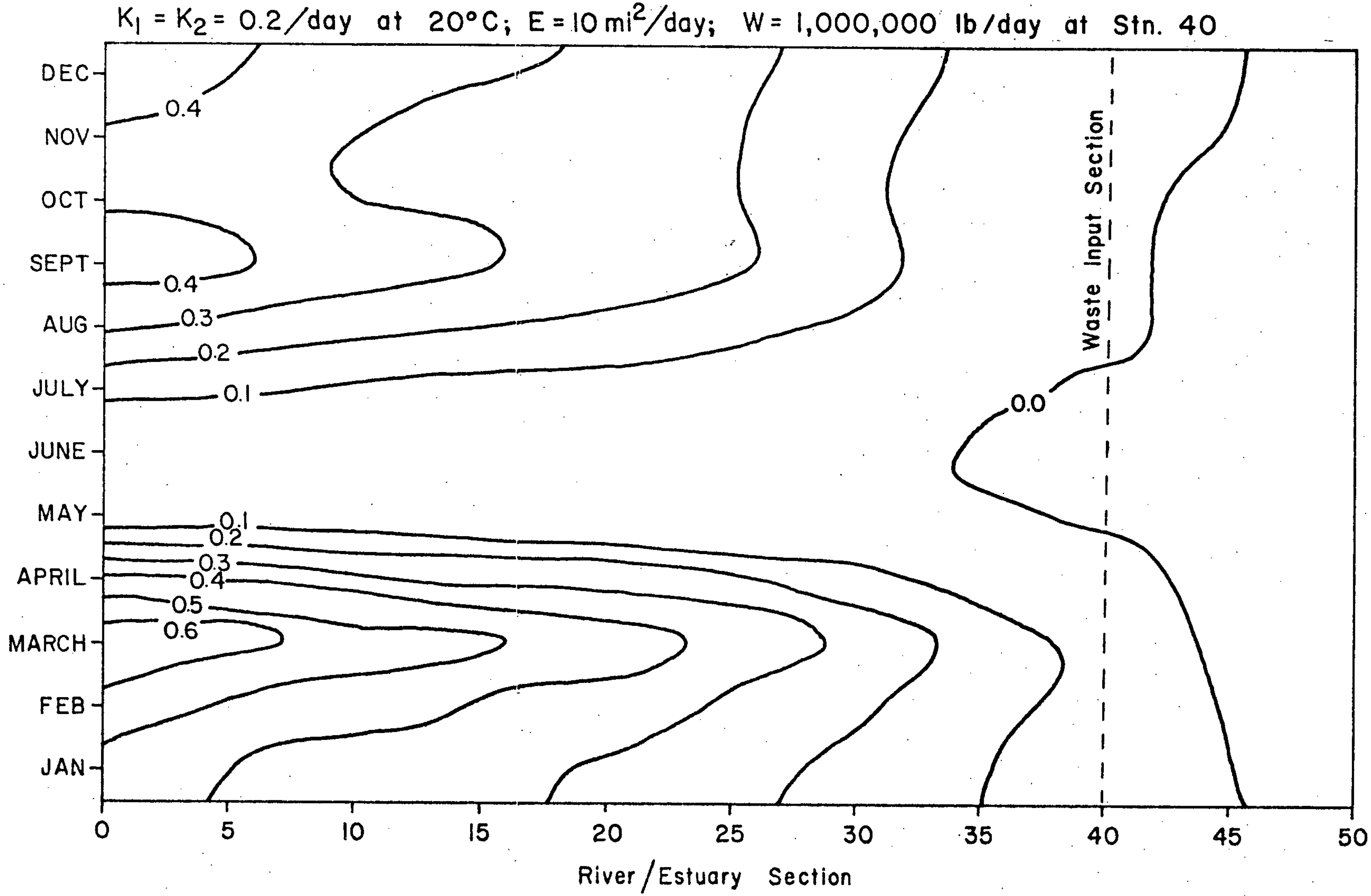


Figure 5.14

Space-Time Plot of DO Deficit Concentrations Using 10 Year Return Period Low Flows and High Temperatures

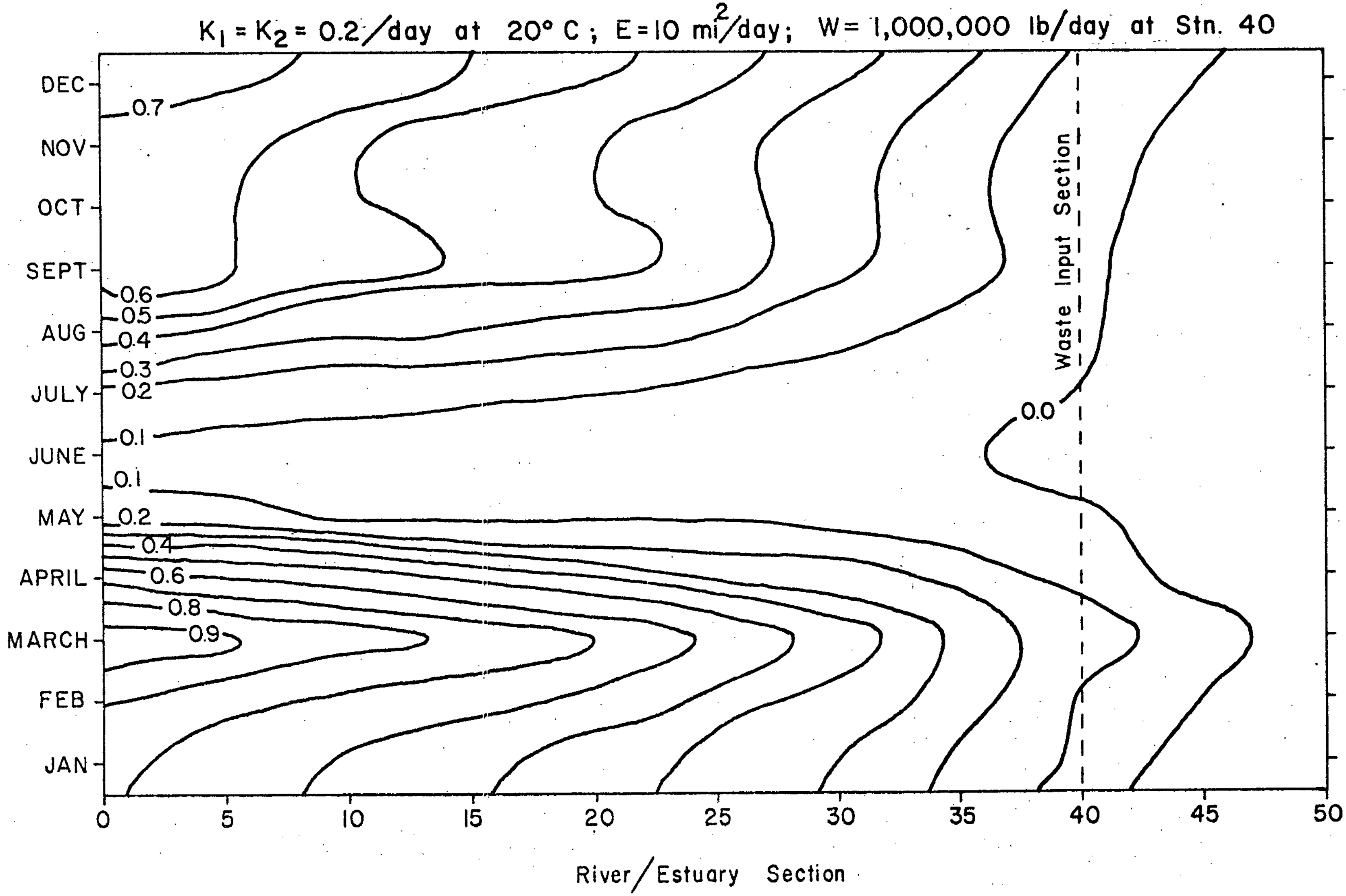


Figure 5.15

Space-Time Plot of DO Deficit Concentrations Using 50 Year Return Period Low Flows and High Temperatures the superposition principle.

From Figures 5.14 and 5.15 it is evident that, according to model predictions, the dissolved oxygen response in the lower Fraser is minimal even when a considerably large waste load is discharged in the upstream reaches. The maximum DO deficit concentrations, 0.6 mg/l and 0.9 mg/l, respectively for the ten and fifty year conditions, are seen in both instances to occur during the low flow period in March. Thus, in terms of oxygen depletions, the effects of low flows outweighed the effects of low water temperatures. Slight DO depletions upstream of the outfall location are observed for all months of the year except during the high flow months - May, June and July.

In order to determine dissolved oxygen concentrations from the results of this simulation, mean monthly DO saturation concentrations are required for the temperature conditions used in each set of analyses. These are shown in Table 5.2.

TABLE 5.2

DO SATURATION CONCENTRATIONS USED IN ANALYSIS

Month	Ten Year Return Period	Fifty Year Return Period
January	13.9	13.8
February	13.8	13.6
March	13.2	13.0
April	12.4	12.2
May	11.4	11.2
June	10.5	10.3
July	9.9	9.7
August	9.7	9.6
September	10.0	9.9
October	11.2	11.0
November	12.5	12.3
December	13.2	12.9

It is evident from a review of Table 5.2 that, even though the maximum DO deficits occur in March, minimum dissolved oxygen concentrations will occur in the period of July to September because of reduced DO saturation concentrations. This indicates that, according to model results, the critical period for dissolved oxygen in the lower Fraser is late summer or early fall, the controlling factor being the influence of water temperature on oxygen saturation levels.

In terms of the magnitude of DO depletions, the results of model analyses show that for the most extreme combinations of low river flows and high water temperatures and an extremely large waste discharge in the upper reaches, the dissolved oxygen response observable in the estuary will be minimal. It is noted that for a distributed load of the same total magnitude, or the similar large load located in the lower river/estuary reaches, the effect on dissolved oxygen response will be even less.

To demonstrate the sizeable ability of the lower Fraser to assimilate organic waste discharges, consider the predicted effects of a ridiculously extreme waste loading. If there were a four-fold increase in the hypothetical discharge located at Station 40 (this represents a single point source discharge with a population equivalent of 20 million), using the fifty year extreme conditions, the predicted, minimum dissolved oxygen concentrations are still above 7.5 mg/l in August and over 9 mg/l in March.

At this juncture a caveat to the foregoing analysis is appropriate. Up to this point, the assessment has been based solely on tidally averaged predictions. Although this should be a good indication of average conditions it does not fully reflect the true nature of river/ estuary dissolved oxygen response. In the analysis of tidally varying dissolved oxygen response (see Section 5.2) it was noted that due to the oscillatory movement of the water mass in estuaries, initial instream effluent concentrations vary throughout the tidal cycle, being characterized by concentration spikes formed during slackwater periods. This tidally varying effluent profile results in a tidally varying dissolved oxygen profile. Indications were that during the low flow period when this effect is most pronounced, in the absence of longitudinal dispersion, peak tidally varying dissolved oxygen deficit concentrations could be up to six to ten times higher than tidally averaged deficits (see Section 2.3). By making the conservative assumption that a two-fold decrease in concentration for extended residence times will account for the effects of dispersion (see Section 5.2.4), the ratio of peak tidally varying to tidally averaged DO deficit would be reduced to the order of three to five. The application of this "expected" intra-tidal DO deficit ratio to temper tidally averaged predictions will give some indication of the significance of tidally varying response. In the case of the hypothetical waste loading of 1,000,000 pounds per day at Station 40 by this calculation there would be a maximum low flow tidally varying deficit of 1.8 mg/l to 3.0 mg/l using the ten year extreme conditions and a 2.7 mg/l to 4.5 mg/l deficit using the fifty year conditions. However, even in the worst instance, minimum DO concentrations would still be above 8.5 mg/l. It should be noted that the effects of intra-tidal DO response will become more significant for large wastewater discharges. For example, consider the extremely large waste discharge (20 million population equivalents) at Station 40: whereas the minimum tidally averaged DO was approximately 9 mg/l using the fifty year conditions, the minimum tidally varying concentration by this analysis will be less than 2.5 mg/l. Thus, for large discharges the tidally averaged model may not be a good indication of river/estuary behaviour. This was one of the major conclusions of Joy's investigation where he used loads of 25 million pounds per day [Joy, 1974].

Although no results using the tidally varying model are available from the critical, late summer period, there is every reason to believe that a similar relationship between the tidally varying and tidally averaged results hold. Thus, one would expect that the effect of waste water discharges on dissolved oxygen would be exacerbated by the influence of tidal action which, although reduced due to higher freshwater flows, would nonetheless be significant.

In summary, according to the results of this analysis, the lower Fraser River/Estuary would seem to have an exceptionally large assimilative capacity. Primarily this is because of the combined influence of large freshwater flows and low water temperatures. The analysis has shown that the critical period for dissolved oxygen is in the late summer in spite of the fact that maximum dissolved oxygen depletions are observed during the low flow period in March. A cursory examination of intra-tidal dissolved oxygen response has indicated that it is an important determinant of lower Fraser dissolved oxygen dynamics and that it will become more important as waste loadings to the river are increased.

CHAPTER 6

SUMMARY AND DISCUSSION

Given the results from the dissolved oxygen models as outlined in the previous chapter it now remains to briefly review and discuss the findings of this study. Attention will be devoted to the implicit as well as explicit details of the research in an effort to glean as much as possible from the results of this attempt at water quality modeling. This chapter will consider separately two aspects of this research: firstly, the development of the dissolved oxygen models including an assessment of their predictive capabilities and, secondly, the results of the application of the models to an assessment of the assimilative capacity of the lower Fraser River/Estuary. This subdivision is useful in that it separates the discussion into two sections, one which deals with the models themselves and the other which deals with the dissolved oxygen resources of the lower Fraser.

6.1 DISSOLVED OXYGEN MODELS

6.1.1 Summary. A review was made of the various oxygen source/sink processes which affect the oxygen balance in waterways. In light of conditions in the lower Fraser River/Estuary it was determined that for the purposes of this study the two basic processes - deoxygenation due to the degradation of discharged organic matter and reoxygenation due to atmospheric reaeration - were the principal factors to be considered in the development of a dissolved oxygen predictive capability. Because of the complex nature of estuary hydraulics which is characterized by unsteady, oscillatory water movement due to the influence of tides, the application of the basic oxygen balance concepts to the modeling of dissolved oxygen in estuaries is many times more difficult than in the analagous river situation. However, in spite of this high degree of hydraulic complexity, mass transport and water movement can be modeled. In this study two different solution methods were utilized: a tidally averaged approach and a tidally varying approach.

The tidally averaged approach by making use of steady-state assumptions simplifies the problem of eliminating the time variable. In essence, the unsteady, estuary flow field is replaced by a steady freshwater flow field and a tidal dispersion component which can indirectly account for the effects of tidal action. All parameters and variables by this solution approach are assigned their mean tidal values.

The second approach to modeling mass transport, the tidally varying approach, does not eliminate the time variable. Thus it attempts to describe "real time" estuary hydraulic behaviour and can account directly for such tidal effects as current reversal.

The incorporation of the basic dissolved oxygen balance formulations into each of the two mass transport models has formed the basis of the two dissolved oxygen models used in this study: the tidally averaged dissolved oxygen model and the tidally varying dissolved oxygen model. To some extent the two models are complementary in their view of river/estuary conditions. The former model allows an analysis to be made of average conditions over a number of tidal cycles whereas the latter by providing a greater degree of temporal resolution allows an analysis to be made of intra-tidal river/estuary behaviour.

6.1.2 Limitations of the Predictive Capabilities. The limitations of the dissolved oxygen models can be classified into three categories: spatial, temporal and calibrational. Of these only the first applies in a similar degree to both the tidally averaged and tidally varying models. This limitation arises from the assumption that all parameters and variables can be approximated by their cross-sectionally averaged values. Choosing this one-dimensional assumption which greatly simplified the solutions of the mass transport equations has resulted in the models being cognizant only of variation along the length of the river/estuary. The models are unable to deal with variability over the river/estuary cross-sections and thus are restricted in application to only the main core flow of the main river channels. Thus the models cannot be applied to the analysis of localized problems such as might occur in the immediate vicinity of an outfall or in the small side channels and slough areas adjacent to the main river. Also by the one-dimensional assumption waste inputs to the models are "completely mixed" either over the cross-section as is the case in the tidally varying model or within a 5,000 foot segment as is the case in the tidally averaged model. This "instant mixing" corollary to the main assumption has as its limitation the fact that mixing is neither instantaneous nor necessarily complete. It is estimated [P. Ward, unpublished data] that at least two tidal cycles are required in the lower river/estuary reaches (with considerably more time being required in the upper reaches) for mixing to be completed. Stratification effects due to the presence of the saltwater wedge may at times inhibit vertical mixing, preventing complete mixing in the vertical plane. Another stratification effect is the sometimes significant increase in freshwater flow velocities which result from the freshwater flowing out over top of the saltwater layer. The presence of the saltwater wedge is not taken into account in either of the models used in this study.

The second class of model limitations are those involving the degree to which the models are a temporal abstraction of the real river/estuary situation. The tidally averaged model is a severe abstraction in the sense that it considers only "steady-state" conditions, assigning all parameters and variables their tidally averaged values. Although to some extent this, in effect, represents an integration over a number of tidal cycles, the averaged conditions have no "real time" meaning. The tidally varying model represents a lesser temporal abstraction as it attempts to simulate "real time" conditions by assigning all variables their average hourly values. Although this method of simulation is the tidally varying model's strong point, it is not without its inherent weakness. The chief limitation is the manner of calculation used to estimate initial effluent dilution rates. During the period around slack-water, the method used is inappropriate because it results in a discontinuous initial dilution curve; the initial effluent dilution becoming zero as the tidally varying velocity approaches zero. Although constraints within the model prohibit the predicted results from ever reaching this extreme condition, the model predictions are nonetheless extremely sensitive to the magnitude as well as the timing of occurence of slackwater velocities. As the slackwater period results in maximum BOD concentrations which in turn ultimately determine maximum DO depletions it must be stressed that this limitation within the tidally varying model may drastically alter the validity of predicted results.

Finally, there is the overriding limitation that the dissolved oxygen models cannot presently be calibrated. Because of the lack of any significant dissolved oxygen depletions in the lower Fraser River/Estuary, traditional calibration procedures were of no use. Consequently the dissolved oxygen response coefficients used in this study were selected solely from empirical relationships described in the literature. Aside from the fact that the dissolved oxygen coefficients may not be appropriate, a number of other limitations of a calibrational nature exist. With regard to the tidally averaged model, because of the peculiar nature of salinity variation in the lower Fraser, it was not possible to estimate the value of the tidally averaged dispersion coefficient (E). As such, values used were chosen on a purely arbitrary basis.

Considering the calibrational limitations of the tidally varying model, it has not been possible to verify hydrodynamic sub-model velocity predictions. Hence the accuracy of these predictions, in terms of both magnitude and timing, is not known. This is an important limitation since velocity predictions serve as the basis for estimating tidally varying mass transport. As well, longitudinal dispersion was neglected in analyses using the tidally varying model. This can be considered as a calibrational limitation since dispersion can be accounted for through use of a non-zero longitudinal dispersion coefficient.

6.1.3 The Modeling Experience. To this point in the discussion it may seem that the limitations of the models have been over-emphasized. This has been done purposefully. The author feels strongly that the limitations and weaknesses inherent in the models must be realized as a necessary perliminary to obtaining a true appreciation of the models' capabilities. Only by learning the limitations can one appreciate the strength of the modeling exercise; its overall strength being determined by the strength of the weakest assumption. All too often modeling studies overstress the strengths and capabilities of the exercise while glossing over its weaknesses which when uncovered point out serious shortcomings in methodology and/or interpretation. Seldom is any attempt made from within the study or from the outside to critically assess the net result and overall utility of the modeling exercise. Because of the author's concern over the importance of this often neglected aspect of the modeling experience, the following is included in this discussion.

In recent years the idea of using mathematical models as an aid in the planning and management of river systems has attained universal acceptance. At the outset of most planning/management studies the question is no longer "Shall we use a model?" but rather "Which model(s) shall we use?" Encouraged by those who seek an easier means of dealing with the increasing complexities of water management and spurred on by rapid developments in digital computer technology, vast numbers of models of varying disciplines have proliferated technical literature. It is not unfair to say that the majority exist in a highly theoretical state, untested in application and unproved by use. Of those studies which have applied water quality modeling concepts to the development of a usable predictive capability, very few have been subjected to intensive critical reviews. A recently published case study entitled The Uncertain Search for Environmental Quality [Ackerman et al., 1974] has made such a review of the Delaware Estuary Comprehensive Study (DECS), critically analyzing a number of technical aspects and criticizing the water pollution policy decisions made by the Delaware River Basin Commission as a result of the DECS findings. In particular, the water quality modeling studies carried out by DECS using Thomann's steady-state BOD-DO model (similar to the tidally averaged model used in this investigation) came under heavy attack. Shortcomings were pointed out in the DECS model study in its failure to deal adequately, if at all, with uncertainties regarding model response coefficients, stormwater and tributary loadings and benthic oxygen demand. Ackerman concludes that as a result of these inadequacies the DECS study gave misleading, perhaps faulty, information on the benefits which would be derived from a "clean-up" program on the Delaware. He goes on to say that in spite of the acclaim given DECS, lauding the sophisticated effort which employed innovative conceptional and institutional tools, it was in the end "unequal to the task" and led to "a failure in modern policy making".

The main reason for this, according to Ackerman, is that in their enthusiasm the DECS staff had failed to impart a rational sense of perspective along with their findings. In presenting their achievements "the research staff emphasized the accuracy of the numbers its model generated", offering an analysis that was "conspicuously devoid of cautions (to the decision makers) about the limitations of its predictions". Thus although the scientific merits of the study were undeniable, its utility as an aid to decision making was questionable, specifically because of misconceived perceptions regarding the accuracy of the water quality model.

In this investigation, because of fortuitous circumstances which led to employment with an interdisciplinary research team, the author has been forced throughout to view the results of this modeling exercise in light of their meaning and adequacy as aids to an understanding which could be communicated to persons from different disciplines. This has been fortunate in that it has resulted in the development of a more balanced sense of perspective in this study than would have otherwise occurred. Hopefully this sense of perspective has been effectively communicated and has resulted in a strengthening of this investigation.

6.2 FRASER RIVER/ESTUARY DISSOLVED OXYGEN RESOURCES

6.2.1 Summary: An Improved Knowledge Base. It can be stated almost without exception that dissolved oxygen levels in the lower Fraser River/Estuary are high, generally being 90 to 100 percent of saturation values. In combination with the characteristically low water temperatures, this results in dissolved oxygen concentrations of 8 to 12 mg/l throughout the year; more than enough to support even the most sensitive aquatic organisms.

It is not difficult to ascertain why the dissolved oxygen resources of the lower Fraser are presently in their healthy state. Primarily it is because, relative to its size and dilutional ability, the lower Fraser receives very little in the way of discharged organic wastes. This can best be illustrated by way of example. Consider the Delaware Estuary in the eastern United States where there has been serious oxygen depletion problems. Average freshwater inflows there are approximately 11,000 cfs while the average estimated organic waste discharge is in excess of 1,300,000 pounds of BOD per day [DECS, 1968]. Compare that to the lower Fraser where average freshwater flows are approximately 98,000 cfs while the estimated BOD load is presently in the order of 250,000 pounds per day; a situation where river flows are on an average approximately nine times higher and total waste loadings are lower by a factor of five. In addition to the considerable difference in the relative magnitude of waste loadings, the situation in the lower Fraser has the added advantage that the bulk of the total BOD loading is discharged from the Greater Vancouver area located on the seaward reaches of the river/ estuary. In the Delaware a large portion of the total load (at least 45 percent) is discharged in the vicinity of Philadelphia near the head end of the estuary, whereas the major portion of the total BOD load (over 80 percent) enters the lower Fraser downstream of the Port Mann Bridge within 20 miles of the Strait of Georgia.

A number of other factors play an important role in the dissolved oxygen dynamics of the lower Fraser. Assessment made through use of the dissolved oxygen models has shown that two in particular are important in defining the nature of the lower Fraser's rather extensive assimilative capacity. These are the characteristically low water temperatures and large freshwater flows. With regard to the large river flows, not only do they afford considerable dilution to effluent discharges, but they also minimize residence times of effluent parcels in the river/estuary. In spite of the retarding influence of tidal action, a major portion of the oxygen demand even for waste discharges in the upper reaches of the river is flushed out of the channelized sections of the river/estuary to be exerted in the Strait of Georgia. The low water temperatures, in addition to providing high dissolved oxygen saturation concentrations, also reduce the effective rates of deoxygenation which again results in a smaller portion of the total oxygen demand being satisfied within the river/estuary. In terms of seasonal variation a complementary pattern exists in the lower Fraser typical of north latitudes where low flows occur in the winter months when water temperatures are also low. Thus during the period when tidal effects are most pronounced and residence times are maximum, deoxygenation rates are low and saturation levels high. In the summer months the situation is reversed with high flow counteracting the effect of higher water temperatures on saturation concentrations and deoxygenation rates. Analysis made using the oxygen models to simulate conditions for all months of the year has shown that the critical period for dissolved oxygen in the lower Fraser is in late summer. Although maximum DO depletions would theoretically occur during the winter months, the predominating influence of water temperature effects on saturation concentration results in the lowest DO levels occurring in August and September.

Another aspect of lower Fraser dissolved oxygen dynamics brought to light in this investigation is the fact that tidal action can cause a tidally varying dissolved oxygen response. This situation arises as a result of the oscillatory water movement in the river/estuary which produces a tidally varying initial efffluent concentration profile, characterized by effluent spikes formed during periods of slackwater. Preliminary indications are that these concentration peaks could be six to ten times higher than the tidally averaged effluent concentrations. Although it has not been possible to determine the exact nature of intra-tidal dissolved oxygen response, cursory investigation has shown that peak tidally varying DO deficits could be three to five times greater than tidally averaged DO deficits.

In summary, the dissolved oxygen models which were developed in this study have shown to be useful in helping to define the nature of dissolved oxygen dynamics in the lower Fraser River/Estuary. Their chief utility has been as an aid to improving the understanding and furthering the extent of knowledge of the lower Fraser dissolved oxygen resources.

6.2.2 Future Conditions. Since the dissolved oxygen models could not be calibrated, a great deal of uncertainty exists regarding the accuracies of their predictions. In order to deal with this uncertainty and to alleviate some of the doubt (the author's included) surrounding the validity of predictions, this study has made use only of very cautious analyses choosing the most conservative estimates. As well, this study has deliberately refrained from the usual practice of making assessments of future conditions which are based on specific sets of possible future waste loading patterns. Instead, emphasis was placed on improving the extent and depth of the knowledge base. Assessment of future conditions has been made indirectly by outlining in an illustrative manner the considerable magnitude of the assimilative capacity of the lower Fraser. Stated simply, this investigation has shown that in the conceivable future there will not be any significant deterioration of water quality in the main stem, lower Fraser River/Estuary, at least insofar as concerns average levels of dissolved oxygen. The nature of dissolved oxygen dynamics in the lower Fraser, coupled with present conservative pollution control policy, should be sufficient to guarantee that dissolved oxygen concentrations are maintained at their present high levels. Pollution control requirements as they affect dissolved oxygen in the lower Fraser may, in fact, be more stringent than they need be. However this is indeed a small price to pay for the adequate protection and maintenance of a very valuable natural resource.

6.2.3 Uncertainties. Hopefully the results of this research investigation has cleared up a number of uncertainties that prevailed at the time of its conception. However, upon its completion there remain a number of areas of concern which have either not been covered completely or adequately by this investigation, or have arisen out of it. Some of these will now be briefly discussed.

The main area of uncertainty involves the fact that the dissolved oxygen models used in this investigation were not calibrated. Hence, all predictions must be viewed with caution and all conclusions must be regarded as being tentative. With respect to the tidally averaged model, its mechanics appear to be sound as evidenced by its manner of response during the sensitivity analysis. Uncertainties with this model lie in the values of the coefficients used. These uncertainties could be alleviated by a more exact determination of model coefficients, theoretically possible but practically, very difficult to accomplish.

With the tidally varying model, on the other hand, it was not possible to fully investigate the sensitivities of its response and, more importantly, there was concern expressed that some of the mechanics of solution method may not be entirely appropriate. It was beyond the scope of this investigation to become involved with modifications to this very sophisticated model and, as a result, it was probably not utilized to its full potential. Consequently, a great deal of uncertainty exists regarding the tidally varying aspects of lower Fraser dissolved oxygen dynamics.

Some of this uncertainty could be cleared up by a future study, more specific than this one, that would obtain better estimates of some parameters, for example, longitudinal dispersion, or would test the model response using field investigations. Then it would be possible to more fully assess the significance of intra-tidal dissolved oxygen response as well as the effects of short term situations such as stormwater discharges or combined sewer overflows which may, sooner or later, become important.

One other major area of concern should be mentioned and that is the water quality in bays, sloughs and other backwater areas in the lower Fraser. This question has not been addressed in this research as the models were restricted in their application to the main core flow.

However, there is evidence to indicate that, in some cases, situations involving significant oxygen depletions already exist [Westwater, unpublished data]. What will be the effects on these areas, some of which are very sensitive, of large discharges that may have no significant effect on main stem dissolved oxygen levels?

In summary, a considerable uncertainty exists, upon conclusion of this investigation, involving a number of aspects of the dissolved oxygen resources of the lower Fraser River/Estuary. Some of these have been briefly discussed and deserve future study.

CHAPTER 7

CONCLUSIONS

The principal conclusions which emerge from this investigation are listed summarily, as follows. Firstly, with regard to the dissolved oxygen models, it can be concluded that:

(1) Verification of the models is not presently possible because of the absence of any significant dissolved oxygen depletions which has precluded model calibration;

(2) In an analysis of the sensitivities of the tidally averaged model its manner of response was found to be reasonable and "well behaved" over the expected range of input parameter variation. Therefore, it is considered to be reliable in its predictions, inspite of the fact that it could not be verified;

(3) There are reservations about the validity of the tidally varying model in this application as its sensitivities could not be fully tested and, in a number of respects, it is considered to be inappropriate;

(4) The tidally averaged model offers a number of distinct advantages over the tidally varying model in terms of the simplicity of its approach, the straightforwardness of its development and the flexibility of its application;

(5) The utility of the highly sophisticated tidally varying model is severely restricted because,as a result of its complex, multi-model solution format, it is operationally both cumbersome and costly;

(6) O'Connor's steady-state, continuous solution model could not be applied in this investigation due to computing problems which resulted from the particular form of cross-sectional geometry variation that characterizes the lower Fraser River/Estuary;

(7) The limitations of model studies must be stressed along with their capabilities so that the modeling experience is effectively communicated in the light of a proper perspective.

Secondly, as applies to the lower Fraser River/Estuary, through use of the dissolved oxygen models, it can be concluded that:

(1) An improved general understanding of the nature of dissolved oxygen dynamics in the lower Fraser River/ Estuary has been achieved;

(2) An appreciation of the considerable magnitude of the lower Fraser's assimilative capacity has been gained, although its accurate quantification is not presently possible because the dissolved oxygen models cannot be verified;

(3) The "critical period" for dissolved oxygen in the lower Fraser is likely to be in late summer;

(4) Tidally averaged dissolved oxygen levels in the main channels of the lower Fraser River/Estuary will not be seriously impaired in the conceivable future. In fact, existing pollution control requirements, as they pertain to dissolved oxygen, may be more stringent than needs be;

(5) As BOD loadings to the lower Fraser increase, there will be brief periods within the tidal cycle, particularly during the low flow periods, when dissolved oxygen concentrations will be reduced below tidally averaged levels. However, in this investigation, the significance of this effect could not be fully assessed.

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