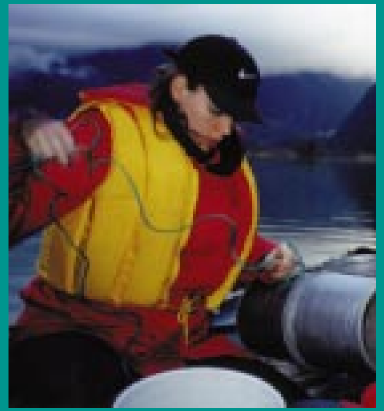

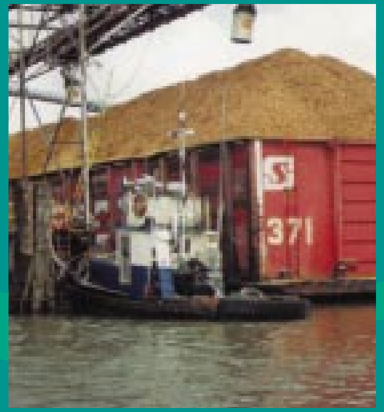

Health of the Fraser River Aquatic Ecosystem Vol. I

Edited by Colin Gray and Taina Tuominen

DOE FRAP 1998-11

Environment Canada
Environnement Canada

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Canada

THE FRASER RIVER BASIN

PHYSICAL

Largest river in B.C. (on length, basin size and flow basis)

3rd greatest mean annual flow in Canada

10th longest river in Canada

5th largest drainage basin in Canada

Basin size: 234,000 km^2 or 1/4 of the area of BC

River length: 1,375 km

Mean annual discharge at Port Mann: 3600 m^3/s or 113x10^9 m^3/yr (113 km^3/yr)

Fraser River at Hope

10-yr Running Mean

Maximum daily discharge recorded at Hope: 15,200 m^3/s (1948); higher discharge in 1894—however there are no discharge records for that period

Minimum daily discharge recorded at Hope: 340 m^3/s (1916)

21% of the total discharge at Port Mann is added by tributaries downstream of Agassiz (e.g., Harrison, Chehalis, Chilliwack, Pitt, Stave and Coquitlam rivers).

Suspended sediment dominated by silt to sand-sized particles

Mean suspended sediment concentration at Mission: 165 mg/L—lower than other large rivers in Western Canada, e.g.:

- Mackenzie (327 mg /L)
- Liard (566 mg/L)
- Peace (753 mg/L)

Range of mean annual precipitation: 270 mm at Kamloops to 2,460 mm/year measured at Alouette Lake in the Lower Fraser Valley.

Range of river water temperatures:

- 0 °C (December and January) to 22 °C (August) at Mission
- 0 °C (November to March) to 19 °C (August) at Marguerite

Range of mean daily high air temperatures:

- January: -7.2 °C at Fort St James to 5.7 °C at Vancouver
- July: 21.7 °C at Vancouver to 28.3 °C at Kamloops

Headwater streams originate in three mountain ranges and a plateau:

- Rocky Mountains
- Columbia Mountains
- Coastal Mountains
- Interior Plateau

Stream order of the Fraser River is 8 at mouth (based on a 1:250,000 scale map)

The bedrock geology can be split roughly into five types:

- igneous intrusive rocks
- volcanic & sedimentary
- foliated metamorphic
- folded sedimentary
- flat lava with some sedimentary

Basin contains 11 biogeoclimatic zones of B.C., which include alpine, interior forest, grasslands and coastal forest.

Basin occurs in 2 Canadian ecozones: montane cordillera and Pacific maritime.

To date, 210 aquifers (of 296 in B.C.) have been delineated in the basin

- sizes range from 0.3 km^2 to >1660 km^2
- 90% are used for drinking water

BIOLOGICAL

Total fish species that use the river: 59

Native freshwater fish species in basin: 40

Basin has the world's most productive salmon river system, supporting 5 species of salmon.

Total species of birds that use the basin: over 300

Species of waterfowl that breed in the basin: estimated at 21

Water birds from 3 continents rely on the estuary as part of the Pacific Flyway

Number of vascular plant species in the lower Fraser basin: 1446

Vascular plant species that are introduced aliens to the lower Fraser Basin: 40%

Native plant species that are rare in the lower Fraser basin: nearly 25%

POPULATION

Basin's population (1997): 2.5 million

By 2021, the basin's population is expected to increase by 50%

2/3 of the population of BC lives in the basin (54% in the lower Fraser area)

Greater Vancouver Regional District (GVRD) population (1996): 1.8 million

Population of the GVRD is expected to reach 2.7 million in 20 years

First Nations in the basin: 8 linguistic groups in 96 communities

LAND USE

Urban area (1996): 5,100 km^2

Agricultural area (1996): 1,510 km^2

Other land area (forest, park, etc.) (1996): 227,500 km^2

About 50% of BC's arable land is in the basin.

Approximately 20% of the farmland is irrigated using water from the Fraser or its tributaries

Contains almost 50% of BC's sustainable yield of timber

Contains 90% of BC's gravel extraction operations, mostly in the Lower Fraser Valley

Unlogged watersheds >50 km^2 from Prince George to Vancouver:

- Stein River Valley
- Siwhe Creek (near the Stein)

Contains 60% of BC's metal mines

Contains 8 pulp and paper mills

Contains 25 major (9m or greater in height) dams; all on tributaries

As of 1991, approximately 6% of the land area in the basin was provincial parkland

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HEALTH OF THE FRASER RIVER AQUATIC ECOSYSTEM

A Synthesis of Research Conducted under the Fraser River Action Plan

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1999

Volume I
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Fraser River Action Plan

Environment Canada / Environnement Canada

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VOLUME II

Executive Summary

The purpose of the Fraser River Action Plan (FRAP) was to restore the environmental health and promote long term sustainability of the Fraser River watershed, in cooperation with all governments and stakeholders. This federal government initiative was undertaken by the Departments of Environment and Fisheries and Oceans from 1991 to 1998. A major thrust of FRAP was a program of research on the extent and severity of contamination in the Fraser Basin's aquatic ecosystem. The program was led by Environment Canada and conducted by scientists from universities, government and the business sector.

New methods and indicators, as well as existing tools, were used to assess the extent of basin contamination and its effect on the health of the aquatic system. Concentrations of contaminants in water, suspended and bed sediment and biological tissues (fish, birds and mammals) were used as one set of indicators of relative contamination. A smaller set of biological response indicators, including the structure of the benthic macroinvertebrate community, fish condition and health, detoxifying enzyme levels, and the reproductive success of selected wildlife species, was also used. The synthesis of information provided by these indicators records the status of the basin's aquatic-based ecosystem between 1992 and 1997.

The research program was focused on the main stem Fraser River from the Rocky Mountains to the estuary, and two major tributaries, the Thompson and Nechako rivers. The effects of effluents from point sources, such as pulp and paper mills, were evaluated in these rivers below Prince George, Quesnel and Kamloops. Non-point sources of contaminants and their effects were assessed in the Brunette River, an urbanized basin in the Greater Vancouver region, and the Sumas River, an agricultural basin in the Lower Fraser Valley. Headwater lakes were investigated for metals and persistent organic pollutants. Lastly, we examined the recovery of the delta foreshore ecosystem from exposure to effluent from a large municipal wastewater treatment plant (WWTP).

Our assessment revealed that smaller tributaries, arising in basins affected by urban or intense agricultural development, are showing many obvious signs of contaminant stress. These signs of stress from urban and agricultural runoff, or non-point source pollution, include: elevated levels of contaminants in water and sediment; the failure of eggs for some amphibian species to hatch; the poisoning of raptors by certain pesticides; and significant changes in the structure of the benthic macroinvertebrate community.

Contamination is also evident in the Fraser estuary and at some sites in the Thompson River where the levels of chemicals such as polycyclic aromatic hydrocarbons (PAHs) and some dioxin and furan congeners in sediment exceed guidelines or draft guidelines established for the protection of aquatic life. The effects of antisapstain chemicals accumulating in estuary sediments have not been fully evaluated. The source of these chemicals is stormwater runoff near sawmills. The recovery of Sturgeon Bank from past exposure to municipal WWTP effluent is evident from the return of benthic communities and decline in sediment metals after the large outfall at Iona Island was extended beyond the intertidal area. Two newly installed secondary WWTPs in the Vancouver area should further reduce the discharge of contaminants to the estuary. However, without preventative action, it is expected that non-point source pollution from urban and agricultural

development and effluent volume from WWTPs will continue to increase with the projected population growth for the Lower Mainland.

The main stem river upstream of the estuary and its major tributaries do not exhibit significant concentrations of contaminants at most locations. This is largely due to production changes in pulp mills in the early 1990s, which resulted in the significant reduction of dioxins and furans in the mills' effluents, and recent improvements in municipal WWTPs. Large reductions in the use of some chemicals, such as polychlorinated biphenyls (PCBs), lead, pentachlorophenol and some pesticides over the past two decades are also responsible for the low levels of contaminants observed. Toxicity data were developed for some chlorophenolic and PAH contaminants that are present at elevated levels downstream of pulp mills. The results indicate that the tested chemicals are not likely to cause effects on biota at the concentrations observed in the river outside of mixing zones. Although the main rivers generally have low levels of contaminants when compared to existing guidelines for the protection of water uses and aquatic biota, municipal WWTPs and pulp mills are still sources of many classes of chemicals.

Biota near these pollution sources show responses to these low levels of contaminant exposure. For example, analyses of fish and bird tissue show that there is greater induction of the detoxifying enzymes, mixed function oxygenases, in samples collected downstream of urban centres and pulp mill discharges than at other locations. While the two fish species surveyed have a relatively high incidence of abnormalities throughout the basin, highly pigmented livers and kidneys are commonly observed only at sites downstream of pulp mills on the Fraser River. Fish at these sites also have reduced gonad development, as do the fish downstream of Kamloops in the Thompson River. The significance of these effects on fish health are not known.

Artificial stream experiments conducted at Prince George and Kamloops demonstrate that pulp mill effluent, at dilution levels experienced in the river at low flow, can stimulate growth of benthic organisms. However, the experiments indicate that should effluent concentrations in the river increase signs of toxic effects will begin to appear.

Persistent organic pollutants were found in the river's headwaters. The likely source of PCBs, DDT and its metabolites, and toxaphene found in fish from these lakes, especially lakes at higher elevations, is long range atmospheric transport and deposition coupled with the release of historic deposits of contaminants from melting glaciers and permanent snowfields. In a warmer global climate the release of contaminants from glacial "storage" will continue.

These results have led to the following recommendations to help sustain the ecosystem in the Fraser River Basin.

Runoff from agricultural and urban areas is now a significant source of contaminants in the basin. Strategies for controlling contaminated runoff from urban and agricultural areas should be a priority in future urban planning and agricultural development programs. Such strategies should include: (1) minimizing impervious area; (2) maintaining riparian buffer zones along streams; (3) incorporating sedimentation basins along streams and in storm sewers; (4) promoting and enforcing best management practices for livestock densities, septic tank servicing, and fertilizer (such as manure) application; (5) and promoting public environmental stewardship.

To monitor progress in restoring and preserving the health of the Fraser Basin, the condition of the basin's aquatic ecosystem should be periodically reassessed. FRAP has provided a baseline of the basin's condition. The future use of the indicators and methods employed in FRAP will enable comparison to this baseline. Opportunities for integrating FRAP methods into programs, like the Environmental Effects Monitoring and Canada-BC Water Quality Monitoring programs, are being explored. For example, environmental effects monitoring could include, under certain circumstances, the benthic macroinvertebrate reference condition approach, in combination with controlled mesocosm experiments, to assess the effects of effluents downstream of discharges. Incorporation of more biological and biochemical components into routine measurement programs will enhance the ability to maintain the health of the basin's ecosystem.

Many knowledge gaps were identified by the participating scientists and during the synthesis of the program results. Two general knowledge gaps and some basin-specific research needs are highlighted here. The first general gap is the lack of understanding of the cumulative impacts of chronic exposure to mixtures of many chemicals, all at low levels. The second is determining the amount of site-specific toxicity information required to complement guidelines developed with standard bioassays, when setting site-specific water quality objectives. Environmental management agencies in the basin should periodically review and apply advances in understanding resulting from future research on these questions.

Several basin-specific research needs were identified in the program. Some of these are: determining the presence and effects of endocrine disrupting chemicals; identifying the causes of abnormalities found in basin fish; assessing the impacts of the antisapstain chemicals on the estuary; and evaluating the contamination of food chains in headwater lakes by atmospheric pollutants. Some of this research has been included in the Georgia Basin Ecosystem Initiative.

Sommaire

Le Plan d'action du fleuve Fraser (PAFF) avait pour objectif de restaurer l'environnement du bassin hydrographique du fleuve Fraser et d'en favoriser la durabilité à long terme, en collaboration avec tous les gouvernements et partenaires. De 1991 à 1998, cette initiative du gouvernement fédéral a été menée par les ministères de l'Environnement et des Pêches et des Océans. Le plan d'action consistait principalement en un programme de recherche sur la gravité et l'étendue de la contamination de l'écosystème aquatique du bassin du Fraser. Ce programme était dirigé par Environnement Canada et mené par des scientifiques des universités, du gouvernement et du monde des affaires.

Des méthodes et indicateurs nouveaux ainsi que des instruments existants ont servi à évaluer l'étendue de la contamination du bassin et son impact sur le milieu aquatique. Les concentrations de polluants dans l'eau, les sédiments benthiques et en suspension et les tissus biologiques (poissons, oiseaux et mammifères) ont été utilisés comme indicateurs de la contamination relative. On a également eu recours à un ensemble plus restreint d'indicateurs de la réponse biologique, notamment la structure de la communauté des macroinvertébrés benthiques, l'état et la santé des poissons, la concentration d'enzymes de détoxification et le succès de reproduction de certaines espèces sauvages. La synthèse de l'information fournie par ces indicateurs permet de suivre l'état de l'écosystème aquatique du bassin entre 1992 et 1997.

Le programme de recherche était axé sur le cours principal du fleuve Fraser, des montagnes Rocheuses à son estuaire, et sur deux de ses principaux affluents, les rivières Thompson et Nechako. Les effets des effluents de sources ponctuelles, comme les usines de pâte et papiers, ont été évalués dans ces rivières en aval de Prince George, Quesnel et Kamloops. Les sources diffuses de contaminants et leurs incidences ont été évaluées dans la rivière Brunette, un bassin urbanisé de la région métropolitaine de Vancouver, et dans la rivière Sumas, un bassin agricole de la vallée inférieure du Fraser. On a effectué des recherches dans des lacs d'amont afin de déceler la présence de dépôts de métaux et de polluants organiques persistants. Enfin, nous avons examiné le rétablissement de l'écosystème riverain du delta, qui a été exposé aux effluents d'une grande station de traitement des eaux usées urbaines.

Notre évaluation a révélé que les affluents plus petits, issus de bassins urbanisés ou soumis à une agriculture intensive, montrent de nombreux signes évidents de perturbation occasionnée par des polluants. Ces signes de perturbation exercée par l'écoulement des eaux de ruissellement agricole et urbain ou par la pollution de source diffuse se manifestent notamment par des concentrations élevées de polluants dans l'eau et les sédiments, par la non-éclosion des oeufs de certaines espèces d'amphibiens, par l'empoisonnement de rapaces par des pesticides ainsi que par des modifications majeures de la structure de la communauté des macroinvertébrés benthiques.

La contamination se manifeste également dans l'estuaire du Fraser et certains endroits de la rivière Thompson où les concentrations de substances chimiques dans les sédiments, comme les hydrocarbures aromatiques polycycliques (HAP) et certains congénères des dioxines et des furanes dépassent les lignes directrices établies ou provisoires visant la protection de la vie aquatique. Les incidences de l'accumulation des substances chimiques anti-taches dans les sédiments n'ont pas encore été entièrement évaluées. Ces substances chimiques sont transportées par les eaux pluviales dans le voisinage des scieries. Le rétablissement du banc Sturgeon, qui a été exposé aux effluents des stations de traitement des eaux usées urbaines, est évident quand on constate le retour des communautés benthiques et la baisse des concentrations de métaux dans les sédiments, après le prolongement du grand émissaire d'évacuation de l'île Iona au-delà de la zone intertidale. Les deux nouvelles usines de traitement secondaire installées dans la région de Vancouver réduiront encore davantage le rejet de contaminants dans l'estuaire. Mais, à défaut de mesures de prévention, la pollution de source diffuse occasionnée par le développement urbain et agricole et le volume des effluents des stations de traitement des eaux usées continueront d'augmenter au rythme de l'accroissement démographique prévu dans le Lower Mainland.

La plupart des secteurs du cours principal du fleuve en amont de l'estuaire et ses principaux affluents n'affichent pas des concentrations importantes de contaminants. Ce fait s'explique en grande partie par la modification des méthodes de production dans les usines de pâte qui a entraîné, au début des années 1990, une réduction importante des concentrations de dioxines et de furanes dans les effluents; les améliorations apportées récemment aux stations de traitement des eaux usées y ont également contribué. La baisse marquée de l'utilisation de certaines substances chimiques, comme les biphényles polychlorés (BPC), le plomb, le pentachlorophénol et certains pesticides, qui a eu lieu au cours des deux dernières décennies, est également responsable des faibles taux de contamination observés. Des données toxicologiques ont été compilées sur certains polluants chlorophénoliques et les HAP qui sont présents en concentrations élevées en aval des usines de pâte. Les résultats indiquent que les substances testées n'ont probablement pas d'effets sur le biote aux concentrations observées dans le fleuve, à l'extérieur des zones de mélange. Les grands cours d'eau affichent généralement de faibles concentrations de polluants, si on les compare aux lignes directrices existantes concernant la protection des utilisations de l'eau et du biote aquatique, mais les usines de pâte et les stations de traitement des eaux usées sont encore à la source de nombreuses catégories de substances chimiques.

Le biote à proximité des sources de pollution réagit à ces faibles taux d'exposition aux polluants. Par exemple, des analyses de tissus de poisson et d'oiseau montrent une plus grande induction de l'enzyme de détoxification, l'oxydase à fonction mixte, dans les échantillons prélevés en aval des centres urbains et des effluents des usines de pâte. Alors que les deux espèces de poisson étudiées présentaient une assez forte incidence d'anomalies dans tout le bassin, la pigmentation prononcée du foie et des reins n'est couramment observée qu'en aval des usines de pâte dans le fleuve Fraser. À ces endroits, le développement des gonades chez les poissons est réduit tout comme chez les poissons de la rivière Thompson, en aval de Kamloops. On ne sait pas si ce phénomène aura une incidence importante sur la santé des poissons.

Des expériences menées dans des cours d'eau artificiels à Prince George et Kamloops montrent que les effluents des usines de pâte, aux taux de dilution enregistrés dans la rivière en période d'étiage, peuvent stimuler la croissance des organismes benthiques. Mais, les expériences montrent que des concentrations un peu plus élevées dans le fleuve font apparaître des signes de toxicité.

Des polluants organiques persistants ont été trouvés dans les eaux d'amont du fleuve. La source probable des BPC, du DDT et de ses métabolites et de la toxaphène décelés dans les poissons de ces lacs, surtout ceux en altitude, est le transport atmosphérique à grande distance et le dépôt des polluants, combinés à la libération, par la fonte des glaciers et des champs de neige permanents, de sédiments accumulés. Dans le contexte d'un réchauffement climatique, le rejet de polluants stockés dans les glaciers se poursuivra.

Ces résultats ont donné lieu aux recommandations suivantes en vue d'aider à préserver l'écosystème du bassin du fleuve Fraser.

Le ruissellement urbain et agricole est maintenant une source importante de polluants dans le bassin. Des stratégies visant à limiter le ruissellement des polluants devraient figurer au nombre des priorités des futurs programmes d'urbanisme et de développement agricole. Mentionnons 1) la réduction des surfaces imperméables, 2) l'aménagement de zones tampons en bordure des cours d'eau, 3) l'installation de bassins de décantation le long des cours d'eau et dans les égouts pluviaux, 4) la promotion et l'application de meilleures pratiques de gestion du bétail, d'entretien des fosses septiques et d'épandage des engrais (comme le fumier) et 5) la sensibilisation de la population à la gérance de l'environnement.

Afin de surveiller les progrès accomplis dans la restauration et la préservation du bassin du Fraser, il faut réévaluer périodiquement l'état de son écosystème aquatique. Le Plan d'action du fleuve Fraser a fourni un point de référence sur l'état du bassin. L'utilisation future des méthodes et des indicateurs issus du plan d'action permettra d'établir des comparaisons. On examine aussi la possibilité d'intégrer les méthodes du plan d'action dans les programmes de surveillance, comme le Programme de suivi des effets sur l'environnement et le programme conjoint Canada-Colombie-Britannique de surveillance de la qualité de l'eau. Par exemple, la surveillance des effets environnementaux pourrait comprendre le recours à la méthode des conditions de référence des macroinvertébrés benthiques, combiné à des expériences de contrôle dans le mésocosme, en vue d'évaluer les incidences des effluents en aval des points de rejet. L'intégration d'un plus grand nombre d'éléments biologiques et biochimiques dans les programmes de mesures périodiques facilitera le maintien de la santé de l'écosystème du bassin.

Les scientifiques qui ont participé à l'étude et compilé les résultats du programme ont décelé de nombreuses lacunes au niveau des connaissances. Deux lacunes dans les connaissances générales et certains besoins de recherche propres au bassin ont été soulignés. La première lacune concerne l'absence de compréhension des incidences cumulatives de l'exposition chronique à des mélanges de nombreuses substances chimiques, toutes à faible concentration. La seconde consiste à déterminer la quantité de données toxicologiques spécifiques à un milieu afin de compléter les lignes directrices élaborées à partir des essais biologiques habituels, lorsqu'on établit des objectifs de qualité de l'eau propres à des sites particuliers. Les organismes de gestion de l'environnement du bassin devraient examiner périodiquement les découvertes issues des recherches ultérieures sur ces questions et les appliquer.

Le programme a permis de cerner plusieurs besoins en matière de recherche qui sont particuliers au bassin. Il faut notamment déterminer la présence de perturbateurs endocriniens et leurs effets, établir les causes des anomalies observées chez les poissons du bassin, évaluer les incidences des substances chimiques anti-taches sur l'estuaire et évaluer la contamination de la chaîne alimentaire par les polluants atmosphériques dans les lacs d'amont. Certains de ces sujets de recherche ont été inclus dans l'initiative concernant l'écosystème du bassin de Géorgie.

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1.0

INTRODUCTION

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The Fraser River Basin is British Columbia's largest river basin, occupying approximately one-quarter of the province's area. The river's headwaters originate in three major mountain ranges (the Rocky, Columbia and Coast mountains) and a large plateau that separates the coastal range from the others. The geological, climatic and landform diversity in this basin is so great that it includes 11 out of the 14 biogeoclimatic zones identified in B.C. After a journey of 1,375 km from its farthest headwaters, the river empties into the Strait of Georgia in the Vancouver metropolitan area, the most densely populated region of the province.

The river and its basin occupy a special historic, political and socio-economic niche in the province. While salmon fishing and fur trading led to minor settlement and agricultural development around Fort Langley and New Westminster before 1855, the discovery of gold in the upper basin in 1856 provided the impetus to use the river's canyon as a path through the coastal mountain barrier (Hutchison 1950). The development of the interior's immense mineral, forest and agricultural resources followed, continuing through the last 140 years. The basin's present economy is still dominated by resource extraction, but now the highly urbanized population increasingly depends on the river to carry away its liquid wastes and on the basin's rivers, lakes and wetlands for recreation. Because of the Fraser basin's importance to the province, the federal government decided, in 1991, to assist citizens, industries and all levels of government in the development of new ecosystem management practices that would ensure the future sustainability of the basin's ecosystem and economy. All activities were coordinated through the Fraser River Action Plan (FRAP), jointly implemented by Environment Canada and the Department of Fisheries and Oceans between 1991 and 1997. This report describes the results of the Environmental Quality Program, one of the several programs that made up Environment Canada's FRAP initiative (Environment Canada 1998a).

RATIONALE FOR THE ENVIRONMENTAL QUALITY PROGRAM

Developing ecosystem-based management practices requires quantitative measures of ecosystem health which can be used to identify components under stress and to measure the results of programs aimed at controlling these stresses. To this end, the Environmental Quality Program was designed to assess the basin's present aquatic ecosystem health and the stresses affecting it, and to develop indicators of stress for subsequent management. The program was focused on pollution stress as there were concerns that this stress, already evident in the basin, would lead to a rapid decline in the ecosystem's biological productivity and diversity. There was also a need to establish a baseline of conditions to track the performance of pollution abatement strategies implemented or recommended by the Pollution Abatement Program of FRAP (Environment Canada 1998b).

The signs that pollution problems were serious or warranted investigation included the following observations:

- The levels of dioxins and furans in fish caught downstream of pulp mills near Prince George, Quesnel and Kamloops from 1988 to 1990 exceeded the human consumption guideline (Mah *et al.* 1989; Dwernychuk *et al.* 1991). This observation and the well documented history of dioxin contamination in the eggs of great blue heron living in the estuary (Whitehead 1989; Elliott *et al.* 1989), many hundreds of kilometres downstream, confirmed the basin-wide dimension of this contamination.
- The rapid population growth, especially in Vancouver's eastern suburbs, where 600,000 people now lived, was resulting in dramatic increases in the discharge of primary-treated municipal wastewater treatment plant (WWTP) effluent into the river at Annacis Island. This location is just upstream of one of the most ecologically productive and habitat-rich environments in the basin: the estuary and its intertidal delta foreshore. In addition, areas served with combined storm and sanitary sewers were contributing a mixture of contaminants from street runoff and untreated domestic sewage (Schreier *et al.* 1991). These combined sewer outfalls (CSOs) were discharging to nearshore habitats, which are important for younger and more sensitive life stages of fish and invertebrates. Along with the growth in population was an even greater growth in vehicle traffic, which was resulting in greater loading of polycyclic aromatic hydrocarbons (PAHs), metals (copper, zinc, chromium, manganese) and petroleum hydrocarbons from street runoff (Hall *et al.* 1991).
- The lumber industry was using up to 600,000 kg per year of new antisapstain formulations in the Lower Fraser Valley. Unknown quantities of the active ingredients in these formulations were being lost to the river in stormwater runoff from sawmill yards. While the industry had recently replaced pentachlorophenol and then 2-(thiocyanomethylthio) benzothiazole with less toxic chemicals, such as DDAC (didecyl dimethyl ammonium chloride) and IPBC (3-iodo-2-propynyl butyl carbamate), the effects of these new chemicals on the estuarine environment were unknown.
- Riparian habitats had been substantially altered in many small tributaries by urban and agricultural development. There was evidence that fish, including salmon, were being increasingly stressed by modifications to the normal physical (flow, temperature, turbidity) and chemical (nutrients, organic carbon, dissolved oxygen) characteristics of the streams (Northcote and Burwash 1991). In addition, contaminants running off suburban and agricultural areas were becoming a significant problem (Schreier *et al.* 1991). These stresses were expected to increase with development and to affect populations of fish and benthic invertebrates. Indeed, one endangered fish species (salish sucker—a genetically distinct form of longnose sucker, *Catastomus catostomus*) now only occupied limited lengths of the headwaters of a few streams in the Lower Fraser Valley (McPhail 1987).
- Finally, contamination of fish in remote lakes by persistent organic pollutants had emerged as an issue in the Rocky Mountains of British Columbia (Donald *et al.* 1993) and in southern Yukon (Kidd *et al.*

1993). Surprisingly high concentrations of polychlorinated biphenyls (PCBs) and toxaphene had been measured in top-predator fish from some of the lakes in these areas. The studies suggested that these pollutants could be transported via the atmosphere from industrial and agricultural areas in North America and Asia and then deposited in these drainage basins. It was not known how widespread this phenomenon was or if deposition rates would be even higher closer to potential sources in the basin or North America.

As the preceding observations indicate, the Fraser was showing signs of stress brought on by contamination from many point and non-point sources. While these problems were the result of past development practices which were being improved, the extent and intensity of urban, industrial and agricultural development was expected to increase over the next 25 years. It was imperative then to obtain an accurate assessment of contaminant stress levels in the basin's aquatic environment. Additionally, there was a need to understand the transport and fate processes that would allow the prediction of which ecosystem components were most at risk from any new contaminant releases to the basin. Both the assessment and the predictive capability were needed to ensure the effective prioritization of pollution abatement and prevention programs by environmental planning and management agencies in the basin.

PAST ASSESSMENTS AND RESEARCH

During the last 25 years, many studies have examined the impacts of specific contaminants in several reaches of the river or the cumulative impacts of many contaminants in a single reach (*e.g.* Federal-Provincial Thompson River Task Force 1976; Fraser River Estuary Study Steering Group 1979; Carey and Murthy 1988; Swain and Walton 1991; Bothwell *et al*. 1992). Most of these studies were reviewed and included in a comprehensive assessment of contamination from point and non-point sources by Hall *et al*. (1991) and Schreier *et al*. (1991). These two reviews highlighted several knowledge gaps that needed to be addressed to generate practical science-based environmental management information. They recognized the importance of characterizing and quantifying non-point source pollution, the critical need for effects-based assessment techniques, and the lack of understanding of the cumulative effects of contaminants. These reviews provided the foundation for designing the Environmental Quality Program.

OBJECTIVES OF THE FRAP ENVIRONMENTAL QUALITY PROGRAM

The Environmental Quality Program set out to answer the following questions about the present and future impacts of contamination in the Fraser basin:

1. What is the relative level of contaminant stress and how is it expressed in media concentrations and biological responses at the species, population and community levels?

2. What contaminant or classes of contaminants are responsible for the stress?

3. Are these contaminants exceeding available guidelines, criteria or objectives for the protection of aquatic life and other uses and are these adequately protective? If a guideline or an objective is not available for a contaminant, should it be developed?

4. Are present pollution abatement programs addressing the contaminants responsible for stress in the system and has the ecosystem responded positively to recently implemented abatement programs?

5. On what sub-basins or ecosystem components should ambient environmental assessment focus in the next decade?

6. Are there new indicator species, biotic communities or media components that would improve the assessment of the level of contamination and its impact on aquatic ecosystem health?

7. Are there better ways to evaluate the impacts of complex effluents in large rivers than the upstream-downstream comparison approach?

Of course, many environmental scientists working in this river basin over the years have asked similar questions. In this instance, FRAP provided the opportunity to apply new techniques and analytical tools on a much larger geographical scale and in a more interdisciplinary manner. Of even more importance, these new studies were undertaken over a relatively short time, making the assessment the first comprehensive "snap-shot" of contamination and its effects that has ever been attempted for the Fraser River and many of its tributaries.

Organization of the research and assessment projects

In order to generate the data and knowledge on the present status of contamination impacts and processes controlling contaminant fate and effect, the program was organized into three components.

The first component assessed the level of contamination in water and sediments in the Fraser main stem, its estuary and its major tributaries (Thompson and Nechako rivers); assessed its impact on specific species or communities; and developed new knowledge on sediment dynamics and fish movements in the basin. This assessment investigated tissue or whole-body contaminant concentrations, enzyme measures of contaminant exposure, abnormalities, growth and condition factors, and reproductive success in fish and wildlife. In addition, benthic macroinvertebrate community structure was determined at sites selected to represent most river orders and ecozones throughout the basin. The strategy for this component was developed, in part, from the results of a workshop, held in December 1992, involving federal, provincial and municipal environmental management agencies and university scientists (Bernard *et al.* 1993).

The second component assessed the impacts of specific sources of pollutants, particularly those from pulp mills, municipal WWTPs, urban areas, agriculture and the atmosphere.

The contaminant impacts from pulp mill effluents, and how to assess them, were discussed at a workshop held in April 1992. The recommendations from this workshop (Marmorek *et al.* 1992), which were largely implemented, included an examination of (a) the interaction of effluents and natural suspended sediments, (b) the impacts on benthic communities in the field and in mesocosms, (c) contaminant exposure and effects in fish and birds, and (d) the development of a contaminant fate model.

Assessing non-point source contaminants and their impacts in small tributaries in urban and agricultural areas was discussed extensively during the development of the Tri-Council Eco-Research study proposal submitted by the University of British Columbia to assess ecosystem functioning in the Lower Fraser Valley (Healey 1992). Selected research projects undertaken on three watersheds in the lower Fraser were enhanced with additional funding from FRAP. Other studies examined the impacts of agricultural runoff on amphibians and the effects of non-target pesticides on raptors.

The impacts of municipal WWTP effluents were assessed "retroactively" through a research project conducted to evaluate the recovery of the northern delta foreshore ecosystem following the diversion of Vancouver's WWTP effluent from the intertidal area to a deeper area 4 km offshore. Towards the end of the program, other WWTP effluents and downstream waters were analyzed for the potential endocrine disrupting chemical, nonylphenol, and other contaminants.

Finally, a project was undertaken to evaluate the extent and trends of atmospheric deposition of contaminants in headwater lakes through the examination of sediment cores and top predator fish.

The third component developed toxicity information and guidelines for contaminants that were identified as being of potential concern in the early stages of the program. A major effort was mounted to assess the toxicity of the two recently introduced antisapstain chemicals (DDAC and IPBC) on fish and invertebrate species living in the basin. A preliminary toxicity study of selected chlorophenolics and resin acids associated with pulp mill effluents was also undertaken.

The 22 chapters comprising this report are organized in five sections, the first being the introduction. Section 2 reviews contaminant sources. Section 3 contains ten chapters that summarize contaminant exposure and effects in the basin, sediment transport processes, and the development of biological indicator species and communities. Section 4 focuses on specific sources of contaminants, such as pulp mills, antisapstain facilities, agriculture and urban runoff, and their effects on the ecosystem. Guidelines for antisapstain chemicals and selected chemicals associated with pulp mill effluents are discussed, and a contaminant fate model is also included. Section 5 presents an integration of all the results, conclusions and recommendations found in the report.

Environmental scientists with appropriate expertise from Environment Canada, Fisheries and Oceans Canada, several universities and the private sector undertook the research and participated in its synthesis. The Environment Canada - Pacific and Yukon Region scientific resources for assessing aquatic pollution and wildlife contamination were heavily involved throughout the program. Major contributions to the program were provided by the Burlington and Saskatoon laboratories of Environment Canada's National Water Research Institute and by the Institute of Resources and Environment at the University of British Columbia. The coordination and development of the program was facilitated through three program workshops (Landucci 1995; Landucci and Hanawa 1996) and several sub-component workshops held during FRAP.

The Environmental Quality Program faced many scientific and organizational challenges during its planning and implementation stages. In the end, the greatest challenge was scientific, not organizational, as the techniques and approaches available to the scientific team could not fully answer every question listed earlier in this introduction. However, we believe the comprehensive assembly and synthesis of the new knowledge, generated by the program and summarized in this report, constitute a valuable legacy for environmental management in the Fraser Basin as well as other river basins in the region.

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CONTAMINANT SOURCES

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Many contaminants have been identified in the Fraser River Basin, both in effluents discharging to rivers in the basin and in the receiving environment. These include trace metals (*e.g.* copper and zinc) and organic compounds such as polycyclic aromatic hydrocarbons (PAHs), chlorinated phenolics, dioxins, furans and pesticides. The major sources of these contaminants and estimates of their loading to the basin have been documented by Environment Canada (1995), FREMP (1996) and Schreier *et al.* (1991).

Major sources include point sources such as pulp mill effluents and municipal wastewater treatment plant (WWTP) discharges, variable-flow point sources such as combined sewer outfalls (CSOs), and non-point sources such as urban, industrial and agricultural runoff. Relatively minor sources include five permitted metal mine discharges, atmospheric deposition and runoff from highways, railway right-of-ways and hydro-electric corridors.

There are effluent characterization data for most sources; however, loading data from many of these point and non-point sources, and their relative contribution to overall contaminant stress in the basin, are not available. Although conventional pollutant indicators such as total suspended solids (TSS), "oil and grease" and biochemical oxygen demand (BOD) are monitored in industrial and municipal effluents, efforts to sample a broader spectrum of contaminants have tended to focus on a few selected sites or industries. Concentrations of trace contaminants (*i.e.* metals and persistent organic pollutants) are not well characterized. Thus, overall mass loading of pollutants cannot be calculated accurately. Nor is there a consolidated database of all industrial effluent discharges in the basin.

Recent changes to manufacturing processes in the pulp and paper industry, and to effluent treatment in municipal WWTPs have resulted in significant reductions in the loading of contaminants of concern to the basin. As a result, the relative importance of non-point sources (*e.g.* agricultural and urban runoff, and atmospheric deposition) as contributors to contaminant loading has increased.

DISCHARGE VOLUMES AND CONTAMINANT LOADING IN THE FRASER BASIN

To get an overview of the relative importance of each major point source category, discharge volume data for WWTPs in 1991 (Environmental Protection Branch 1998) and estimates for industrial and pulp mill effluents in the middle to late 1980s (Schreier *et al.* 1991) were used (Table 1). Pulp mills and WWTPs were the major sources of waste water in the pre-Fraser River Action Plan (FRAP) era. The relative importance of these sources likely changed during FRAP in response to population increases in the Vancouver area in particular and the lack of recent industrial expansion in the basin as a whole.

Table 1. *Relative volumes of major discharges in the Fraser River Basin.*

	ANNACIS AND LULU ISLAND MUNICIPAL WWTPs	OTHER MUNICIPAL WWTPs[1]	INDUSTRIAL DISCHARGES[2]	PULP MILLS
Relative Volume	27%	9%	26%	38%

[1] excluding the Iona Island Municipal WWTP which discharges directly to the Strait of Georgia
[2] excluding pulp mills

The WWTP discharges from the Annacis and Lulu Island plants are separately tabulated to highlight the importance of these two Vancouver area plants which accounted for 75 per cent of the discharge from this category. This percentage is expected to increase over the next 20 years as the population is projected to increase by 50 per cent (GVRD 1997a). Assuming future industrial expansion does not result in large increases of water use, the WWTPs will become the dominant source, in terms of volume, in the near future.

The major pulp mill effluent discharges are located in the upper Fraser and Thompson rivers at Prince George, Quesnel and Kamloops. Two small-volume mill discharges are located at New Westminster and Burnaby in the lower Fraser River. While industrial discharges occur in Prince George and Kamloops, the majority of industrial activity is centred around Greater Vancouver, particularly along the North Arm of the lower Fraser River.

Detailed data on loading for many contaminants are not routinely collected, making loading estimation difficult. More traditional measures of effluent quality, such as BOD or TSS, are regularly obtained as part of permit requirements. Municipal effluent discharges represent a major source of loading for both BOD and TSS. TSS can be an approximate indicator of the loading of chemical contaminants, such as metals and organic compounds which are adsorbed to particles in the effluent.

Another measure of effluent quality, implemented in recent years, is that of effluent toxicity which can be measured using standard bioassays. Effluent toxicity integrates the cumulative effects of many contaminants. It can be used as a surrogate for determining the relative risk to aquatic ecosystems which receive effluent discharges. Much of the recent data on effluent toxicity in the Fraser Basin was collected under the auspices of FRAP funded projects, and the results of these studies are summarized below according to contaminant source.

MUNICIPAL DISCHARGES

Fraser Basin upstream of the Pitt River

Each day, 150,000 m^3 of sewage effluent is discharged to the Fraser River Basin from 87 treatment plants upstream of Langley (Environment Canada 1997d). As part of FRAP, Environment Canada initiated an effluent monitoring program at 15 WWTPs in the basin. These WWTPs were selected from within three basin sub-regions: (1) the upper Fraser including Prince George, Quesnel and Williams Lake; (2) the middle Fraser including Lytton, Lillooet, Cache Creek, Salmon Arm, Merritt, Hope, Enderby and Kamloops; and (3) the lower Fraser including the Kent Institution, Chilliwack, Aldergrove and North West Langley (Figure 1). The majority of these facilities have secondary treatment, the exceptions being Lytton, Lillooet and Hope which have primary treatment. The largest facilities included in the program were located in Prince George, Quesnel, Williams Lake, Kamloops, Chilliwack and Northwest Langley.

The 96-hr LC_{50} acute toxicity bioassay using rainbow trout (*Oncorhynchus mykiss*) was used to assess effluent quality. Each site was tested in the summer, fall, winter and spring. Results for 1992 and 1996 were similar, relative to the number of samples showing non-toxic conditions, although fewer samples passed the toxicity test in 1996 than in 1992. Based on the 1996 testing, 45 per cent of the wastewater effluents discharged to the upper and middle basin passed the bioassay, compared to 51 per cent in 1992.

In the middle Fraser, the Salmon Arm, Cache Creek, Enderby and Merritt WWTPs all have secondary treatment systems, and effluents were not acutely toxic on three out of four sampling occasions. Similar results were obtained in the lower Fraser at the Kent and Northwest Langley WWTPs which also have secondary treatment systems. The balance of the facilities frequently exhibited acute toxicity with 96-hr LC_{50} rainbow trout bioassay values as low as 41 per cent effluent concentration. These results suggest a trend of declining overall effluent quality. It is known that these facilities are experiencing steadily increasing volumes due to local population increases. As the results are based on a relatively limited sampling program, a more intensive study is required to confirm the apparent trend of increasing toxicity identified above.

At the least, these findings indicate that secondary treatment, although designed to reduce the loading of conventional parameters such as BOD and TSS (and thus contaminants such as trace metals and organic compounds, which are associated with these variables), may not consistently reduce acute effluent toxicity. No companion toxicity-identification evaluation (TIE) studies were conducted on the relative contribution of ammonia, metals or other effluent constituents to the observed toxicity. TIEs can identify causal agents of toxicity (Mount and Anderson-Carnahan 1988), and provide specific data necessary to recommend appropriate pollution abatement measures.

The Kamloops WWTP, which discharges effluent into the Thompson River just prior to its entry into Kamloops Lake, was also sampled for dioxin and furan compounds. None of these compounds were detected in the effluent (Environment Canada 1997d).

Fraser River Estuary

In 1995, the Annacis Island, Lulu Island and Iona Island WWTPs discharged a combined volume of 1.01 million m^3/day into the estuary and Strait of Georgia. Respectively, the Annacis and Lulu Island facilities discharged 421,000 and 57,400 m^3/day of wastewater effluent to the estuary (Environment Canada 1997d). The Iona Island WWTP discharged approximately 537,800 m^3/day directly to deep waters in the Strait of Georgia, and therefore, this discharge has not been included in the totals shown for the Fraser Basin in Table 1.

The wastewater of approximately one million basin residents is treated at the Annacis and Lulu Island WWTPs. On a basin-wide scale, the Annacis and Lulu Island WWTPs accounted for approximately 90 per cent of the TSS discharged from municipal WWTPs into the Fraser Basin as of 1995. By comparison, the

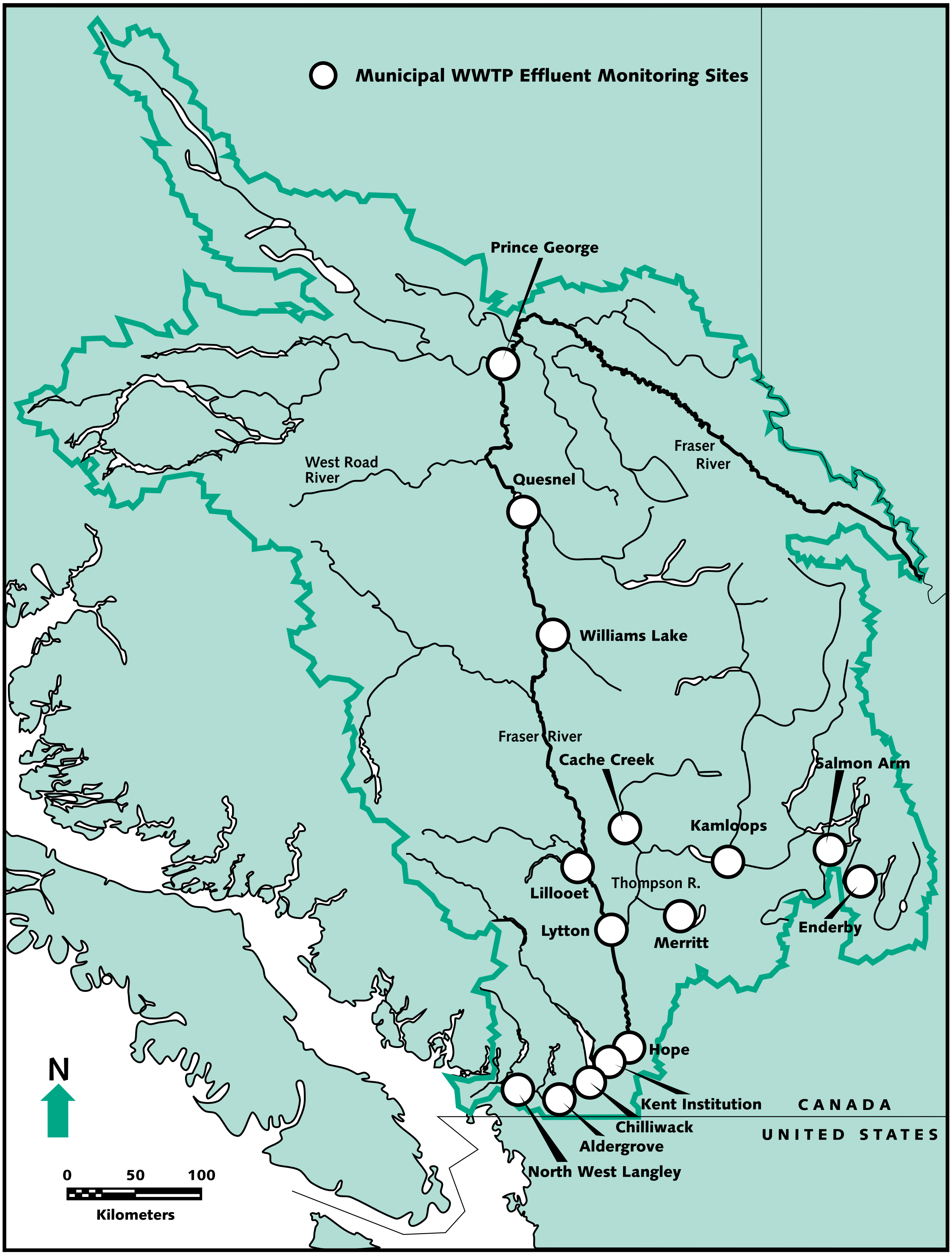

Figure 1. Municipal WWTPs sampled in the Fraser River Basin for FRAP.

next largest WWTP, the Prince George WWTP, contributed only 3.7 per cent of the basin's loading for TSS. Since completion of secondary treatment facilities at the Annacis and Lulu Island plants in 1998, a reduction of 70 per cent in TSS and of 85 per cent in BOD loading from WWTPs has occurred (Environmental Protection Branch 1998).

A summary of the annual loading of conventional contaminants from the Annacis and Lulu Island WWTPs in 1996 is given in Table 2. The nutrients and metals are monitored routinely at these WWTPs. A recent study conducted under the GVRD's Liquid Waste Management Plan also measured the levels of organic contaminants (*e.g.* PAHs, phthalate esters, nonylphenol and dichlorobenzene) (GVRD 1999).

The dominance of the Annacis and Lulu Island plants in the Fraser Basin is one of the reasons why they received special attention from federal and provincial agencies for upgrading to secondary treatment. As an example of the decrease in loading that can be expected from increased levels of effluent treatment, preliminary monitoring by the GVRD indicates that reductions in copper, zinc, TSS and BOD loadings of 85, 30, 85 and 87 per cent, respectively, have occurred. While the concentrations of contaminants in the effluents are now substantially reduced, there is a possibility that ammonia concentrations in the effluents could occasionally be high enough to be acutely toxic to fish (P. Wong 1997, pers. comm.).

Table 2. *Summary of annual loading in 1996 from the Annacis and Lulu Island WWTPs.*

CONSTITUENT MEASURED	ANNACIS ISLAND WWTP (TONNES/YEAR)	LULU ISLAND WWTP (TONNES/YEAR)
N-Kjeldahl	4000	640
N-nitrate+nitrite	<49	<2
N-ammonia	2900	450
Fluoride	10	5
M.B.A.S.[1]	590	100
Sulphate	4900	810
Total Phosphorus	570	93
Oil & Grease	4600	550
Phenol	8	0.9
Cyanide	<4	<0.5
Aluminum	100	20
Arsenic	0.2	<0.03
Boron	24	3.3
Cadmium	<0.09	0.02
Chromium	1	0.33
Cobalt	<4	<0.05
Copper	24	3.6
Iron	798	55.3
Lead	1	0.2
Manganese	29	2
Mercury	<0.09	<0.02
Molybdenum	<6	<0.08
Nickel	2	1.5
Selenium	<0.2	<0.03
Silver	0.5	0.28
Tin	<20	<3
Zinc	10	2

(Source: Greater Vancouver Regional District 1997a)
[1] Methylene-Blue-Active-Substances: this category includes anionic surfactants

COMBINED SEWER OUTFALLS

Combined sewer outfalls (CSOs) collect and convey both domestic sewage and urban stormwater runoff. CSOs are designed to accommodate high wet-weather flow conditions and are prevalent in the Lower Mainland area of the Fraser River Basin. Under dry weather operating conditions, sanitary sewage is conveyed to WWTPs. During wet weather conditions, the influx of large volumes of stormwater can overwhelm the piping system capacity. To reduce public health concerns and risks of flooding, overflow structures have been constructed to discharge excess flows to nearby receiving waters.

An inventory of CSOs within the Fraser Basin (including Burrard Inlet) was completed under FRAP (UMA Engineering 1992). Monthly discharge volumes and frequencies were estimated for each of the CSOs. Twenty of these CSOs, with an estimated annual discharge volume of 6.27 million m^3, discharge directly to the Fraser Basin. The CSOs discharging to the Fraser River make up approximately three per cent of the total volume of sewage effluent discharged to the basin.

Detailed overflow characterization data from four CSO systems in the GVRD showed concentrations of BOD, TSS and nutrients (ammonia and phosphorus) to be four to 14 times below that of typical GVRD raw sewage (GVRD 1996), reflecting the range of dilution from stormwater in these systems.

FREMP (1996) summarized contaminant loading from all major CSO discharges to different areas of the Lower Fraser River (Table 3). Loading estimates from this source made by Hall *et al.* (1998) were of a similar magnitude. CSO discharges occur intermittently during wet weather conditions when stormwater flows cause the sewer pipe capacity to be exceeded. This overflow mechanism results in highly variable daily and seasonal flows of additional amounts of raw sewage to the Fraser River. Operational improvements to CSOs in the New Westminster waterfront area are planned which will reduce discharge volumes by approximately 15 per cent.

Table 3. *Relative loading of ammonia, copper and suspended solids from CSO discharges to the Lower Fraser River.*

SOURCE	SUSPENDED SOLIDS (tonnes/yr)	AMMONIA (kgN/yr)	COPPER (kg/yr)
CSOs in Lower Main Stem of the Fraser River	59	1426	50
CSOs in North Arm of the Fraser River	210	5032	176

(Source: FREMP 1996)

URBAN STORMWATER RUNOFF

Considerable differences may exist in the relative volume of stormwater runoff from sites in the upper basin and in the lower reaches, due to the heavy rainfall experienced in the coastal zone. The relative impact of urban runoff can vary significantly with the degree of dilution that occurs when runoff mixes with the receiving waters. Smaller tributaries are therefore more affected by these runoff events than the main stem of the Fraser River.

The quantification of contaminant loading from urban runoff in the Fraser River Basin was the subject of a FRAP-sponsored study carried out by Stanley Associates Engineering Ltd. (1992). Annual contaminant loading for different portions of the basin was estimated from data on typical contaminant concentration ranges found in North American urban runoff and local urban runoff volumes. These loading estimates are summarized in Table 4.

Contaminants associated with urban runoff at Kamloops were the subject of a special investigation by Environment Canada (1997c). Elevated levels of heavy metals were present in all sediment and water samples collected from 15 storm sewer outlets discharging to the Thompson River. The highest level of heavy metal contamination was found at the Guerin Creek site which drains the highly industrialized area of Kamloops known as Aberdeen.

A FRAP-sponsored study of the Brunette River watershed in the Lower Mainland area of the Fraser Basin (Macdonald *et al.* 1997) was carried out to determine contaminant loading rates in a highly industrialized urban stream. This study found higher export coefficients (loadings expressed in kg N/ha/yr) for nitrate, ammonia and organic nitrogen than mean values reported in three other North American studies. The authors could not give a reason for such high export coefficients relative to the literature values. Elevated coefficients for copper and zinc were associated with runoff from the busiest traffic intersections. Traffic intensity and type (slow moving, high speed, parked) were found to be key factors related to runoff characteristics. Approximately 50 per cent of the cadmium, copper and zinc and up to 75 per cent of the manganese, chromium, lead and nickel were associated with suspended sediments in the street runoff. In a related study, Larkin and Hall (1998) found that 75 to 97 per cent of the flow-weighted concentrations of hydrocarbons in runoff was associated with the particulate fraction.

An accurate estimate of the total contaminant loading from stormwater runoff to the entire Fraser Basin cannot be made because data for the entire basin are not presently available. However, certain relationships between increased urban development and contaminant loading in parts of the basin have been established as a result of FRAP-supported studies. For example, Hall *et al*. (1999) have shown that contaminant loading increases proportionately with increases in vehicle traffic and with increases in the percentage of impervious surface area in the watershed.

Table 4. *Estimated annual contaminant loading (in tonnes) from urban runoff in the Fraser Basin.*

CONTAMINANT	FRASER BASIN	LOWER FRASER	THOMPSON	MIDDLE FRASER	UPPER FRASER
Suspended solids	62782	54584	1689	913	5596
Ammonia	75.3	65.5	2.0	1.1	6.7
Nitrate/nitrite	351.6	305.7	9.5	5.1	31.3
Total nitrogen	878.9	764.2	23.7	12.8	78.3
Total phosphorus	175.8	152.8	4.7	2.6	15.7
Lead	75.3	65.5	2.0	1.1	6.7
Copper	17.6	15.3	0.5	0.3	1.6
Zinc	75.3	65.5	2.0	1.1	6.7
Chromium	5.0	4.4	0.14	0.07	0.45
Cadmium	4.0	3.5	0.1	0.06	0.36
Nickel	12.6	10.9	0.3	0.18	1.1
Arsenic	6.5	5.7	0.2	0.1	0.6
Phenols	6.5	5.7	0.2	0.1	0.6
Total Hydrocarbons	2009	1747	54.1	29.2	179.1
PAHs	0.50	0.44	0.01	0.01	0.004

(Source: Stanley Associates Engineering Ltd. 1992)
Lower Fraser denotes the Fraser River watershed from Hope downstream
Thompson denotes the North and South Thompson and Thompson River watersheds
Middle Fraser denotes the Fraser River watershed upstream of Hope to about the West Road River confluence with the Fraser
Upper Fraser denotes the Fraser River watershed upstream of the West Road River confluence with the Fraser

PULP AND PAPER EFFLUENTS

Upstream of Hope, pulp mills represent the largest source (by volume) of effluent discharged to the Fraser River. In the late 1980s, elevated levels of dioxins and furans in Fraser Basin sediments, fish and wildlife (Mah *et al.* 1989; Elliott *et al*. 1989) prompted federal and provincial regulators to enact legislation requiring significant reductions in the permitted levels of dioxins and furans in mill effluent. Federal regulations also required mills to conduct effluent and receiving-water monitoring programs.

There are eight pulp and paper mills that discharge into the Fraser River Basin (Northwood Pulp and Timber Ltd., Canadian Forest Products Ltd. and Intercontinental Ltd. at Prince George; Cariboo Pulp and Paper Co. and Quesnel River Pulp Co. at Quesnel; Weyerhaeuser Canada Ltd. at Kamloops; Scott Paper Ltd. and Crown Paper Ltd. in the Lower Mainland) (Figure 2). All are subject to the federal Pulp and Paper Effluent Regulations. These federal regulations prohibit the discharge of acutely toxic effluent, and establish the quantities of BOD and TSS which may be discharged.

The following improvements in pulp mill effluents have been documented since 1990. The figures are for all B.C. mills, but can be considered as representative of those in the Fraser Basin.

- The average toxicity of the effluent improved from 50 per cent fish survival in a 65 per cent concentration of effluent to 100 per cent fish survival in 100 per cent concentration of effluent.
- The number of days toxic effluent was discharged decreased by 99 per cent.
- The quantity of BOD discharged decreased by 88 per cent, and is currently 13 per cent less than the maximum allowable amount.

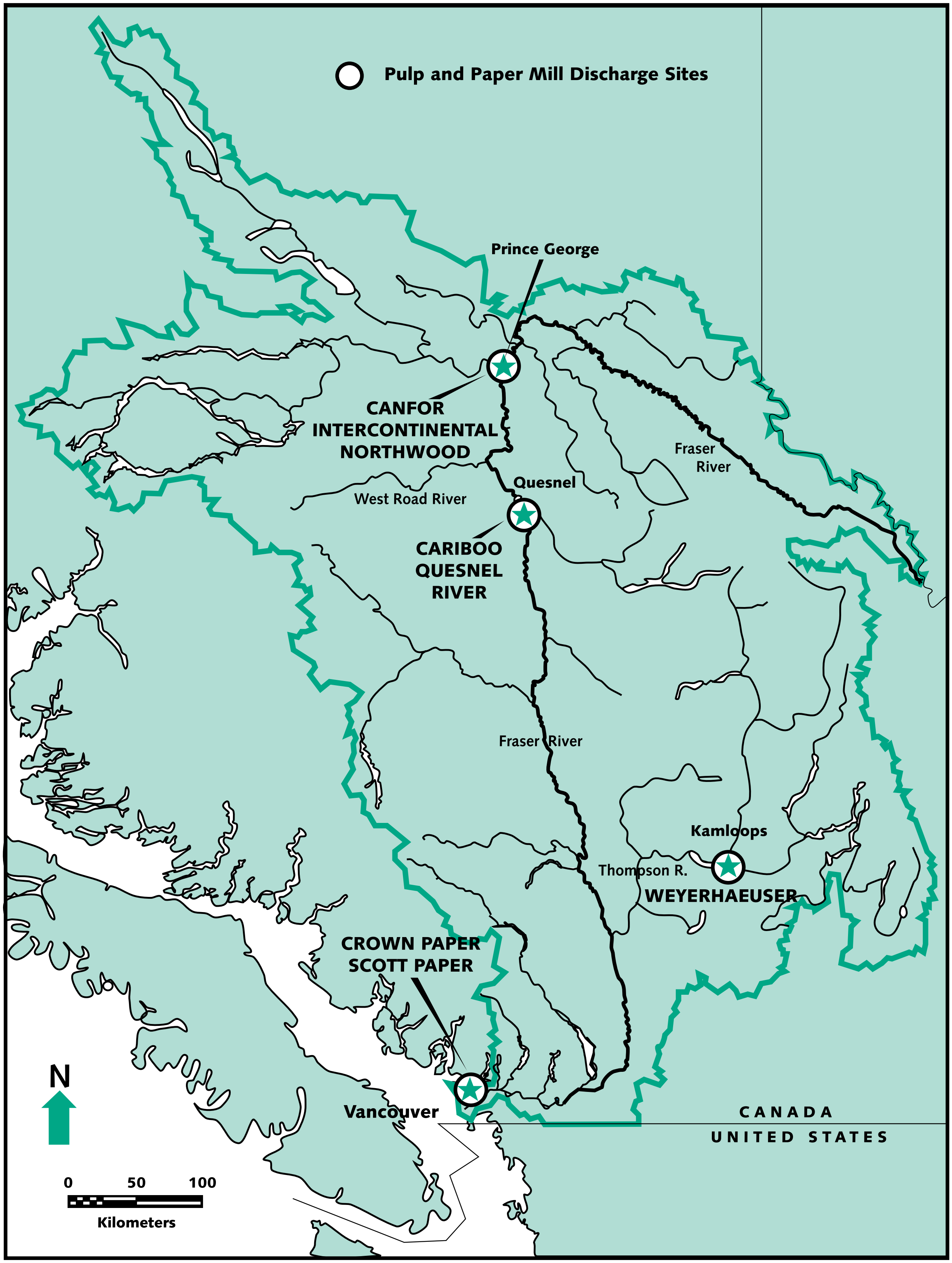

Figure 2. Location of pulp and paper mills in the Fraser River Basin.

- The quantity of TSS discharged decreased by 34 per cent, and is currently 26 per cent less than the maximum allowable amount.
- In 1996, B.C. mills demonstrated a 98.4 per cent compliance rate with the requirements of the Chlorinated Dioxins and Furans Regulations of the federal *Canadian Environmental Protection Act.* This resulted in a decline of over 99 per cent in the discharge of dioxins and furans since the regulations came into effect (see Figure 3; Environment Canada 1997e).

Although regulatory initiatives have focused on dioxins and furans, a range of other chlorinated and non-chlorinated organic compounds such as catechols, guaiacols, vanillins, and resin and fatty acids have been measured in pulp mill effluents (IRC 1994). Low levels of PAH compounds have also been detected in some pulp mill effluents in the Fraser Basin (Derksen 1997).

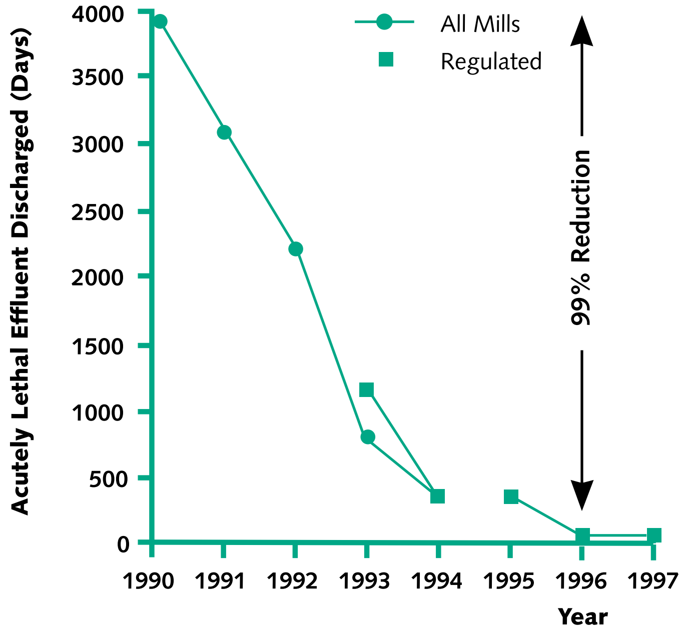

Figure 3*. Accumulated number of days (cumulative for all 23 mill discharges in B.C.) that Acutely Lethal Effluent (ALE) was discharged from B.C. pulp and paper mills.*
Source: Environment Canada 1997e.

OTHER INDUSTRIES

Lumber products industry

Antisapstains are chemicals used to protect freshly cut softwood lumber from discolouration due to the growth of fungi and moulds. The chemicals can be released to the receiving environment as the result of spills into storm sewers, or by being washed off freshly treated wood during rainstorm events. In British Columbia, the most common antisapstain fungicides are DDAC (didecyl dimethyl ammonium chloride) and IPBC (3-iodo-2-propynyl butyl carbamate). DDAC is one of the most heavily used pesticides in B.C. (Environment Canada 1997a) because of its use in treating large quantities of lumber for export. Currently, the majority of B.C. coastal mills use the commercial formulation NP1, a mixture of DDAC and IPBC. The second most common formulation in use today is F2, containing DDAC and disodium octaborate tetrahydrate as active ingredients.

Prior to 1987, there were approximately 100 facilities in British Columbia (70 in the Fraser River Basin) that discharged an estimated 260 million m^3/yr of acutely toxic stormwater. As shown in Figure 4, this volume had been reduced to approximately 1.6 million m^3/yr by 1996; a 99 per cent improvement (Environment Canada 1997a). In contrast, the amount of antisapstain chemicals used increased from about 350,000 kg/yr in 1987 to 846,000kg/yr in 1996. This increase was due to the replacement of chlorophenate with less toxic chemicals such as DDAC and IPBC which require heavier applications to achieve the same efficacy. Based on the 1996 inspection program carried out by Environment Canada, 87 per cent of the recommended best management practices had been implemented at the sites examined (Environment Canada 1997a). Best management practices include such operational changes as improved chemical handling, more effective application technology, and covered storage for the treated lumber. The actual loading of DDAC and IPBC to the river cannot be calculated accurately due to the lack of regular rainfall-event monitoring at most mills.

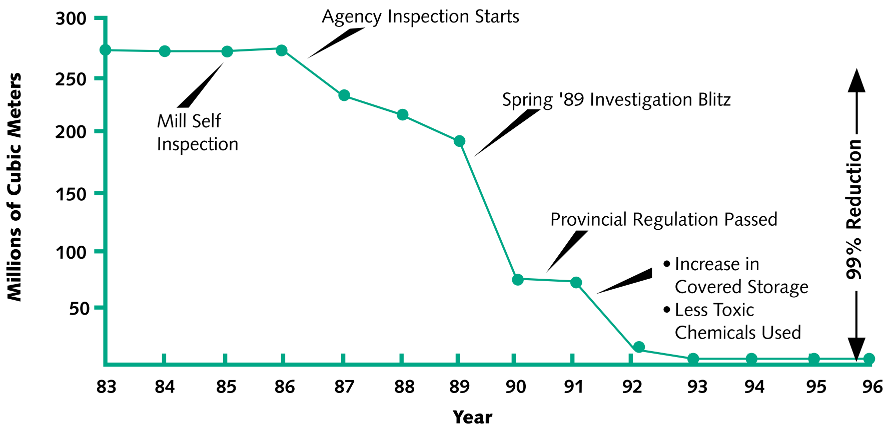

Figure 4. *Quantity of toxic effluent discharged annually from antisapstain facilities in British Columbia.*
Source: Environment Canada 1997a.

Heavy duty wood preservation industry

The heavy duty wood preservation industry is responsible for the manufacture of preserved wood products such as treated lumber, railway ties, pilings and utility poles (*e.g.* telephone poles). The wood is preserved with chemicals such as creosote, pentachlorophenol (PCP), chromated copper arsenate (CCA) and ammoniacal copper arsenate (ACA). Creosote and PCP are oil-based preservatives used in the manufacture of railroad ties, marine pilings and utility poles. ACA and CCA are water-based preservatives used in the manufacture of lumber, timbers and utility poles.

It is estimated that, prior to the initiation of FRAP, the annual effluent discharge from the heavy duty wood preservation industry along the Fraser River was 600,000 m^3. Contaminant loading from these effluent discharges is estimated to have been reduced by more than 90 per cent due to the FRAP enforcement initiative targeted at this industry sector (Environment Canada 1997b).

Permitted discharges from these facilities are typically limited to steam condensate from water used to make steam for site operations. Under normal conditions, any water discharged to the Fraser River is first run through an activated carbon/sand or similar filter system. Typical contaminant permit criteria specified by BC Ministry of Environment, Lands and Parks include maximum allowable concentrations for oil and grease, phenolics as phenol, pentachlorophenol, tetrachlorophenol, trichlorophenols, arsenic and copper. Unfortunately, PAH compounds, which are components of creosote, are not routinely measured.

Stormwater is generally collected and reused on-site in the water-based preservative operations. Thus, it is not considered a source of contaminant loading to the Fraser River Basin.

Inspections by Environment Canada have found that historical seepage of creosote into the soil at former industrial operations along the lower Fraser River has resulted in significant underground reservoirs of contaminants. These deposits may be future long term sources of contamination to the river (Environment Canada 1997b). Actions are presently underway to initiate remedial programs at these affected sites.

AGRICULTURAL OPERATIONS

Agricultural practices create non-point sources of contaminants. These contaminants, including nutrients, bacteria and pesticides, can impair the water quality of both ground and surface waters. Agricultural activities are a major source of contaminants to streams in the Lower Fraser Valley, the Thompson/Shuswap, Vanderhoof and Cariboo regions of the Fraser River Basin.

Agricultural intensification in the Lower Fraser Valley has reached levels where pollution of streams and groundwater is widespread (FREMP 1996; Schreier *et al.* 1999). While accurate estimates are not available for nitrogen and phosphorus loading to tributaries in the central and eastern sectors of the Lower Fraser

Valley, these loadings, combined with the 1,400 tonnes of nitrogen and 300 tonnes of phosphorus (FREMP 1996) estimated for the western end of the valley, likely result in a total loading to the lower Fraser River of the same order as those contributed by GVRD WWTPs.

Detailed estimates of nutrient losses to streams throughout the basin require data on fertilizer application rates, number of animal units/area (see Schreier *et al.* 1999), soil characteristics and local hydrogeology. All of these data are not available for estimating basin-wide loading of nutrients. Similarly, data are not available on the rates of pesticide applications and the amounts entering waterways in the basin. Consequently, pollution control has focused on the development and implementation of "best management practices" by the agricultural community (Environmental Protection Branch 1998) and monitoring conditions in affected waters.

ATMOSPHERIC SOURCES

Atmospheric contaminant sources that have been assessed recently in British Columbia include vehicular emissions, agricultural emissions and long-range transport. Studies have focused on the Lower Fraser Valley, which has the highest population density and the most intense agricultural activity in the province.

Contaminants associated with vehicular emissions include metals and organic chemicals (McLaren *et al.* 1996; Pott 1996; Barrie and Vet 1984). These contaminants have been measured in the ambient air of the Greater Vancouver area, as well as in snow samples from the mountains along the northern slopes of the Lower Fraser Valley. Snow from one-metre-depth cores collected in 1995 from the mountains in the Lower Fraser Valley showed few patterns in contaminant concentrations relative to distance from the Greater Vancouver urban centre (Belzer *et al.* 1998c). The exception was zinc, and possibly copper and manganese, which had maximum values at sites closest to the urban centre. These three metals are often associated with transportation sources.

Agricultural activities have intensified in the Lower Fraser Valley (Schreier *et al.* 1999) and, as a consequence, have resulted in increases in the atmospheric concentration of some of the chemicals found in fertilizers and pesticide formulations. For example, ammonia volatilizing from manure is likely the major natural source of nitrogen compounds to the atmosphere. Ammonia has been linked to the formation of atmospheric particulates which are believed to create episodes of poor air quality in eastern portions of the Fraser Valley (Pryor *et al.* 1997).

Table 5 presents a summary of nitrogen compounds and sulphate measured over a year in dry deposition and for seven months in wet deposition at Abbotsford, an agricultural community in the Lower Fraser Valley (see Figure 5). Based on these data, the mean total nitrogen deposition is about 11.5 kg nitrogen/ha/yr (Belzer *et al.* 1998a). This represents a significant component of the 50 to 100 kg nitrogen/ha/yr which has been suggested as an acceptable level of nitrogen loading to valley agricultural soils (Summary Report Steering Committee 1997). However, the largest source of nitrogen loading to these soils is from manure application. For example, some areas in the Lower Fraser Valley are estimated to receive over 300 kg nitrogen/ha/yr, largely from manure (Summary Report Steering Committee 1997).

Pesticides and herbicides that have been applied over agricultural crops were observed in the air and precipitation at Agassiz and Abbotsford (Table 6) (Belzer *et al.* 1998b). Some of the pesticides (*e.g.* aldrin, dieldrin, chlordane) measured within this airshed are no longer applied in the local area, suggesting that their presence may be due to long-range transport or emission from local soils (Finizio *et al.* 1998). Long-range transport of contaminants is also identified as the major source of pesticides, and possibly PCBs, to the aquatic environment of large lakes in the basin (MacDonald *et al.* 1999).

Table 5. Mean nitrogen and sulphate deposition at Abbotsford. Measurements are based on weekly samples taken as follows: dry deposition sampled from January 1996 to February 1997; wet deposition sampled from July 1996 to February 1997. (Yearly estimates are based on the number of days for either dry [199 days] or wet [165 days] deposition; some sample periods had both wet and dry deposition contribution).

MEAN (DEPOSITION AS mg N OR SO_4)	NH_3	NO_3	NO_2	SO_4
Daily dry deposition (mg/m^2/day)	1.41	0.041	0.011	0.212
Daily wet deposition (mg/m^2/day)	4.04	1.12	0.050	4.40
Yearly dry deposition (mg/m^2/year)	281	8.21	2.12	42.2
Yearly wet deposition (mg/m^2/year)	666	184	8.25	726
Yearly Total deposition (mg/m^2/year)	947	192	10.4	768

(Source: Belzer et al. *1998a)*

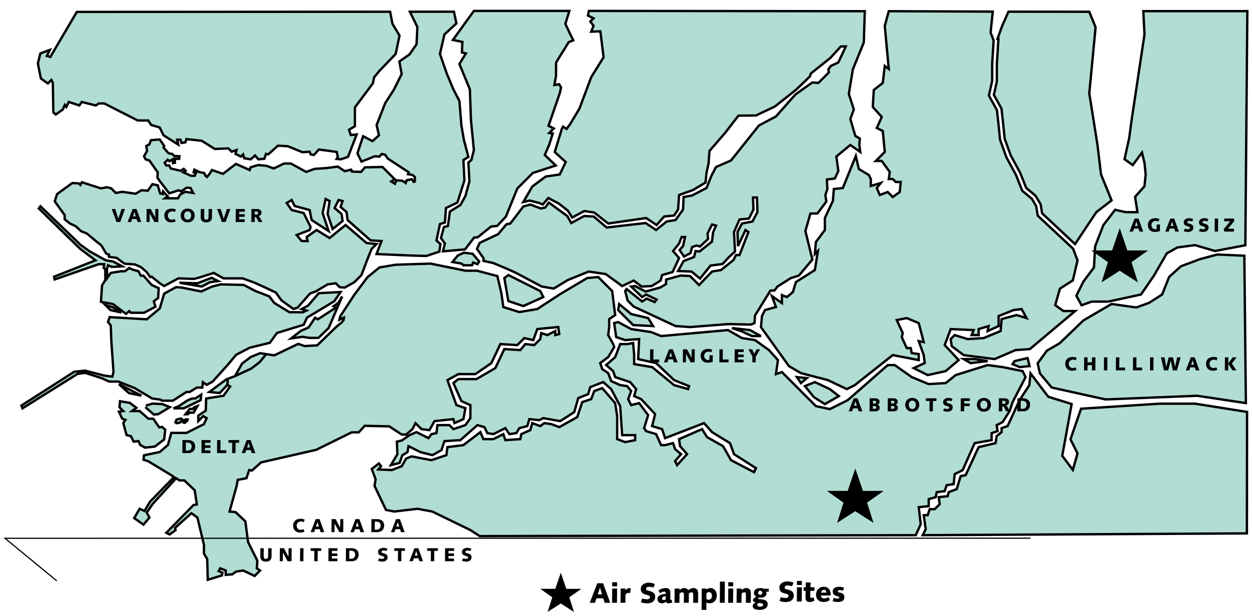

Figure 5. Lower Fraser Valley air quality sites sampled for contaminants associated with agricultural activities.

SUMMARY OF MAJOR SOURCES OF CONTAMINANTS TO THE FRASER BASIN AND LIKELY FUTURE TRENDS

Based on the information presented above, the major sources of chemical contaminants to the Fraser River Basin have been listed in Table 7, together with the most likely future trends for each source.

CONCLUSIONS

- Due to increased regulation, enforcement and compliance of point-source effluent discharges, non-point sources such as runoff from intensive agriculture and heavily urbanized areas are increasingly responsible for chemical contamination in the basin.

Table 6. Mean concentrations (ng/m³) of agricultural chemicals in ambient air samples collected weekly from January 1996 to December 1996.

PESTICIDE GROUP	CHEMICAL	AGASSIZ	ABBOTSFORD
Organochlorine Pesticides	Aldrin	ND	0.246
	Captan	1.448	1.823
	cis-Chlordane (a)	0.226	0.188
	trans-Chlordane (g)	0.260	0.102
	Dacthal	0.363	0.478
	4,4'-DDE	0.139	ND
	Dicofol	0.337	ND
	Dieldrin	1.010	0.062
	Endosulfan I	0.708	0.620
	Endosulfan II	0.253	0.184
	Heptachlor	0.148	1.024
	Heptachlor Epoxide	0.288	0.131
	Hexachlorobenzene	0.474	0.190
	Lindane (g-BHC)	0.338	0.213
	cis-Nonachlor	0.184	ND
	trans-Nonachlor	0.217	0.077
	Oxychlordane	0.278	0.244
Herbicides	2,4-D	6.646	2.301
	Dicamba	1.708	ND
	Dinoseb	ND	4.770
	Silvex (2,4,5-TP)	2.065	1.242
	Atrazine	5.529	2.622
Organophosphate Pesticides	Chlorpyrifos	0.612	0.666
	Diazinon	0.484	4.664
	Dichlorvos	2.990	1.172
	Dimethoate	0.340	ND
	Fonofos	0.957	0.128
	Malathion	1.963	3.688
	Mevinphos	ND	5.556
	Parathion Methyl	0.157	0.418
	Terbufos	0.512	1.246

(Source: Belzer et al. *1998b)*
N.D. denotes not detected

- Municipal wastewater discharges will continue to be a major source of contaminants to the basin due to projected population growth and the continued acute toxicity of many municipal effluents receiving secondary treatment. However, discharges from CSOs should decrease.
- Discharges of dioxins and furans from pulp mills have declined by 99 per cent and effluents now consistently pass acute toxicity tests.
- A large decrease in the release of antisapstain chemicals has occurred since the early 1990s. Increases are not likely as lumber production in the lower Fraser area has declined.
- Heavy duty wood-preservative chemical loading has declined by over 90 per cent since the early 1990s.
- Atmospheric deposition of locally generated airborne contaminants is likely to escalate with population growth. There is evidence of deposition of airborne contaminants brought by long-range transport; however, loading from this source has not been quantified.

RECOMMENDED FUTURE STUDIES AND ACTIONS

- Develop strategies for monitoring non-point source contaminant input to sensitive aquatic ecosystem components of the basin (*e.g.* tributaries affected by increasing urbanization and areas of intensive agricultural operation).
- Expand the routine use of bioassays (including sublethal tests) on contaminant sources as a measure of their cumulative toxic loading. Toxicity identification evaluation (TIE) studies should be used to identify specific causal toxic agents and guide mitigation efforts.

Table 7. Summary of likely future trends for major sources of contaminant loadings.

LOADING SOURCE	LIKELY FUTURE TREND	COMMENT
Municipal Discharges	↑	- volumes expected to increase with increases in basin population - loading dependent on level of treatment and its effectiveness.
Combined Sewer Outfalls	↓	- loading to decrease with implementation of Liquid Waste Management Plans.
Pulp and Paper Effluents	→←	- if controls are required for other compounds besides those currently regulated, total contaminant loading will decrease.
Urban Stormwater Runoff	↑	- expected to increase in proportion to increasing population and urbanization.
Agricultural Runoff	↑	- reversal of present increasing trend dependent upon successful implementation of best management practices in areas of intensive agricultural operations. - more detailed tracking of fertilizer, pesticide and other agro-chemical use are required to evaluate losses to surface and ground waters
Lumber Products Industry	→←	- future loadings of antisapstains dependent on lumber exports and any changes in effluent guidelines.
Heavy Duty Wood Preservation Industry	↓	- decrease in loading dependent on high degree of regulatory control on discharges, and good industry compliance record.
Atmospheric Sources	↑	- regional emissions are expected to increase with population - long-range transport of contaminants has not been quantified.

↑ *= indicates increased loading and a high degree of ecological concern*
↓ *= indicates decreased loading and a low degree of ecological concern*
→← *= indicates a degree of uncertainty with respect to future trend in contaminant loading and in ecological concern; may be dependent upon successful completion of best management practice or other pollution abatement program or compliance with future regulations*

- Implement routine monitoring of the effectiveness of pollution abatement and prevention measures with pre- and post-treatment assessments including collection of data for both contaminant loading and aquatic ecosystem responses.
- Expand the characterization of pulp mill effluents to include more organic compounds of concern, as well as dioxins and furans.
- Evaluate impacts of existing levels of airborne contamination and monitor changes in deposition of airborne contaminants from local and global sources.
- Develop an easily accessible watershed database for contaminant loading data from different industry and government programs.
- Develop an information system to track the total amounts of agrochemicals used in sub-watersheds.

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CONTAMINANTS IN LAKE SEDIMENTS AND FISH

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Lakes form an integral part of the Fraser River system. They are unique freshwater habitats that store large amounts of water and trap suspended sediments delivered by inflowing rivers. As integrators and collectors within their sub-basins, lakes provide an opportunity to study contaminant entry and transport within a river system on a watershed by watershed basis. For this purpose, we collected sediment cores from six lakes (Fig. 1; Table 1) and, from four of those, collected one species of fish. These lakes span basin-scale physiography including the Rocky Mountains, the Interior Plateau and the Coast Mountains. Some of the watersheds have seen little or no development (Moose, Chilko, Stuart, Harrison), whereas others have considerable industrial, municipal or agricultural activity (Kamloops, Nicola). The residence time ranges from eight weeks (Kamloops) to 17 years (Chilko) and all lakes are deep enough to be thermally stratified.

Lake sediments often contain a contaminant record of, for example, metals (Spliethoff and Hemond 1996; Von Gunten *et al*. 1997), organochlorines (Eisenreich *et al*. 1989; Muir *et al*. 1996; Oliver *et al*. 1989; Pearson *et al*. 1997; Sanders *et al*. 1994), polycyclic aromatic hydrocarbons (PAHs) (Christensen and Zhang 1993; Furlong *et al*. 1987; Vilanova *et al*. 1995; Wakeham *et al*. 1980) and other chemicals (Reiser *et al*. 1995). Indeed, sediments may provide the only means to reconstruct contaminant histories where monitoring is lacking. Although the literature gives many examples of lake sediment-core studies, these usually focus on an individual lake and a specific contaminant or source. Recently, northern Canadian lakes (Muir *et al*. 1996) and reservoirs in the United States (Van Metre *et al*. 1997) have been studied systematically to determine long-range transport of volatile contaminants over large scales. The Fraser River Action Plan

Table 1. Hydrographical and limnological characteristics of the lakes.

	MOOSE	STUART	CHILKO	KAMLOOPS	NICOLA	HARRISON
Area (km^2)	13.9	358	200	52.1	24.9	510
Volume (km^3)	0.73	9.3	23	3.70	0.57	80.5
Mean Depth (m)	52	26	137	71	23.5	158
Maximum Depth (m)	87	95	366	143	54.9	270
Bulk residence time (yr)	1.6	2.3	17	0.16	3.0	5.6
Discharge (km^3/yr)	0.46	4.1	1.3	22	0.19	14.3
Watershed area (km^2)	1640	14600	2110	40386	2990	7870
Lake Yield (m/yr)	33	11.5	7.7	422	7.6	28
Watershed Yield (m/yr)	0.28	0.28	0.62	0.54	0.06	1.8
Lake Elevation (m)	1032	680	1172	336	627	10
References	2,3,4,6	2,3,4	2,3,4,5	1,2,3	2,3,4,5	2,3,4,7

1. Pharo and Carmack 1979
2. Provincial Maps giving bathymetry, areas, volumes and hypsometry
3. Water Survey of Canada. 1990a.
4. Water Survey of Canada. 1990b.
5. Provincial Lake file (Richard Dobrowski, pers. comm.)
6. Desloges and Gilbert 1995
7. Desloges and Gilbert 1991

offered the first opportunity to conduct a systematic survey of contaminant signals in dated cores from a number of lakes within the varied watersheds of this vast river basin.

Burbot (*Lota lota*) are freshwater cod with circumpolar distribution (Scott and Crossman 1973). In lakes and rivers, these predators typically occupy the top trophic position. In some areas of Canada, burbot are valued as a sport fish and the livers, in particular, are consumed by First Nations fishers. This species is therefore particularly appropriate for the study of persistent contaminants owing to the high trophic status, the lipid-rich livers which sequester hydrophobic organic contaminants, and the large existing database for comparison (Kidd *et al*. 1993; Kidd *et al*. 1995c; McCarthy *et al*. 1997; Muir *et al*. 1990; Muir *et al*. 1997). Unfortunately, burbot are not found in Chilko or Harrison lakes (McPhail *et al*. 1998), but are present in the remaining four lakes.

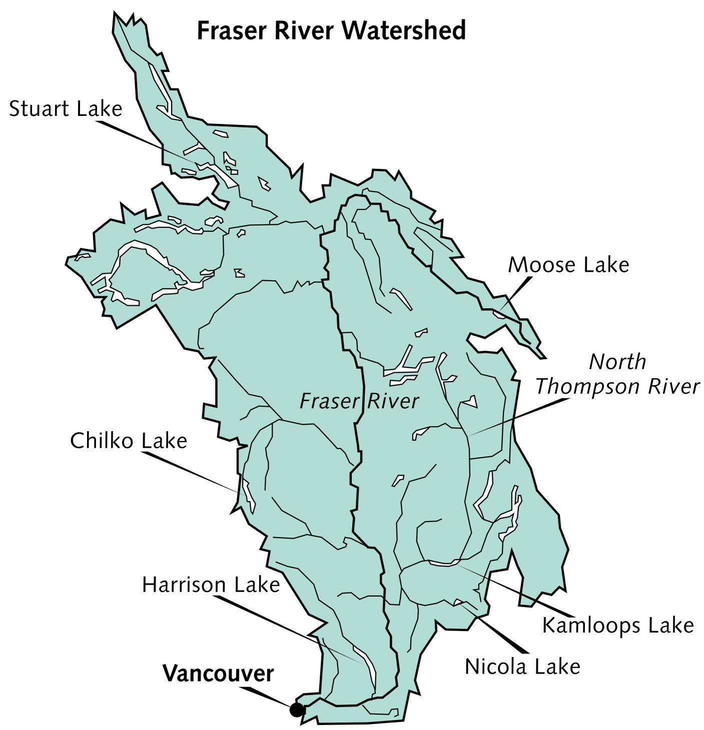

Figure 1. The Fraser drainage basin showing the locations of the six lakes.

Contaminants can enter lakes in at least two ways: (1) long-range atmospheric transport with deposition directly into the lake or onto the drainage basin and from there, through runoff, into the lake and (2) local, or regional, inputs of contaminants either through industrial or municipal effluent or via runoff from contaminated soils or surfaces. Contaminants that become strongly bound to particles may then be trapped in lakes where sediments tend to settle. Provided there is little sediment mixing or post-deposition mobility, sediments can provide a reliable history of contaminant burdens to the lake. Often, the source of the contaminant can be surmised either from the composition of classes of chemicals or from the date of contaminant appearance in a core compared to a known date of discharge.

For simplicity and brevity we present here only salient or representative features of the contaminant data in sediment cores and in burbot liver tissue. The sediment contaminant analyses include selected metals, hydrocarbons (PAHs and alkanes), organochlorine pesticides, polychlorinated biphenyls (PCBs) and chlorinated dioxins/furans (PCDD/Fs). Not all core sections have been analyzed for all contaminants; however, with the exception of dioxins and furans which were analyzed only for Kamloops Lake, measurements have been made for all of the contaminants on at least some core sections from each of the six lakes. The burbot liver analyses include organochlorine pesticides and metabolites, dominant toxaphene chlorobornanes, PCB congeners (both non-ortho and coplanar PCBs), total mercury (Hg) and methyl mercury.

The motive for the study was to assess whether long-range transport of airborne contaminants was a concern in the Fraser Basin. In that context, headwater systems above the influence of point sources should provide the best opportunity to isolate atmospherically derived contaminant signals and there was good evidence that such signals would be encountered (Donald *et al.* 1993; Gregor 1990). Sediment coring was used to determine the rates and recent history of contaminant input, whereas burbot liver was used to evaluate contaminant partitioning into biota and whether concentrations in fish approached guideline levels for human consumption or for the protection of wildlife dependent on fish.

METHODS

Cores were collected with a Kahl-Benthos gravity corer with a 10 cm acrylic tube (no catcher or cutter). Coring sites were generally situated in bathymetric depressions toward the centres of the lakes. All cores were sectioned within a few hours of collection by extruding the core upward incrementally. Sub-samples were stored frozen in cleaned glass (organic compounds) or plastic (metals, ^{210}Pb) containers. The cores collected for this study were dated using ^{210}Pb, ^{137}Cs measurements and by counting varves, where present. See Table 2 for calculated depth and mass accumulation rates based on these measures. The original data (dry-weight concentration vs. sediment depth) have then been converted to concentration versus time using these estimated rates (Table 2). Uncertainties in these rates should be borne in mind when considering the time assignments for sediment core slices or in assigning exact dates of entry of any given contaminant. Analytical methods and quality assurance protocols are fully described elsewhere (Macdonald *et al.* 1998; Macdonald and Paton 1997) and by the contract laboratories.

Table 2. *Sedimentation rates at the core sites.*

LAKE	LAT (N)	LONG (W)	DEPTH (m)	SED RATE g/cm²/yr	95% CI g/cm²/yr	SED RATE cm/yr	VARVES cm/yr	^{137}Cs cm/yr
Moose-1	52° 57.132	118° 54.395	80	0.33	0.27–0.44	0.31	0.35	—[2]
Moose-2	52° 57.727	118° 56.225	74	0.20	0.18–0.24	0.214	0.26	—
Stuart-1	54° 39.669	124° 50.873	31	0.040	0.033–0.050	0.16	n/a[1]	—
Stuart-2	54° 31.010	124° 32.010	45	0.030	0.026–0.035	0.13	n/a	0.14
Chilko-1	51° 29.344	124° 12.054	153	0.064	0.050–0.089	0.14	n/a	—
Chilko-3	51° 26.650	124° 11.836	162	0.11? (slump)	—	—	0.22?	n/a
Kamloops-1	50° 46.669	120° 45.760	94	0.26	0.19–0.38	0.51	0.45	—
Kamloops-2	50° 44.808	120° 40.611	91	0.32	0.24–0.46	0.49	0.55	—
Nicola-1	50° 11.131	120° 29.899	50	0.090	0.071–0.12	0.38	n/a	0.35
Harrison-1	49° 34.519	121° 52.980	258	0.18	0.15–0.22	0.34	n/a	—

1. Not applicable: varves were not observed
2. — : not analyzed

Five large burbot (>50 cm total length) were collected by angling and trapping from Moose, Stuart, Kamloops and Nicola lakes. Livers were analyzed individually by contract laboratories. Detection limits varied between analytes and with sample mass, but averaged less than 1 ng/g wet weight. Analytical details are described fully in Shaw and Gray (in prep.).

INDIVIDUAL LAKE OBSERVATIONS

A summary of all the analytical and calculated results for contaminants in sediments and fish and the sedimentary fluxes of the contaminants are presented on the following pages in Tables 3, 4, 5, 6 and 7 to facilitate easier access, as each lake's results are discussed.

Table 3. *Summary of metal data for sediment cores.*

METAL µg/g	MOOSE	STUART	CHILKO	KAMLOOPS	NICOLA	HARRISON
Pb (background)[1]	31.0 ± 2.1	15.5 ± 0.86	13.2 ± 2.8	26.3 ± 2.6	8.4 ± 0.6	17.1±1.2(8)
Pb (maximum)[2]	**38.1**	**22.3**	20	**37.5**	**11.3**	**26.5**
Zn (background)	86.7 ± 6.4	119 ± 9	164 ± 15	147 ± 13	87.0 ± 6.6	182 ± 10
Zn (maximum)	**130**	136	188	173	103	**213**
Cu (background)	35.0 ± 1.9	47.6 ± 4.7	83.8 ± 8.5	62.3 ± 5.1	77.4 ± 6.9	105 ± 6
Cu (maximum)	**50.3**	**65.9**	92.6	69.9	89.8	**124**
Ni (background)	41.8 ± 4.6	91 ±8	46.0 ± 11.7	79.6 ± 8.5	45.8 ± 4.3	42.5 ± 1.2
Ni (maximum)	**134**	**117**	63	89.9	52.9	**59.5**
Co (background)	17.9 ± 1.2	17.0 ± 0.7	27.7 ± 3.5	30.0 ± 2.5	19.3 ± 1.7	28.2 ± 1.4
Co (maximum)	**32.2**	**22.3**	32.3	35	22.5	29.5
Cr (background)	113 ± 4	78.5 ± 8.5	84.0 ± 7.5	151 ± 10	106 ± 4	79.4 ± 4.2
Cr (maximum)	**140**	102	102	169	**132**	**110**
Hg (background)[4]	12.5 ± 1.3	19.4 ± 3.3	29.6 ± 7.5	20 ± 3	—[3]	—
Hg (maximum)[4]	**24**	**158**	42	**52**	—	—

1. *Background values were calculated as the average of the bottom 3-4 samples for the cores (± 1 sd).*
2. *Bold numbers identify where a case can be made for significant contamination based on the metals profile in the core.*
3. *Not analyzed*
4. *ng/g*

Table 4. *Summary of maximum concentrations for organochlorine compounds in sediment cores.*

COMPOUND ng/g	MOOSE	STUART	CHILKO	KAMLOOPS	NICOLA	HARRISON
# of samples	18	5	5	8	11	12
HCB	0.2	0.44	0.4	0.17	1.5	0.06
Total HCH	0.14	0.09	0.13	0.29	6.1	0.06
Heptachlor	<0.02–<0.1[1]	<0.06–<0.17	<0.04–<0.43	<0.03–<0.73	<0.003–<0.38	<0.003–<0.07
Aldrin	<0.002–<0.01	<0.002–<0.01	<0.007–<0.09	<0.007–<0.09	0.14	<0.002–<0.02
Chlordane	0.01	0.07	0.02	<0.006–<0.33	3.1	0.06
Total DDT	1.73	1.25	0.32	1.95	323	8.7
Total Nonachlor	0.04	0.06	0.06	<0.006–<0.37	1.4	0.05
Mirex	0.007	0.008	0.01	<0.001–<0.08	0.25	0.06
Dieldrin	0.06	<0.06–<0.22	<0.05–<0.47	0.92	<0.06–<0.62	<0.02–<0.22
Endrin	<0.09–<1.3	<0.22–<0.8	<0.08–<2.1	2.6	<0.15–<1.2	<0.04–<0.53
Methoxychlor	<0.14–<1.7	<0.32–<1.0	<0.2–<4.2	<0.18–<7.9	<0.21–<2.6	<0.2–<0.94
Toxaphene	<0.12–<0.57	<0.46–<1.8	<0.04–<1.1	<0.2–<4.3	<0.13–<2.2	<0.08–<1.2
Total PCB	0.86	0.68	0.94	7.7	1.49	1.5

1. *The range of sample detection limits given where compounds were not detected.*

Table 5. Summary of biological data for five burbot obtained from each lake and chemical analyses of their livers.

VARIABLE	KAMLOOPS	NICOLA	STUART	MOOSE[7]
Biological Variables[1]				
Age (years)	7 (5+–8+)	7 (6+–7+)	10 (9+–12+)	7 (5+–8+)
Fish Length (cm)	55 (47–61)	52 (48–58)	79 (70–96)	53 (40–63)
Fish Weight (g)	1248 (841–1634)	855 (580–1173)	2585 (1994–3586)	929 (352–1348)
Liver Wt (g)	135.2 (98.1–223.3)	24.0 (6.7–58.9)	69.2 (51.6–98.9)	21.4 (9.2–28.8)
HSI (%)[2]	10.8 (7–14)	2.3 (1.2–5.0)	2.83 (1.5–4.3)	2.37 (2.1–2.6)
Percent Lipid	61 (52–71)	24.5 (5–56)	31.4 (13–62)	31.7 (22–36)
Summary Contaminant Concentrations[3,4]				
Total PCBs (ng/g)	321.5 (250.9–407.9)	138.9 (91.7–251.3)	130.4 (74.2–203.1)	1912.5 (1198.8–3017.2)
Total Drins (ng/g)	7.0 (4.8–12.4)	1.8 (0.7–3.4)	2.0 (1.0–5.3)	5.2 (3.8–7.0)
Total Chlordane (ng/g)	28.2 (9.5–50.8)	16.5 (9.7–26.7)	27.9 (14.7–39.5)	95.8 (80.0–123.9)
Total HCH (ng/g)	20.1 (13.7–35.3)	3.4 (1.5–8.7)	7.2 (2.8–15.0)	3.1 (2.3–3.8)
Total DDT (ng/g)	289.8 (220.2–414.5)	974.3 (715.0–1563.5)	53.2 (24.5–102.6)	619.2 (415.4–904.4)
Ratio DDE/Total DDT	0.82 (0.74–1.0)	0.80 (0.76–0.83)	0.72 (0.64–0.80)	0.61 (0.56–0.65)
PCB-TCDD TEQs (pg/g)[5]	13.6 (9.7–18.4)	4.4 (3.1–7.2)	6.6 (4.0–8.7)	25.1 (18.5–34.8)
Total Toxaphene (ng/g)	132.5 (94.0–210.0)	6.9 (1.6–39.0)	105.9 (56.0–180.0)	600.2 (470.0–720.0)
Methyl Mercury (µg/g)	0.01 (0.01–0.02)	0.03 (<0.05–0.07)	0.03 (0.02–0.09)[6]	0.03 (0.02–0.08)
Total Mercury (µg/g)	<0.05	0.04 (<0.05–0.08)	0.04 (<0.05–0.10)[6]	0.04 (<0.05–0.12)
Lipid-Normalized Concentrations[3]				
Total PCBs (ng/g lipid)	532 (363–755)	782 (163–2584)	517 (208–1562)	6140 (3329–13714)
Total Drins (ng/g lipid)	12 (8–19)	10 (6–15)	8 (7–9)	17 (14–21)
Total Chlordane (ng/g lipid)	47 (15–94)	93 (27–267)	111 (64–236)	307 (222–460)
Total HCH (ng/g lipid)	34 (19–68)	19 (15–32)	29 (21–77)	10 (9–10)
Total DDT (ng/g lipid)	480 (317–768)	5,496 (1277–18502)	211 (101–733)	1988 (1154–4111)
Total Toxaphene (ng/g lipid)	222 (149–404)	39 (8–390)	420 (290–786)	1927 (1306–3273)

1. *arithmatic means and range*
2. *hepatosomatic index: liver weight/body weight *100*
3. *geometric means and range*
4. *non-detects set to 1/2 sample detection limit*
5. *toxic equivalent units calculated using dioxin-like PCB TEFs of Ahlborg et al. (1994)*
6. *due to QA problem, one analysis removed: n=4*
7. *summary does not include one aberrant individual*

Table 6. Summary of hydrocarbon data for sediment cores.

COMPOUND ng/g	MOOSE	STUART	CHILKO	KAMLOOPS	NICOLA	HARRISON
Number of samples	18	5	5	8	11	12
Background Total PAH[1]	15 (est)	125 (est)	29 ± 8	35 (est)	46 ± 20	7 ± 2
Highest PAH[2]	**45**	209	38	**245**	**109**	**201**
Background Total Alkane	1155 ±320	7500 (est)	1600 ± 890	3800 (est)	11250 (est)	1790 ±270
Highest Total Alkane[2]	2053	8943	3170	**19300**	20600	4880

1. Background values were calculated as the average of the bottom 3–4 samples for the cores (± 1 sd).
2. Bold numbers identify where a case can be made for significant contamination based on the hydrocarbon profile in the core.

Table 7. Estimated contaminant fluxes to sediments for the six lakes.

METAL OR COMPOUND		MOOSE	STUART	CHILKO	KAMLOOPS	NICOLA	HARRISON
Pb[2]	surface	0.8	0.05	<0.4	0.4	0.1	0.5
	maximum	2.3	0.3	<0.4	3.6	0.3	1.7
Zn[2]	surface	12.5	0.2	<1.2	5.1	0.0	5.6
	maximum	14.3	0.5	<1.5	8.3	1.4	5.6
Cu[2]	surface	5.0	0.03	<0.3	0.1	0.2	3.4
	maximum	5.0	0.5	<0.6	2.4	1.1	3.4
Hg[3]	surface	1.8	0.2	0.0	0.0	n/a[1]	n/a
	maximum	3.8	5.6	<0.8	10.2	n/a	n/a
Total DDT[3]	surface	0.13	<0.04	<0.06	<0.3	0.62	0.05
	maximum	0.6	0.05	0.02	0.6	29	1.6
Total HCH[3]	surface	0.02	0.004	<0.02	<0.09	0.03	0.002
	maximum	0.05	0.004	0.01	0.09	0.54	0.01
Total PCB[3]	surface	0.16	0.02	<0.6	<3	0.14	0.06
	maximum	0.28	0.03	0.06	2.5	0.13	0.27
Total PAH[3]	surface	6.9	3.4	0.0	33	0.41	9.0
	maximum	9.9	3.4	0.6	67	5.7	34.9

1. not available
2. μg/cm²/yr
3. ng/cm²/yr

Moose Lake

This deep and relatively high elevation lake (1,000 m) receives most of its water from sub-alpine forest, permanent snow field, and glacial runoff from the uppermost portion of the Fraser River drainage in the Rocky Mountains. Ice covers the lake from December to mid-April in most years and the water remains fairly cool and turbid throughout the summer because of the glacial meltwater. Sediment trapping efficiency of the lake is probably high, due to both the residence time (greater than one year) and cool summer temperatures which would cause the influent river plume to plunge deep into the lake. Although remote from any settlement, a hydro power-line, a highway and a railway run along one side of the lake and the incoming river. Sedimentation rates determined by ^{210}Pb agree well with laminae counts (Table 2) and with previous work (Desloges 1995), leading to confidence both in the assigned dates and in the assumption that little or no mixing has occurred after deposition.

There is evidence of minor contamination by several heavy metals (Table 3; Pb, Zn, Cu, Ni, Co), the best example of which is Zn (Fig. 2). Some or all of the Pb contamination likely comes from highway automobile traffic (leaded gasoline used until the late 1970s) but the other contaminating metals must have another source, probably linked with the highway/railway corridor (*e.g.* road salt, highway construction). For these

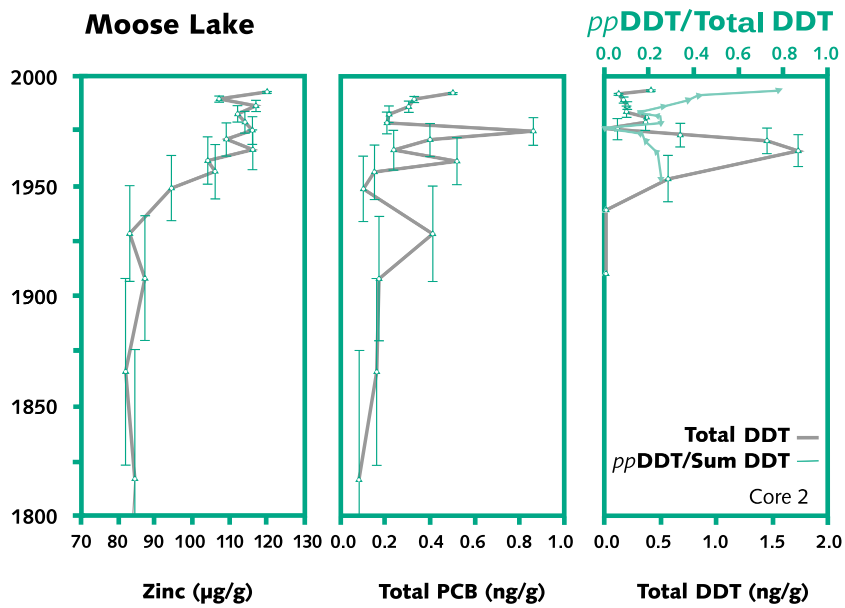

Figure 2. *The concentration of Zn, Total PCBs, Total DDT and the ppDDT/total DDT ratio in Moose Lake sediments as a function of time. Error bars denote 95% confidence intervals.*

metals, the contamination is first evident from about 1945–1950 for both cores. Hg also shows small, "noisy", but detectable contamination in the upper layers of the sediment (after about 1950).

Pesticides are observed at very low levels in sediments (Table 4) with hexachlorocyclohexane (HCH), hexachlorobenzene (HCB) and dichlorodiphenyltrichloroethane (DDT) providing the clearest records for contamination. DDT enters the lake after about 1940 (Fig. 2) and peaks in the late '50s to mid '60s, consistent, within dating and sectioning resolution, with known aerial spraying of DDT in B.C. from 1946 to 1962 (Prebble 1975). For the deeper sections of the core, DDD and DDE comprise 80 per cent of the Total DDT (sum of DDE, DDD and DDT), suggesting substantial weathering (Addison 1978). Sediment concentrations in the upper part of the core show that DDT fluxes markedly decreased after the 1960s. However, the surface sediment is elevated in Total DDT, 80 per cent of it composed of *pp*DDT, suggesting a recent input of relatively unweathered DDT.

Total PCB concentrations in the sediments are exceptionally low (<1 ng/g) compared with other remote lakes where observed values are an order of magnitude, or more, higher (Eisenreich *et al.* 1989; Muir *et al.* 1996; Sanders *et al.* 1994). These low values result in a noisy plot of Total PCB concentration as a function of time (Fig. 2), making it difficult to infer reliable long-term trends or to determine an accurate congener distribution. The low PCB concentrations in sediments may derive partly from a relatively rapid sedimentation rate (*i.e.*, dilution by inorganic particles) and partly from a hydrologic regime which causes some of the PCB (especially the more soluble, lighter components) either to volatilize or simply to pass through the lake (cf. Amituk Lake study in Barrie *et al.* 1997). The total PCB burden accumulated in these sediments since the 1940s, estimated at 20 ng/cm^2 (focus-correction would reduce this number), is, however, similar to inventories measured elsewhere in North America for lakes receiving PCBs via the atmosphere (Rapaport and Eisenreich 1988; Van Metre *et al.* 1997).

While Moose Lake sediments are relatively clean, the fish results are surprising (Table 5). Burbot livers contain elevated concentrations of PCB, total DDT and toxaphene. The levels of PCB were particularly high, averaging 1900 ng/g. This finding is similar to the results of Donald *et al.*(1993) who found that trout muscle tissue from Moose Lake contained the highest PCB levels of 14 continental divide lakes sampled. In several of the burbot collected in the FRAP study, PCB exceeds the 2000 ng/g wet weight human consumption guideline (BC MELP 1998). While the levels of DDT and toxaphene are below the 5000 ng/g wet weight US Food and Drug Administration legal limit for commercial food (US EPA 1989), Germany has recently established a legal tolerance level for toxaphene of 100 ng/g (de Geus *et al.* 1999). Both compounds averaged 600 ng/g of liver tissue in Moose Lake burbot.

On a lipid-weight basis, concentrations of organochlorines (OCs) in Moose Lake are among the highest measured in Canada (cf. Muir *et al.* 1997) although still lower than concentrations in lake trout from Lake Ontario (Kiriluk *et al.* 1995). It is interesting to note that the ratio of Total DDT to PCB is similar in fish from Stuart Lake, which would likely only receive DDT and PCB from the atmosphere (Fig. 3). This fact supports the hypothesis that the source of PCB in Moose is also atmospheric. In addition, the overall pattern of contamination by PCB, toxaphene and DDT in Moose Lake mirror data for top predator fish in other remote lakes receiving atmospheric contaminants (Muir *et al.* 1997; Nakata *et al.* 1995).

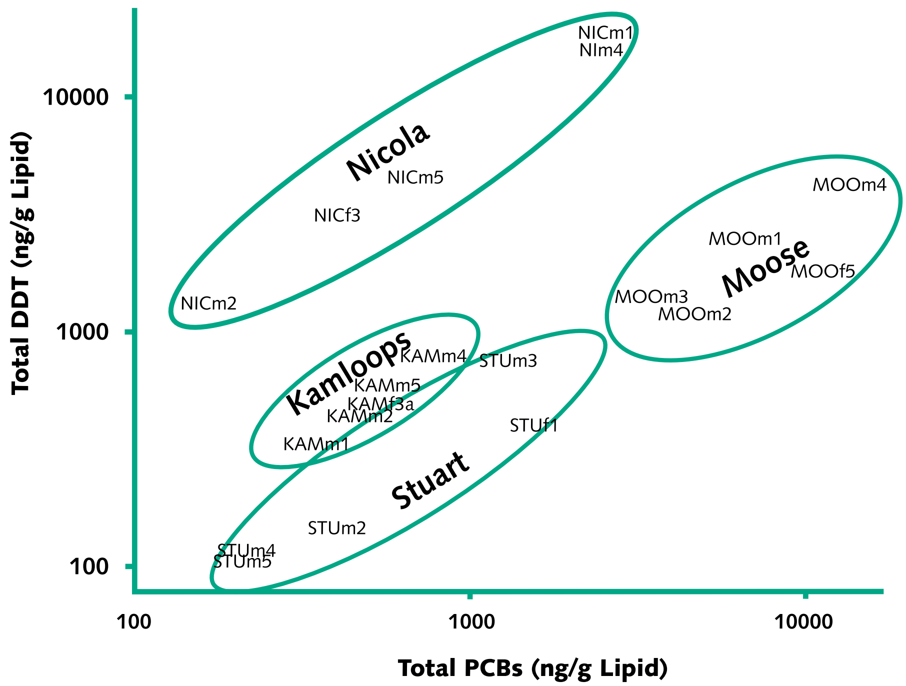

Figure 3. Lipid corrected Total DDT versus Total PCBs in burbot liver.

The DDE/Total DDT ratio in Moose Lake burbot was consistently less than 0.70 (Fig. 4) implying relatively unweathered contamination (Addison 1978; Sanchez *et al.* 1993). We infer from both the sediment and burbot results that a small amount of unweathered DDT has entered Moose Lake either directly from the atmosphere or in meltwater. The DDT could have been recently delivered, but it is also plausible that it derives from older DDT that has been preserved undegraded in the snow and ice of the basin and released during a season of exceptional meltback. Total DDT concentrations exceeding drinking water guidelines (0.03 mg/L) have been found in a permanent snowfield layer deposited in the late 1980s in nearby Banff National Park. Most of this total DDT was composed of the parent-DDT congeners (Donald 1997, pers. comm.).

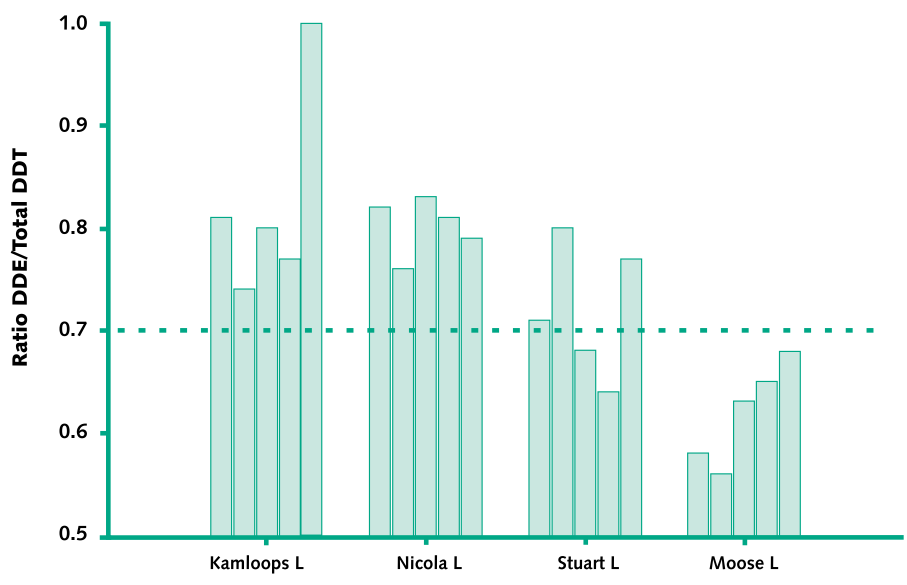

Figure 4. The ratio of DDE to Total DDT in burbot liver for the four lakes sampled for fish.

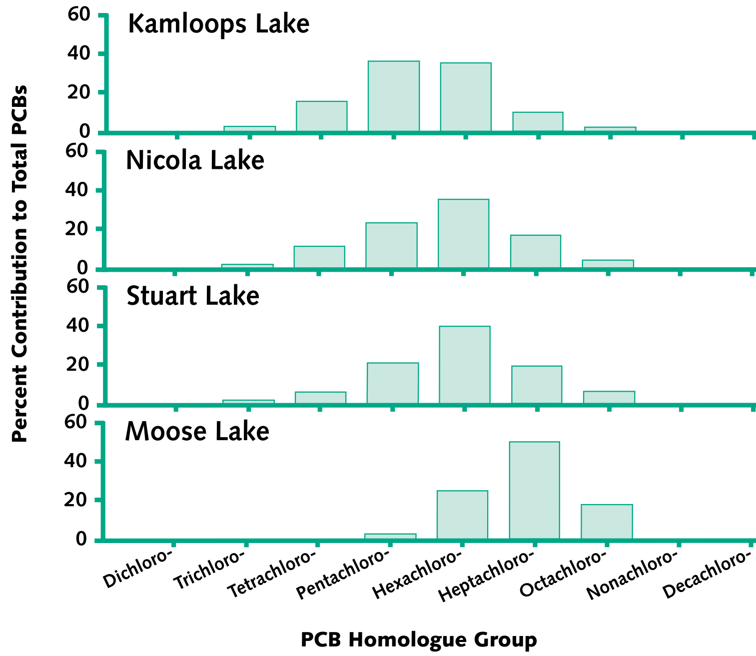

Figure 5*. Patterns in PCB homologue groups in burbot liver for the four lakes sampled.*

It is well known that biomagnification can produce extraordinarily high OC burdens in predatory fish from lakes (Kidd *et al.* 1995a; Kidd *et al.* 1995c; Kiriluk *et al.* 1995; Muir *et al.* 1997; Rasmussen *et al.* 1990). The PCB, toxaphene, DDT and chlordane data for burbot suggest that biomagnification must be at least partly implicated in Moose Lake. The OC data alone indicate that burbot are at the 4th to 5th trophic level and that the Moose Lake food web is longer than that of other lakes studied here or by Donald *et al.* (1993). What remains to be explained, however, is why the Total PCB/toxaphene ratio in Moose Lake burbot is consistently greater than three when in most remote lakes receiving only atmospheric contaminants this ratio is usually <1 (Muir *et al.* 1997), and why heavy congeners tend to dominate the PCB composition (Fig. 5). Where high Total PCB/toxaphene ratios have been observed elsewhere (Lake Laberge, Char Lake), local contamination by PCB was inferred—although not proven conclusively. The sediment record in Moose Lake appears to discount a local PCB spill and we must look elsewhere for an explanation. Perhaps unique physical and chemical processes are responsible for the transfer of these two contaminants to the lake and their apparent absence in sediments.

Due to its low Henry Law constant, toxaphene partitions strongly into water (Barrie *et al.* 1997). Therefore, we can expect the toxaphene concentrations in the lake to reflect concentrations in the incoming river modified by air-water exchange at the lake surface dependent on water temperature and the rate of mixing of deeper waters to the surface. In this scenario, the basin simply supplies cold, toxaphene-saturated water to the lake.

In contrast to toxaphene, PCBs, especially the heavier congeners, partition strongly to particles. Continuing with the rationale, the lighter congeners which are more volatile and less particle-attracted could be lost to the atmosphere during the annual cycle of partial snowfield melt and re-freezing while the heavier congeners become associated with suspended sediments in the runoff (Barrie *et al.* 1997; Gregor 1990). The loss of lighter PCBs (penta-, tetra-, tri-chlorinated biphenyls) from the congener profile in Moose Lake burbot (Fig. 5) could thus be explained. Certainly, the congener pattern to the right of CB136 (*i.e.*, PCBs with 6 or more chlorine substitutions) is very similar for all four lakes. This suggests a common, basin-scale (*i.e.*, atmospheric) source which has been altered in the case of Moose Lake by the partial loss of lighter components. In addition, both toxaphene and PCB concentrations in meltwater could be enhanced by sublimation of ice or snow, especially at higher altitudes or even by permanent snowfield and glacier ablation, releasing contaminants deposited at an earlier time when atmospheric deposition was greater.

These concentrating processes have to be more efficient with heavy PCBs than toxaphene to explain the higher ratio in burbot tissue. The lack of accumulation in sediments, particularly of PCBs, is also puzzling.

The hypothesis that physical and biological processes, rather than local contamination, lead to the OC patterns observed in Moose Lake burbot needs to be tested directly in comparative food-web studies.

Stuart Lake

This large and relatively shallow (mean depth 26 m) lake is the last in a chain of large lakes on the Nechako Plateau which together supply most of the flow to the Stuart River, which subsequently joins the Nechako River more than 60 km above its confluence with the Fraser at Prince George. The residence time of water in Stuart Lake is about 2.3 years and sediment input is very low, partly because most of the incoming rivers originate in large lakes and partly because local topography is not mountainous. As a result, the sedimentation rate is very low, and sediment composition is dominated by organic matter. The lake is usually ice-covered from mid-December to mid-April. Fort St. James (pop. ~5,000), by the lake near the outlet, has a lumber mill as do several smaller aboriginal communities in the basin. Treated sewage effluent is discharged to the river downstream of the lake. The Pinchi Fault, running along the northeast side of the lake, is known to contain abundant Hg mineralizations, some of which have been redistributed during glaciation (Plouffe 1995). The only known local contaminant source is a mercury mine and reduction plant, which operated during the 1940s and early 1970s, located near Pinchi Lake which drains into Stuart Lake (EVS 1996).

The background concentrations of PAH in lake sediments are relatively high in Stuart Lake (Table 6). High organic carbon content probably contributes to natural enrichment, and slow sedimentation rates imply a lack of diluting inorganic material as demonstrated by the relatively low and constant PAH fluxes (Table 7).

Most of the heavy metals in Stuart lake show no or only very slight trends with time. For example, there are minor enrichments of Pb observed between about 1940 to 1980 which can be attributed to the general use of leaded gasoline. An exception, however, is Hg which provides a very clear contaminant signal in the sediments (Fig. 6). Contaminant Hg entered the sediments in the 1940s and then decreased from 1950 to 1980 so that it has now returned almost to the original background. The appearance of Hg in Stuart Lake sediments coincides with the operation of the Hg mine near Pinchi Lake from 1940 to 1944, during which time the mine disposed of tailings into Pinchi Lake (EVS 1996). The mine reopened and operated from 1968 to 1975, but this later operation is not evident in the Stuart Lake sediment profiles, probably due to better environmental controls (EVS 1996). Integrating the contaminant Hg burden for the lake sediments at the two sites and prorating it for the sedimentary basin implies that as much as 100–200 kg of Hg entered the lake sediments during the war years—or about 0.01 per cent of the production.

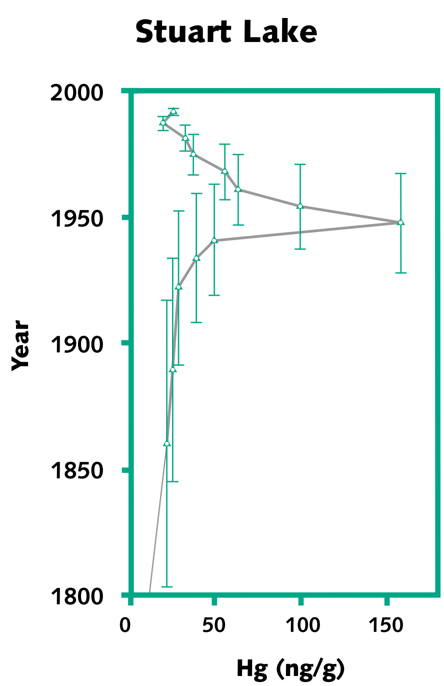

Figure 6. Hg in Stuart Lake sediments as a function of time (at S-1).

Organochlorine determinations were made for only five core sections. These few data points reveal low sediment concentrations for pesticides and PCBs (Table 4), similar to observations from Moose Lake and consistent with long-range transport to the basin occurring predominantly after 1950. For example, the depth profile observed at S-1 (4 points; not shown) suggests that Total DDT was highest in the 1960s, in agreement with the documented use of DDT in aerial spraying (Prebble 1975), and consisted predominantly of weathered components (DDD, DDE). Deeper sections contain no detectable organochlorines, consistent with their age (mid-1930s or earlier).

Burbot collected from Stuart Lake were longer (average 78 cm), heavier (average 2.3 kg) and older (average 9.3 years) than fish collected in the other three lakes. These factors have all been shown to correlate with higher concentrations of persistent OCs and Hg (Kidd *et al*. 1995b; Muir *et al*. 1997) and, therefore, fish from this lake might be expected to display relatively high pesticide and Hg levels. This was not the case, however, and concentrations of most OCs were in the lower end of the range of values measured at the other three lakes, particularly for DDT (Table 5). Organochlorine concentrations in burbot are entirely consistent with long-range atmospheric delivery to the lake watershed.

Despite gross Hg contamination of Pinchi Lake in the 1940s (Ableson and Gustavson 1979), and a fish advisory recommending limited consumption of lake trout from Pinchi Lake (Martin *et al*. 1995), there is no evidence of elevated Hg levels in burbot recently caught from Stuart Lake. This is not unexpected as Hg concentrations in sediments have declined to pre-1940 levels.

Chilko Lake

This remote and extremely deep (366 m) lake is nestled in the heavily glaciated Coast Range Mountains at over 1,100 m elevation. While the drainage basin is only 30 per cent larger than Moose Lake's, the water yield is 120 per cent larger. This is possibly due to lower evaporation/sublimation around Chilko as the annual precipitation in both areas is in the same range (600–1,200 mm [ESWG 1995]). Because the residence time is so long (17 yrs), the lake is probably a very effective sediment trap.

Unfortunately, the sediment core evidence suggests that the steep bathymetry exposes the profundal sediment basins to slumps of accumulated slope sediment which then disturb the stratigraphy. The lake is often very clear (Secchi depth ~40 m) in the early summer before glacier melting occurs and even then turbidity only affects the inlet area (Shortreed 1997, pers. comm.). The sedimentation rates were difficult to determine with precision due to disturbances in the sediment. Nevertheless, sediments show little evidence of metal trends with depth that could be interpreted as contamination. Similarly, there were only a few organochlorine compounds detected, and where they were detected, concentrations were uniformly low (Table 4). This suggests long-range atmospheric inputs only. Perhaps, in contrast to the situation in the Rockies, much of the annual contaminant deposition in the Coast Range drainage basin is not released in the annual summer melting, but is instead stored in snowfields and glaciers. Of course, Moose Lake had relatively low concentrations of OCs in its sediments in contrast to the high concentrations in burbot and, therefore, top predator fish (*e.g.* bull trout) should be sampled in Chilko Lake to see whether a similar condition exists there.

Kamloops Lake

Kamloops Lake is a riverine lake, situated downstream of the confluence of the North Thompson and South Thompson rivers, with the largest drainage basin (40,000 km^2) of the six lakes and is not a headwater system. While its residence time is extremely short (8 weeks), Kamloops Lake is deep and the physical limnology is conducive to sediment trapping so that laminae are easily visible in cores (Pharo and Carmack 1979). The major sediment source is post-glacial lacustrine sediment banks along the lower reaches of the two rivers which become mobilized during high water and flows of the freshet (May–July). As there are many large lakes upstream in the drainage basin to trap sediments and their associated atmospherically derived contaminants, Kamloops Lake probably receives contaminants predominantly from the City of Kamloops. The lake does not freeze in most winters and continually cools with wind mixing to temperatures below 4°C (St. John *et al*. 1976).

While the largest industrial discharge today is the pulp mill (Hatfield Consultants 1995), in the past there was a small copper and rare-earth metal smelter (Bernard 1983) and a petroleum refinery. The municipal sewage treatment plant (serving 90,000 people) has been discharging a tertiary-treated effluent during the April to September period and has not been discharging during the fall/winter since the late 1970s. Two railways, a hydro-power corridor, and the Trans-Canada Highway along the shore and the upstream rivers represent potential contaminant sources.

For most of the metals there is little evidence of sediment contamination. Pb may show slight enrichments after about 1950 similar to the other lakes and marine coastal regions of B.C. (*e.g.* see Macdonald *et al.* 1991) and Hg also shows a slight, but significant, enrichment in the upper portion of the sediment (Table 3) which could be related to use at the pulp mill as a slimicide (Waldichuk 1983).

Previous sampling in Kamloops Lake and experience elsewhere in B.C. led us to expect that polychlorinated dioxins and furans (PCDD/Fs) produced by pulp mills using chorine bleach and/or pentachlorophenol (PCP) contaminated wood chips would be an important component of the OC burden in Kamloops Lake sediments (Macdonald *et al.* 1992; Trudel 1991; Mah *et al.* 1989). As anticipated, PCDD/Fs from pulp mill activities proved to be the dominant regional OC class, and a detailed analysis of their history and sources as recorded by the sediments is discussed in Macdonald *et al.* (1998).

Principal Component Analysis (PCA) of the OC data distinguished three major groups of PCDD/Fs: (I) lightly chlorinated compounds produced when chlorine was used to bleach pulp, (II) pulp mill digester-mediated condensation products of polychlorinated phenoxyphenol contaminants in polychlorophenates (polyCPs), and (III) highly chlorinated products of combustion and/or PCP contamination. In addition to the PCDD/Fs, a fourth group was identified consisting of most of the PCBs and possibly octachlorodibenzo-*p*-dioxin (OCDD) (Group IV). When the sediment core samples are plotted on a PCA biplot (Fig. 7) they show that sediments were contaminated first by atmospheric inputs of low levels of PCB and higher chlorinated dioxins (Group IV). We infer that prior to the construction of the pulp mill, Kamloops Lake was similar to the other lakes in this study, receiving OCs primarily from the atmosphere. After construction of the mill, sediments were contaminated first by condensation products (a sign that chlorophenol-contaminated wood was used in the pulping process) and then by the pulp chlorination products (*e.g.* TCDD/F). The pulp mill stopped using CP-contaminated wood stock and phased out chlorine bleach with the result that recent sediments show a dramatic downturn in these compounds.

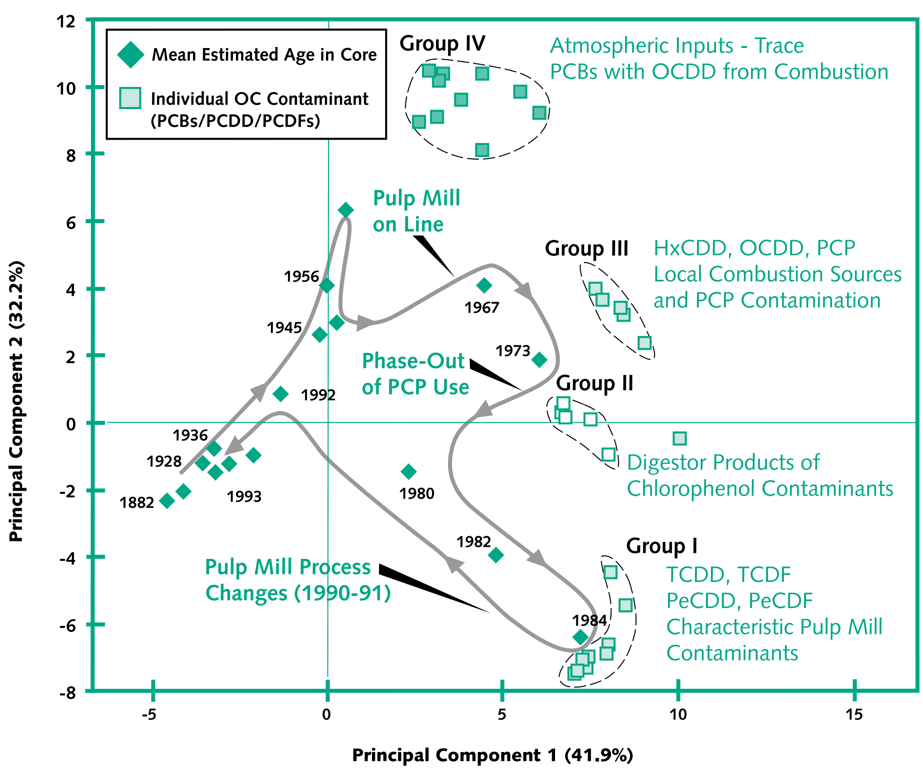

Figure 7. PCA biplot showing the association between the contaminants in sediment cores from Kamloops Lake and particular organochlorine chemical groups (PCDD/F, chlorophenolics and PCBs) and their relationship to major contaminant sources. Dates refer to ^{210}Pb estimates for core sections, which are linked in date order by the arrow line. The group designation and composition was determined from cluster analysis.

The predominant historical source of TEQ (dioxin toxicity equivalent units) to the lake sediments has been from TCDD/Fs produced when bleaching pulp with chlorine (Fig. 8; TEQs based on Toxic Equivalent Factors proposed by NATO[1988]). Although the production of these compounds was sharply curtailed in about 1990, they continue to be available for uptake in biota (Macdonald *et al.* 1998) and 2,3,7,8 TCDF was easily detected in peamouth chub and mountain whitefish sampled in 1994 (Raymond *et al.* 1999). Two other significant results emerged from the PCDD/F and PCB determinations: (1) the pulp chlorination process produced several PCBs (CBs 13, 15, 37) and (2) a secondary source of PCDD/Fs, witnessed as a pulse in the late 1940s to early 1950s, probably entered the sediments from lake-side treatment of power poles or railway ties with contaminated PCP (Fig. 9).

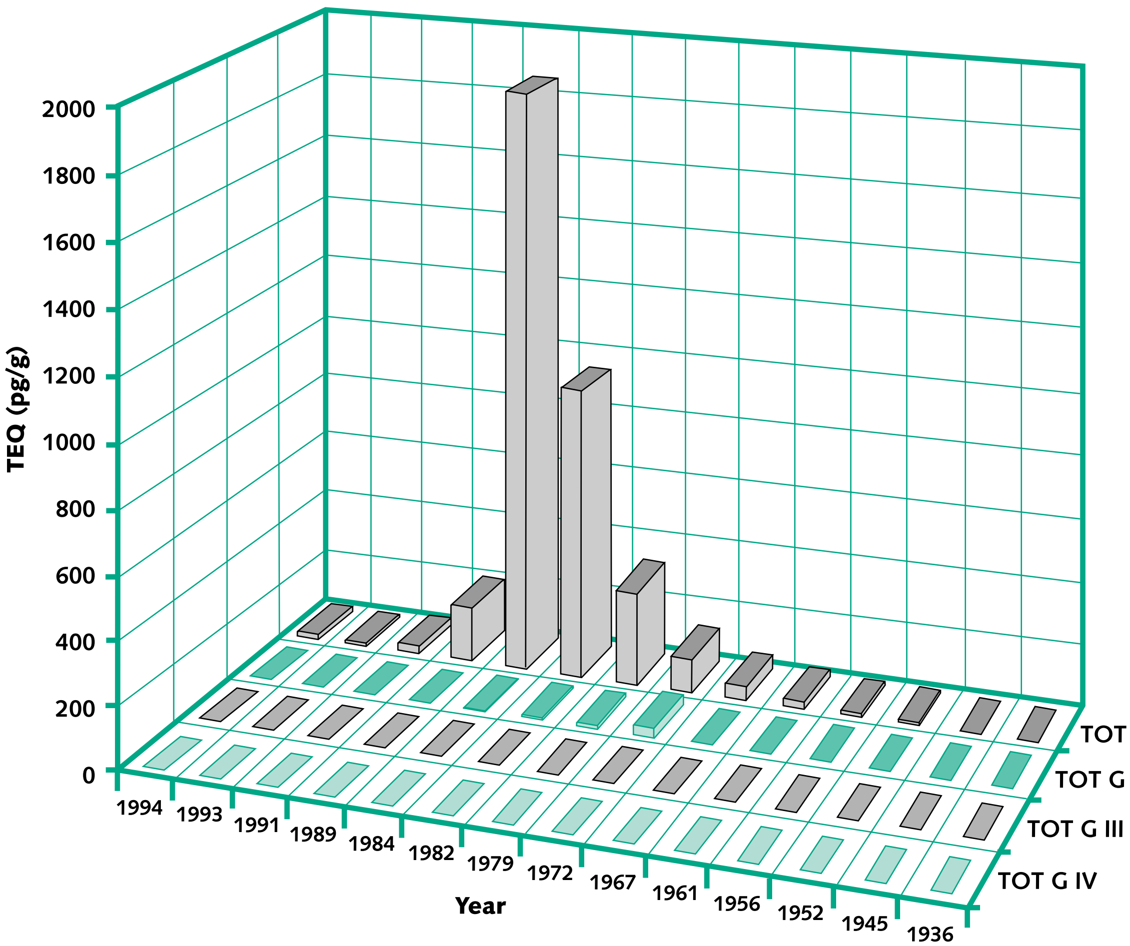

Figure 8. A plot of TEQ for the PCDD/F congener groups in Kamloops Lake sediments as a function of depth.

Other organochlorines, for example the pesticides, are found only at low or undetectable concentrations in sediments, comparable with the other lakes and consistent with large-scale atmospheric sources (Table 4). Dieldrin, endrin and DDT were detected in some of the core sections.

There is strong evidence of PAH contamination in Kamloops Lake where background concentrations of total PAH of 35 ng/g increase to 100–250 ng/g in the upper layers of the core (Table 6). Contaminant PAH could derive from a variety of sources including beehive burners, hogfuel burning, inputs of petroleum from the oil refinery and vehicles and from the use of various fuels including coal and oil. Entry of PAH into the lake could occur both in runoff and in atmospheric deposition.

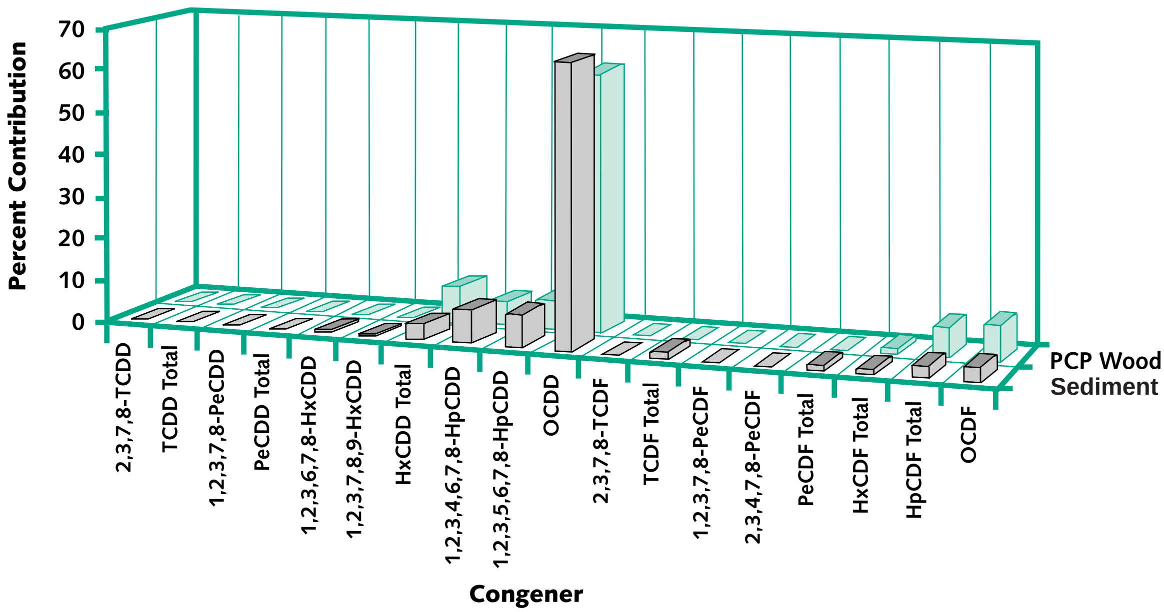

Figure 9. A comparison of the PCDD/F composition of the 20–25 cm sediment sample from Kamloops Lake with that of wood contaminated by PCP as reported in Van Oostdam and Ward (1995).

Despite the potential for local municipal, industrial and agricultural sources of contaminants to Kamloops Lake, concentrations of the measured organochlorines and Hg in burbot liver are not exceptional when compared to the other lakes. Burbot from Kamloops Lake were, however, unusual in having both a liver lipid content two to three times greater and hepatosomatic indices (HSI) which were three times greater than those found in fish from the other lakes (Table 5). Relatively elevated liver lipid content (though not HSI) was also found in peamouth chub in the vicinity of Kamloops Lake (Raymond *et al.* 1999), and has been attributed to the relatively high productivity of both the lake and Thompson River. However, liver enlargement and high lipid content are also common responses to contaminant exposure (Goede and Barton 1990; Oikari and Nakari 1982; Poels *et al.* 1980). Should these biological effects have been induced by toxic substances, then it likely involves contaminants other than those listed in Table 5, which show Kamloops burbot generally to be near the low end of concentration ranges.

Nicola Lake

Nicola is a warm, relatively shallow (mean depth 24 m), productive lake on the Thompson-Okanagan Plateau. Surrounding the lake are grasslands and hay fields used for cattle ranching, which grade into sub-alpine forest at the higher elevations. On the headwater streams entering Nicola Lake are other lakes which intercept the higher elevation inputs. There are no permanent snowfields or glaciers in the drainage basin, nor does the lake freeze over consistently in winter. Although the watershed population is sparse, atmospheric emissions from a major highway and the downstream City of Merritt (pop.~10,000), with its several sawmills, could contribute contaminants to the watershed.

With the exception of Pb, the sediments show little evidence of metal contamination. The minor Pb contamination (1–3 mg/g) could be supplied by local traffic which used leaded gasoline until the late 1970s.

Nicola Lake sediments exhibit gross contamination by DDT (Fig. 10) and minor contamination by chlordane and HCH (Table 4). At maximum concentration, the Total DDT in the sediments exceeds that of the other lakes by at least two orders of magnitude. The DDT is partly degraded, consisting of about 40 per cent DDT with the rest contributed by DDD and DDE. The dating shows that DDT entered the lake during the late 1970s and mid 1980s—significantly later than the known use of DDT for aerial spraying (Prebble 1975) and the record of that spraying as found in Harrison Lake sediments (Fig. 11). Based on lake area, and assuming that accumulating sediments occupy half the lake, the inventory of Total DDT for Nicola Lake is roughly 50 kg. The amount and timing of DDT accumulation in the sediments suggests strongly that local applications have been heavy and that some applications may have occurred after the ban on its use in 1972.

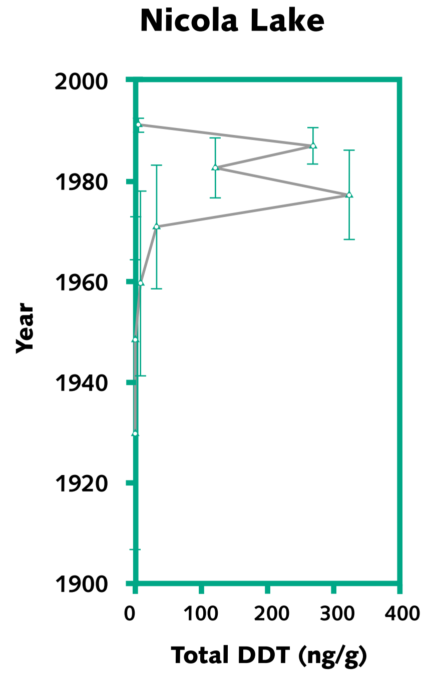

__Figure 10.__ Total DDT concentration in Nicola Lake sediments as a function of time.

The remarkable Total DDT concentrations in burbot liver (one sample contained 18,000 ng/g lipid, Table 5) suggest that some of the DDT released over a decade ago continues to be available for uptake in biota. The absence of corresponding high values of other organochlorines, specifically toxaphene which is actually lowest in Nicola Lake, leaves little doubt that these high Total DDT levels in burbot are produced by local contamination. Figure 3 also dramatizes the order of magnitude increase in the DDT/PCB ratio seen in Nicola Lake relative to the lakes receiving only atmospheric input. Comparison of the sediment and fish data with results from the other lakes suggests that over 90 per cent of the DDT in Nicola Lake must derive from local sources. This conclusion is supported by the large Total DDT inventory of 350 ng/cm^2, which is similar to that observed in Lake Ontario sediments, where DDT has entered from inflowing rivers (Wong *et al.* 1995), and much more than the 2.6–21.6 ng/cm^2 observed elsewhere in North America for atmospheric fallout (Rapaport and Eisenreich 1988). The prevalence of weathered products in fish (DDE/Total DDT >0.7, Fig. 4) and sediments ([DDD+DDE]/Total DDT >0.6) suggests that recycling of weathered DDT is occurring rather than recent input of fresh DDT.

It is surprising that the level of toxaphene is an order of magnitude lower than in the other lakes because some small lakes in the Nicola Lake drainage basin have been treated with toxaphene to kill undesired fish prior to stocking with trout (Stringer and McMynn 1960; Nordin 1997).

Harrison Lake

Harrison is a very deep (270 m) oligotrophic lake occupying a major valley in the Coast Range. It receives meltwater during summer from the extensive snowfields and glaciers and runoff from heavy rainfall at lower elevations during winter. While a significant portion of the sediment in the glacial runoff is trapped immediately upstream in Lillooet Lake (Gilbert 1973), sedimentation in Harrison remains substantial (Table 2). The lake, which is only 10 m above sea level, rarely freezes.

The drainage basin is relatively undeveloped, except for logging around the lake and some agriculture (north of Lillooet Lake). Atmospheric transport of contaminants from the urban and intensive agricultural areas of the lower Fraser Valley is likely (B. Thomson 1997, pers. comm.), and there has been a sawmill located at the outlet since the first settlement in the late 1800s.

Harrison Lake shows evidence of metal contamination by Pb and perhaps also Cr and Ni. Contamination by organochlorines is detectable, but at low concentrations consistent with an atmospheric source (Fig. 11). PAHs (Fig. 11) increase above background as early as 1890, reaching a maximum during the 1950s. Thereafter, levels decrease to the present time, although current surface concentrations still exceed those deep in the core. The source of the contaminant PAH is not clear and will require further detailed analysis of the compound patterns. Likely sources include local slash and beehive burning as well as a larger-scale input from urban and industrial areas of the lower Fraser Valley. The substitution of coal by oil in the 1950s has been suggested as the reason why peak PAH fluxes are often observed in sediment horizons dating from that period (*e.g.* see Gschwend and Hites 1981; Macdonald and Crecelius 1994).

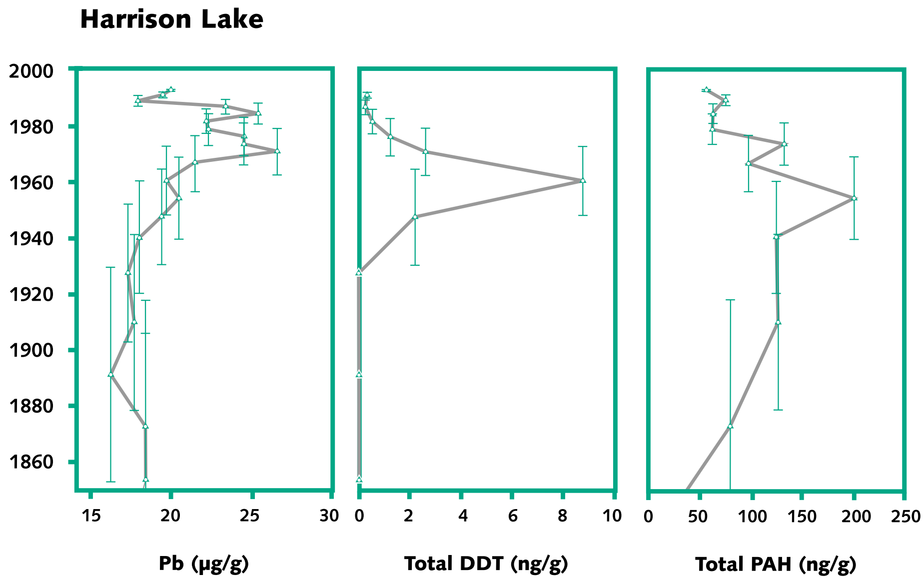

Figure 11*. Pb, Total DDT and total PAH concentration in Harrison Lake sediments as a function of time.*

SYNTHESIS

The data presented here suggest that four factors influence the contaminant concentrations observed in sediments and fish of lakes in the Fraser Basin: (1) long-range atmospheric transport of a variety of compounds (*e.g.* PCBs, OCDD, PAH, Pb, pesticides); (2) local contamination from activities within a lake's drainage (*e.g.* DDT, Hg, some metals, PAH, PCDD/Fs); (3) the physical setting of the lake (*e.g.* climate, drainage basin characteristics, lake hydrology); and (4) the food-web structure of the lake. The broad design of this reconnaissance study is not ideally suited to determining exactly how a given contaminant is delivered, altered during transport, and subsequently biomagnified and/or sequestered to sediments in each lake setting. Nevertheless, because we have measured a number of organochlorine contaminants in both sediments and the top-predator fish that concentrate them, we can infer how these factors interact and which may be most influential.

Metals

The surface layers of lake sediment reveal minor contamination as enrichments. Virtually all of the lake cores exhibit Pb contamination, usually after about 1950, as shown for Harrison Lake (Fig. 11). Compared to Kootenay Lake where local mining activities produced gross contamination (Macdonald *et al.* 1994; Rieberger 1992), the Pb enrichments are small (Table 3). Contamination is consistent with a widely distributed atmospheric source augmented by local emissions which can be ascribed predominantly to the once-common use of leaded gasoline. At their maximum (during the 1970s) these Pb fluxes (Table 7) fall within the deposition fluxes for "rural U.S. remote" (1.5 mg/cm^2/yr) to "rural U.S. typical" (4 mg/cm^2/yr) (Patterson and Settle 1987) and are less than those observed for the Strait of Georgia (6.4 ± 2.2 mg/cm^2/yr) (Macdonald *et al.* 1991). Recently, Pb fluxes have been decreasing in response to the ban of leaded gasoline in the late 1970s. It is noteworthy that the largest Pb fluxes were observed in Kamloops, Moose and Harrison lakes, each of which has either a highway or a large urban centre as an obvious local source.

Of all the metals, Hg is of greatest concern due to its toxicity and potential to bioaccumulate. Stuart Lake, which received Hg contamination from a mine above it on Pinchi Lake, shows the clearest contaminant Hg signal, but Moose and Kamloops lakes also show minor Hg enrichments in recent sediments. Measurements of Hg in burbot liver, however, show no obvious biomagnification, suggesting methyl mercury production in these sediments is minimal.

Polycyclic Aromatic Hydrocarbons (PAHs) and Alkanes

PAHs in lake sediments derive from petrogenic (petroleum, diagenetic) or pyrogenic (combustion) sources which can, in principal, be distinguished by the pattern of PAH compounds (Yunker and Macdonald 1995). Note, however, that both petrogenic and combustion PAHs can be produced naturally or anthropogenically and the detailed pattern of PAH compounds must be examined to infer probable source. Parent PAHs (*i.e.* unsubstituted rings) tend to dominate the homologue series in the sediments collected from the six lakes implying that combustion is probably the most important PAH source (*e.g.* forest fires and burning of coal and oil). Kamloops, Harrison and Moose lakes show the clearest evidence of contaminant PAH (Table 7) and it is significant that these same lakes had the clearest Pb signal in their sediment. Both the Pb and PAH contamination can be explained by nearby sources (either a transport corridor or large urban centre). In particular, Kamloops Lake continues to receive a significant amount of contaminant PAH.

Alkane hydrocarbons often show marked enrichments toward the surface of sediment cores (Table 6) with prominent components of the alkane envelope being nC17 and the odd carbon alkanes between nC23 and nC33 (Yunker and Macdonald 1998). Although inputs of petroleum could enhance alkane concentrations in recent lake sediments, it is far more likely that the observed alkane enrichments derive naturally from lake algae (nC17) and terrestrial plant waxes (odd carbon nC23–nC33) (cf. Eglinton and Hamilton 1967; Meyers and Ishiwatari 1993). Therefore, the enrichment of alkanes in surface sediments of the lakes can be understood as predominantly an input of natural alkanes which are subsequently metabolized as the sediment ages. Kamloops Lake sediments have an additional alkane envelope between nC17 and nC23, possibly indicating petroleum input (Yunker and Macdonald 1998). However, a much more detailed analysis will be required to estimate the strength and pattern of this source.

Organochlorine Compounds

Most of the recent lake sediments (*i.e.* surface samples) exhibit a low background of OC contaminants. When compared with other locations reported in the literature (Table 8), the inventories and fluxes of PCB and DDT are consistent with a delivery by the atmosphere either globally or regionally, with the notable exceptions of DDT in Nicola Lake and PCB in Kamloops Lake. In both cases, local inputs are clearly responsible. DDT is often at a maximum in the 1960s sediment horizons reflecting use patterns in B.C. (Figs. 2, 10). Toxaphene was not detected in any sediment sample (detection limits varied from <0.2 to 2 ng/g), again consistent with low fluxes supported only by atmospheric deliveries (Pearson *et al.* 1997).

Table 8. *Surface concentrations, fluxes and inventories for Total PCB and Total DDT for sites in North America receiving predominantly atmospheric organochlorines.*

		TOTAL PCB			TOTAL DDT		
DATE	LOCATION	SURF. CONC ng/g	SURF. FLUX ng/cm²/yr	INVENTORY ng/cm²	SURF. CONC ng/g	SURF. FLUX ng/cm²/yr	INVENTORY ng/cm²
1993-1994	Fraser Lakes[1]	0.2–1.0	0.02–<3	<8–36	0–6.9	0.04–0.62	<2–350
1990	Arctic Lakes[2]	2.4–39	0.01–0.43	1.9–18			
1981	Lake Ontario[3]	250	12–18	500-600	50	1.9	250–500
1992	Wisconsin Lakes[4]	2.6–89	0.19–0.76	1.3–8.8			
1986, 90	Lake Ontario[5]	7–17.6	0.10–0.52	4.4–28			
Post 1980	Peat Cores,[6] Eastern NA		0.01–0.4	2.9–15.9		0.04–0.15	2.6–21.6
Post 1980	Lake Superior[7]			0.1–8			10.4–27

References
1. *This study*
2. *Muir* et al. *1996*
3. *Eisenreich* et al. *1989*
4. *Swackhamer and Armstrong 1986*
5. *Jeremiason* et al. *1994*
6. *Rapaport and Eisenreich 1988*
7. *Eisenreich 1987*

As observed elsewhere (Muir *et al*. 1997), three particular OCs are always found at high concentration in predator fish—toxaphene, PCB and DDT. Much of the lake-to-lake variation in fish OC concentration, therefore, probably derives from the physical and biological characteristics of each lake rather than variation in sources. In Moose Lake there must be an exceptionally strong partitioning into biota because neither sediment concentrations nor fluxes of PCBs, for example, are high relative to Kamloops Lake, downstream of an industrial city (~0.2 vs. ~3 ng/cm^2/yr). This observation points to a more effective food chain bioconcentration process in Moose as opposed to the other lakes and we conclude, therefore, that Moose Lake is the most vulnerable of the lakes to long-range contaminants, either by virtue of its basin characteristics (seasonally frozen and alpine) or its food-web structure, or both. Out of a total of 17 lakes studied here and by Donald *et al*. (1993) for burbot and/or trout, the levels in Moose Lake are highest. The PCBs in Moose Lake burbot have not been linked to a local source and may simply reflect the accumulation of many years of atmospheric deposition in the snowfields which has not drained annually to the main valley-bottom river. Years of extreme meltback may subsequently transfer this "archived" PCB (and DDT) to the lake and river system. A similar process may be occurring in Chilko Lake but top predator fish were not obtained for study. The recent discovery that OC accumulation in snow in the Rocky and Coast mountains increases with elevation (Blais *et al*. 1998) also suggests that alpine lakes will receive a higher loading.

Not all OC contaminants come from distant points. A large amount of DDT entered Nicola Lake arriving during the 1980s from a local source, and weathered DDT compounds continue to be available for uptake in the lake's biota. In a second example, PCDD/Fs and PCBs entered Kamloops Lake from a pulp mill between 1965 and 1990, but these have been dramatically reduced through process changes at the mill. Nevertheless, the levels of PCDD/Fs in peamouth chub and mountain whitefish from the Fraser Basin in 1994 were highest in the Thompson River sub-basin (Raymond *et al*. 1999) and burbot livers were twice as fat as those collected from the other lakes which can be an indicator of significant contaminant exposure.

CONCLUSIONS AND RECOMMENDATIONS

The continued presence of PCBs, toxaphene and DDT residues in fish long after the chemicals have been banned in Canada and the United States is a concern on two fronts. First, the detections indicate a potential problem to both wildlife and human consumers of these fish. Even the lowest burbot liver levels found in the Fraser Basin lakes exceed Canadian Tissue Residue Guidelines for the Protection of Wildlife Consumers of Aquatic Biota (0.79 pg TEQ/g for total PCB [values in Table 5 are based on Toxic Equivalent Factors from Ahlborg *et al*. 1994]; 6.3 ng/g for toxaphene; and 14.0 ng/g for DDT [Environment Canada 1999]). The only samples to come close to these guidelines are those from Nicola Lake for toxaphene, but they exceed the DDT guideline by the largest margin. The relevance of guidelines being exceeded in one tissue of one fish species needs to be considered. For one thing, as these compounds are lipophilic and burbot liver is lipid-rich in comparison to their muscle tissue, concentrations based on whole fish would be much lower. For another, the level of contamination in other fish species that are more likely to be eaten by other predator fish or osprey (which feed mostly on shallow water species like suckers) is not known.

The assessed potential health risk to human consumers of this species from these lakes can only be assessed if data are available on the consumption habits of people regularly eating burbot, especially the livers. The guidelines mentioned earlier are for food consumed regularly and are difficult to apply in this instance where consumption is sporadic or unknown.

The second issue addressed by this study is the trend in atmospheric loading of persistent organic chemicals and metals to the Fraser's headwaters. In most cases, sediment cores indicate that loading of DDT and PCBs to the lakes has declined dramatically from peak loading rates in the 1970s. The data from Moose Lake (Fig. 2), on the other hand, suggest that they may now be increasing from the rates seen during the

1980s. Whether this is in response to releases from glacier and snowfield reservoirs of these chemicals or an increase in atmospheric deposition cannot be ascertained from this survey. (Unfortunately the detection limit for toxaphene in sediments was not low enough to obtain historical or present trends in its loading.) The loading of PAHs has declined as seen in the Harrison Lake core (Fig. 11) and their present sedimentary concentrations are well below guidelines. With respect to metal loading, lead, in particular, has declined in response to the introduction of unleaded gasoline. Loadings of other metals, while slightly elevated from their pristine rates, are not of concern.

Two studies are recommended:

- The data base on contaminants in top predator fish in B.C. lakes should be expanded to determine if there are other lakes, like Moose Lake, that exhibit inordinately high concentrations of PCB, toxaphene and DDT. Candidates for a survey should include a variety of lakes with differing hydrological conditions and a diversity of trophic structures. With a sufficient number of lakes, it should be possible to infer the particular lake and drainage basin characteristics that lead to high concentrations in top predators. This predictive capability can then be used to select remote lakes for future fish contaminant surveys.
- The importance of trophic structure versus contaminant loading characteristics to the level of contamination in the food web should be examined in Moose Lake, along with other selected lakes identified in the survey suggested above.

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3.2

WATER QUALITY IN THE FRASER RIVER BASIN

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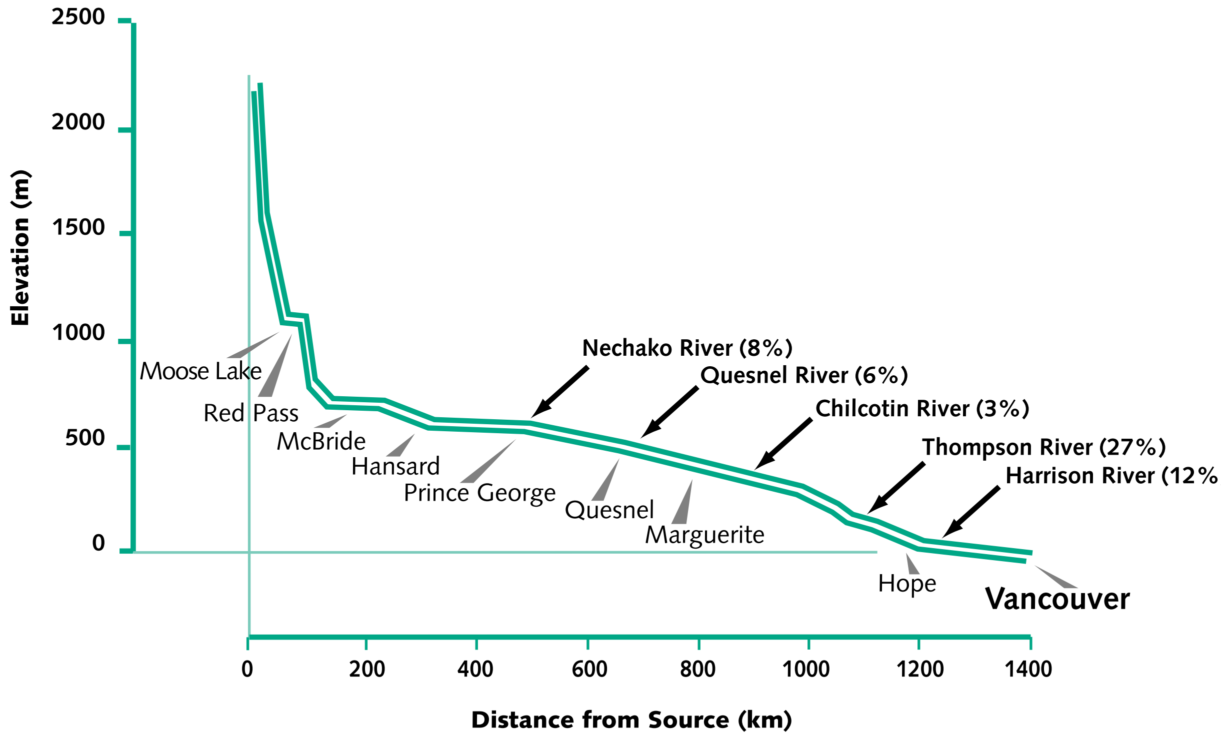
Evaluation and study of water quality is fundamental to protection of both the aquatic ecosystem and human use of the Fraser River and its tributaries. Under the Fraser River Action Plan, three aspects of water quality in the Fraser River Basin were studied: an analysis of long-term trends; measurements of industry-specific contaminants related to major discharges in the upper basin and evaluation of water quality in the Fraser River Estuary area.

BACKGROUND

From the headwaters in the alpine glaciers of the Rocky Mountains above Moose Lake, the Fraser River drains north and west collecting drainage from the northern Rockies and the Cariboo Mountains. Between the first major population centre at Prince George and the town of Hope roughly 750 km downstream, the Fraser River is joined by its major tributaries—the Nechako, Quesnel, Chilcotin and Thompson rivers. These four tributaries constitute, on average, about 44 per cent of the mean annual flow at the river mouth (Figure 1).

Downstream of Hope, the Fraser enters the fertile Fraser Valley, a reach of relatively low gradient and high human activity, including extensive agriculture (cropping and livestock), dense residential housing and diverse industries.

The Fraser River tends to be turbid. In the distant headwaters, suspended sediment is supplied as a combination of glacial flour and insoluble silts and clays from bedrock dissolution. In passage through Moose Lake, these suspended particles are lost in the only major sedimentation basin in the entire river (Desloges and Gilbert 1995). Downstream, the river gradually reacquires its muddy appearance through erosion of fine glacial deposits near river banks, particularly through the middle Fraser upstream of Hope.



Adapted from Cameron et al. 1995.

Figure 1. *Elevation profile of the Fraser River from the headwaters above Moose Lake to the mouth near Vancouver. (The contribution of the major tributaries is indicated as a percentage of the average mean annual flow at Hope).*

Longitudinal studies of water quality (Cameron 1996; Whitfield 1983; Whitfield and Clark 1992), particularly with respect to patterns in dissolved ion content, have demonstrated the primacy of bedrock dissolution in determining overall water quality in the Fraser River Basin (Hall *et al.* 1991). High concentrations of dissolved sulphate, calcium and magnesium are characteristic of headwaters where the underlying bedrock is dominated by evaporites and limestones. Mixing of the Fraser River with the generally softer and clearer water of major tributaries such as the Nechako, Quesnel, Chilcotin and Thompson rivers dilutes and alters significantly the character of the main stem water quality downstream. An example of this effect on water hardness is illustrated in Figure 2, where the addition of tributary waters either reduces or reverses the downstream trend toward increasing hardness in the main stem Fraser River. Dilution by the Thompson River, which comprises roughly one quarter of the flow downstream of its confluence, is particularly evident in the Fraser River at Hope (Figure 2).

Industrial and municipal discharges in the upper basin, particularly in the reach between Hansard and Marguerite on the main stem Fraser, and downstream of Kamloops on the Thompson River, have the potential to affect instream concentrations of common dissolved constituents. French and Chambers (1995) estimated total pulp and paper mill effluent volumes to the upper Fraser at 4.1×10^5 m^3/day, and domestic wastes at about 3.5×10^4 m^3/day—quantities which together can constitute as much as two per cent of the total winter in-river flow at Marguerite. In addition to the commonly measured water quality variables, pulp mills and municipal treatment plants also discharge a wide range of halogenated and non-halogenated organic compounds, metals, dissolved ions and dissolved nutrients (French and Chambers 1995; Norecol 1993). Some inexpensively measured variables are often used to assess the dilution and the extent of contamination by some effluents. For example, the principal tracers of pulp and paper mill effluents are levels of dissolved sodium and dissolved chloride, both of which are highly conservative and present in high concentrations in the discharges (Hall *et al.* 1991).

The largest input of wastewater occurs in the Fraser River Estuary, largely through effluents discharged from the Vancouver area wastewater treatment facilities. Together, three plants in the region release roughly 4.29x10^5 m^3/day (FREMP 1996, Table 5.1). A wide variety of industrial discharges, including metal fabricating plants, sawmills, pulp and paper mills, chemical plants and other activities release a range of contaminants in relatively low-volume discharges to the Fraser River. Other more poorly characterized effluents are those released through the 20 combined sewer overflows (McGreer and Belzer 1999), which transmit an estimated volume of 6.27x10^6 m^3/yr to the North Arm and main stem of the Fraser (FREMP 1996). Evaluating the environmental effects, and even the measurement of ambient concentrations of these contaminants, is complex due to the array of inputs, the spectrum of contaminants and the physical interactions between the river flow and tidal intrusion (Ages 1988; Chapman and Brinkhurst 1981; Hall *et al.* 1991).

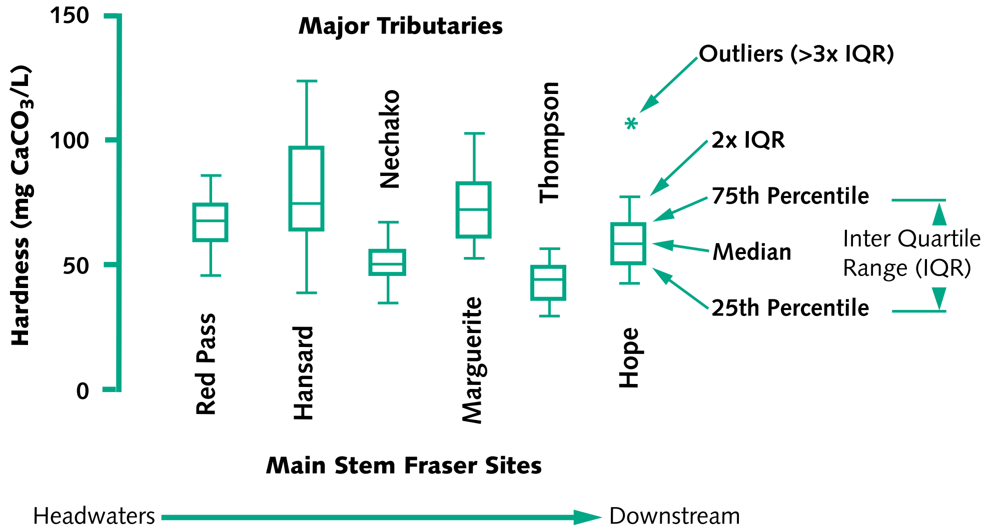


Figure 2. Longitudinal pattern of water hardness in the Fraser River, showing the effect of dilution by confluence with water from major tributaries.

LONG-TERM TREND ASSESSMENT

Water quality monitoring data have been collected since about 1985 at nine sites in the Fraser River Basin under the *Canada-BC Water Quality Monitoring Agreement*—on the main stem Fraser River at Red Pass, Hansard, Stoner (downstream of Prince George), Marguerite (downstream of Quesnel) and Hope; on the Nechako and Thompson rivers, the Fraser's major tributaries; on the Salmon River, near Salmon Arm; and on the Sumas River in the Fraser Valley (Figure 3).

Water samples have been collected and analyzed for dissolved ions, nutrients, trace metals and a range of other variables on a bi-weekly basis at many of the sites since the mid-1980s. Analyses of contaminants such as organochlorines (*e.g.* chlorophenols, guaiacols, catechols) are costly, and despite their release into the environment by industrial discharges, are generally excluded from the suite of monitoring variables. A relatively inexpensive indicator of organochlorine contamination, AOX (adsorbable organohalides), is being measured at the Stoner and Marguerite sites, which are downstream of pulp and paper mills. Preliminary graphical assessment and evaluation of these data relative to water quality guidelines, criteria and objectives have been reported jointly by BC Ministry of Environment, Lands and Parks (BC MELP) and Environment Canada as part of a "State of Water Quality" report series (*e.g.* Lilley and Webber 1997; Wipperman and

Holms 1997a; Wipperman and Holms 1997b). More comprehensive analyses have been conducted by Environment Canada under the Fraser River Action Plan (FRAP) and will be considered below.

In more than ten years of monitoring, measured concentrations of most variables in the main stem Fraser River have not exceeded levels which would compromise potential water uses. Some of the current water quality guidelines for total trace metals developed by the *Canadian Council of Ministers of the Environment* (CCME) were frequently exceeded, particularly the 0.3 mg/L guideline for protection of aquatic life from total iron. However, these high metal concentrations were related to native metal in suspended sediments and represent background levels in the river. In addition, despite major effluent improvement, the present stringent water quality objective for AOX (no significant increase from upstream to downstream of a discharge, Swain *et al.* 1997) was exceeded at locations downstream of pulp and paper mills for much of the period of record.

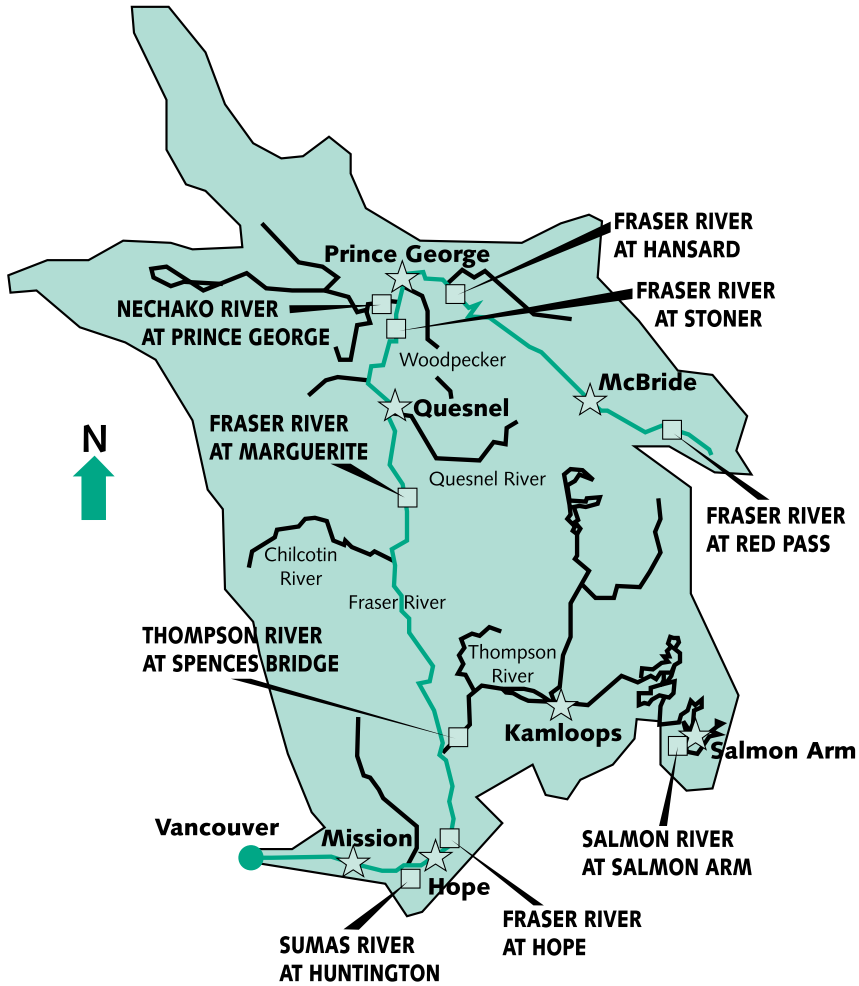


Figure 3. *The Fraser River Basin showing locations of long-term water quality monitoring sites.*

Most variables demonstrate a strong seasonal pattern, associated with seasonal changes in flow, as shown in Figure 4. Dissolved ion constituents and related variables, such as specific conductivity and hardness, show an inverse relationship with flow due to dilution on the rising hydrograph. Others, such as total metal variables (*e.g.* Al, Cr, Co, Fe, Pb, Mn), total phosphorus and turbidity have a strong association with suspended sediments which increase as bed sediment erodes with increasing flows. For many of these variables, the flow-discharge relationship depends on the previous flow state—that is, the relationship at the onset of freshet will often differ from that at the end. This produces a hysteresis (Figure 4) which is particularly common to sediment-related water quality variables, such as turbidity, non-filterable residues, and total metals (Whitfield and Clark 1992; Whitfield and Schreier 1981; Williams 1989). Other variables respond more quickly to changes in flow (*e.g.* conductivity, Figure 4) and show relatively little hysteresis.

Statistical analysis of trends in the Fraser Basin long-term water quality monitoring data has been the subject of two FRAP reports. The first (Shaw and El Shaarawi 1995) examined the available statistical techniques and presented an analysis of a five-year portion of the data set, and a second (Regnier and Shaw

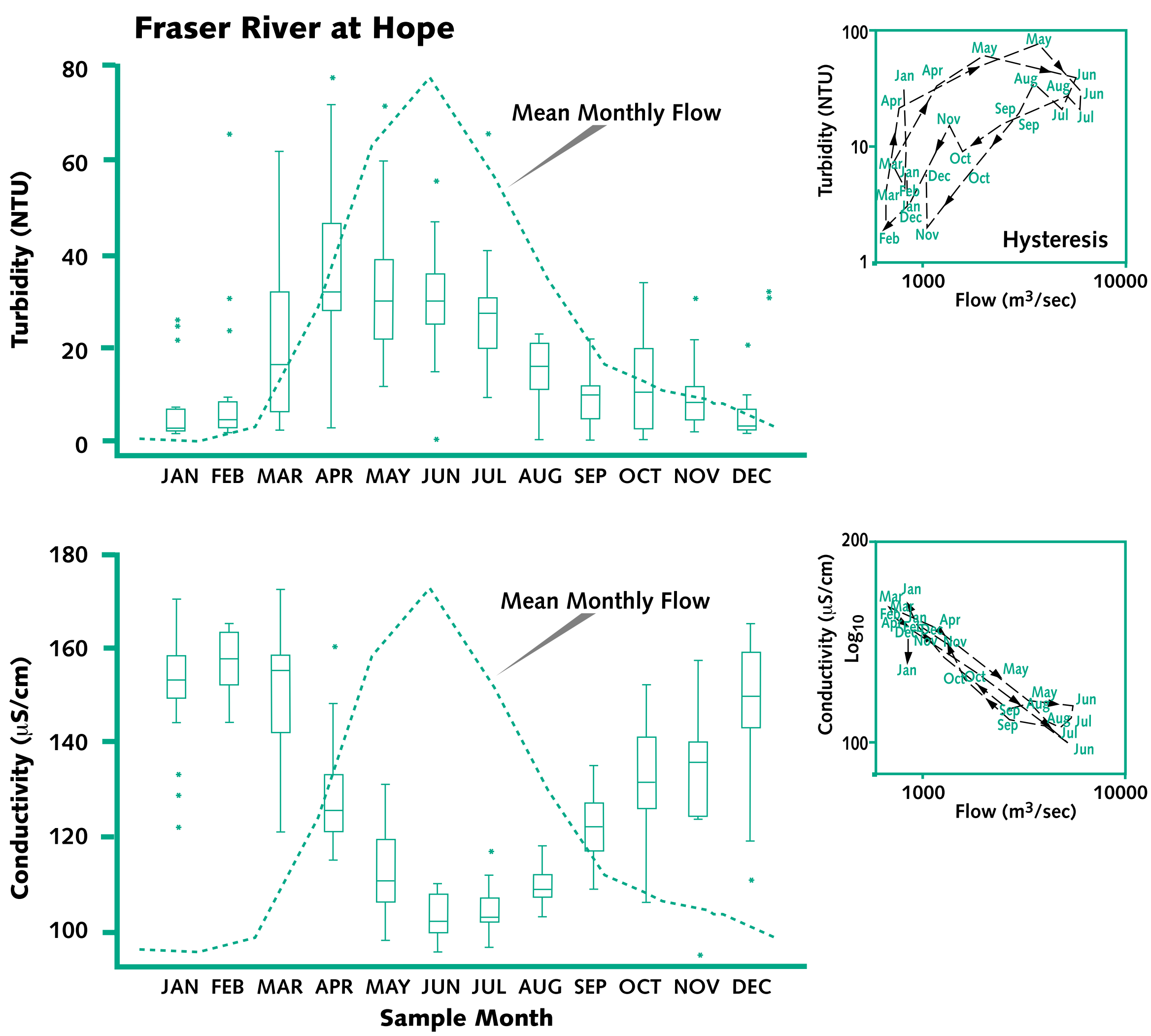


Figure 4. Characteristic patterns in seasonality of water quality constituents as illustrated by data from the Fraser River at Hope.

1998) re-applied these methods on a full 10-year time series. Methods included a variety of non-parametric statistical tests and regression modelling of the time series, in addition to graphical summary and presentation. The non-parametric statistical tests are particularly useful as robust indicators of monotonic trends, either increasing or decreasing, but are unable to detect more complex patterns. For example, a pattern in which a constituent showed an increase for some period (as from a new discharge) followed by a general decline (as from improvements in effluent quality) would be indicated as an overall non-changing trend. Regression modeling, while having a somewhat more rigid underlying set of assumptions, does permit elucidation and modeling of these more complex patterns.

The results of the most robust of these trend analyses are summarized in Table 1. For economy of presentation, Table 1 shows: (1) the directions of trends without indication of magnitude, (2) only those variables where either an increasing or decreasing trend was indicated in data from at least one site, and (3) only those results where both non-parametric and parametric results showed similar trends. Trends in constituent concentrations, either increasing or decreasing, were relatively few and the strongest trends were most clearly

Table 1. Summary of trend analyses of long-term water quality data from sites in the Fraser River Basin, 1985 to 1995.

CONSTITUENT	RED PASS	HANSARD	MARGUE-RITE	HOPE	NECHAKO RIVER	SALMON RIVER	THOMPSON RIVER
Specific Conductivity	➚ ∩	➚ ↗	➔ ↗	➔ ↗	➚ ∪	➔ ↗	➚ ↗
Turbidity	➔ →	➔ →	➔ →	➔ →	➔ →	➚ ↗	➔ ∪
Total Alkalinity	➔ ↗	➔ →	➔ →	➔ ↗	➔ ∪	➔ ↗	➚ ↗
Chloride	➚ ∩	➚ →	➘ ↘	➘ ↘	➔ →	➚ ↗	➘ ↘
Magnesium	➔ ↗	➔ ↗	➚ ↗	➚ ↗	➚ ∪	➔ ↗	➚ ↗
Potassium	n/a	➚ ↗	➚ ↗	➚ ↗	➚ ↗	➔ ↗	➚ ↗
Sodium	➚ ∩	➚ ↗	➔ →	➔ ∪	➚ ∪	➔ ↗	➘ ↘
Sulphate	➚ ↗	➚ ↗	➚ ∩	➚ ∩	➔ ∩	➔ →	➔ ↗

Shaded cells indicate concordance between the non-parametric (Seasonal Kendall's Tau statistic) and the parametric (regression modeling) results. Results are significant at a p<0.10 for the non-parametric analysis and p<0.05 for regression analysis.

➚ *= Increasing trend found by non-parametric tests*
➘ *= Decreasing trend found by non-parametric tests*
➔ *= No trend found by non-parametric tests*
n/a *= Constituent was not analyzed*

↗ *= Increasing linear trend found by regression methods*
↘ *= Decreasing linear trend found by regression methods*
→ *= No trend found by regression methods*
∩, ∪ *= Quadratic trends found*

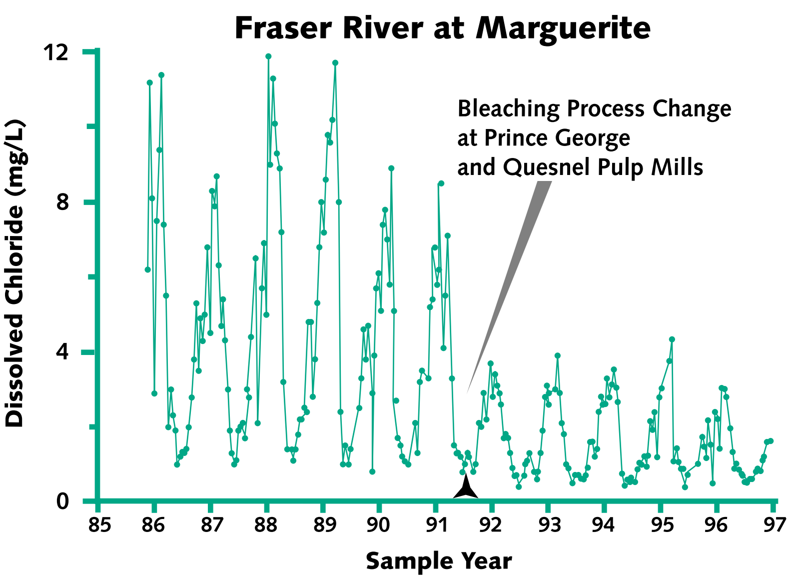


Figure 5. Time-series of dissolved chloride measurements from the Fraser River at Marguerite, showing the immediate effect of process changes in bleaching chemistry by upstream pulp and paper mills.

related to changes in loadings from industrial or municipal effluents. For example, dramatic declines in dissolved chloride (Figure 5) and observable declines in adsorbable organohalides (AOX, Figure 6) at sites downstream of effluent discharges from Prince George and Quesnel are consequences of changes in the bleaching process implemented at most British Columbia kraft pulp mills in 1990–1991 (Krahn 1995; BC Ministry of Environment, Lands and Parks 1995).

Where trends in the main stem Fraser River and large tributaries were detected, none indicated rapid approaches to levels near existing water quality guidelines or criteria, or were at levels where potential water uses might be compromised. This result was not unexpected, since subtle changes in the levels of water quality constituents in the main stem Fraser River, resulting from either natural or anthropogenic causes, would be very difficult to discern over both the natural annual variability and the very high dilution. Smaller tributaries, which are rarely monitored for long-term water quality, are more sensitive to activities within the watershed. Relatively small changes in land-use, effluent discharges or land management practices translate to clear and rapid changes in downstream water quality because of the smaller potential for dilution and the close proximity of the activity to the watercourse. These small watersheds are a critical component in the total Fraser Basin aquatic ecosystem, particularly in their important role as critical spawning and rearing habitat for native salmonids.

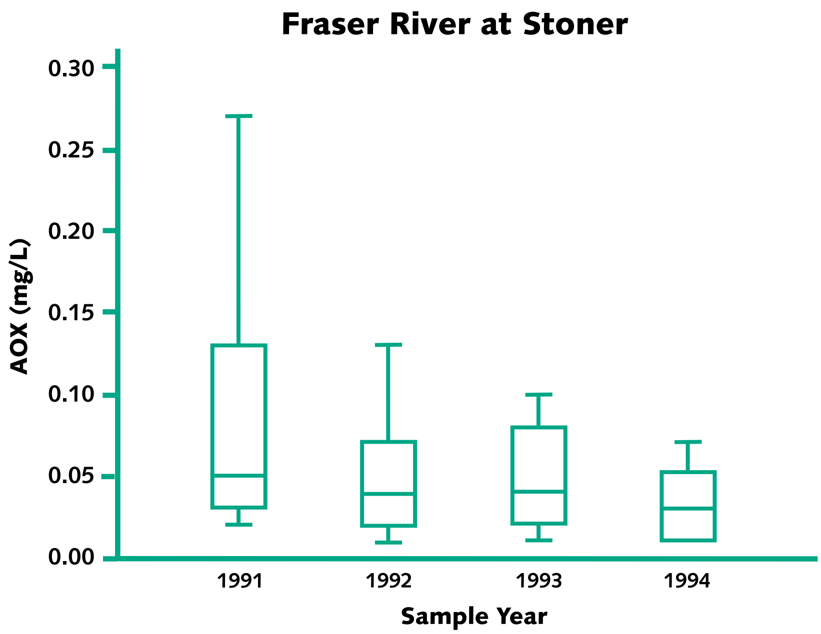


Figure 6. Time-series in concentration of adsorbable organohalides from the Fraser River at Stoner showing the effect of bleaching process changes by upstream pulp and paper mills.

Two Fraser Basin long-term monitoring sites, the Salmon River at Salmon Arm and the Sumas River near Huntington, are within smaller watersheds. Data from the Sumas River are of relatively short duration and sporadic, but the analysis of the nearly ten years of data from the Salmon River illustrates some effects of development in a rural setting on basin water quality. Recent human history of the Salmon River basin includes a range of land-altering activities, such as clearing for agriculture, cattle production, timber harvesting and residential development, which have affected the instream flows through increasing water withdrawal and changing hydrologic conditions. Removal of riparian vegetation and resulting siltation from increasing sediment movement have also resulted in loss of critical salmonid spawning habitat (McPhee *et al.* 1996). These changes in land-use were reflected in the analyses of water quality data from the Salmon River. Increasing trends in concentration of many of the dissolved ions (Figure 7) and in turbidity were detected in trend analyses (Table 1). Parametric analyses revealed an increasing trend in all dissolved ions in the Salmon River, supported by non-parametric statistics in some, though not all, cases (Table 1). This pattern is likely caused by over-allocation of instream water withdrawals (McPhee *et al.* 1996) resulting in low instream water level and movement of ion-rich groundwaters into the stream channel. In addition, groundwater used for irriga-

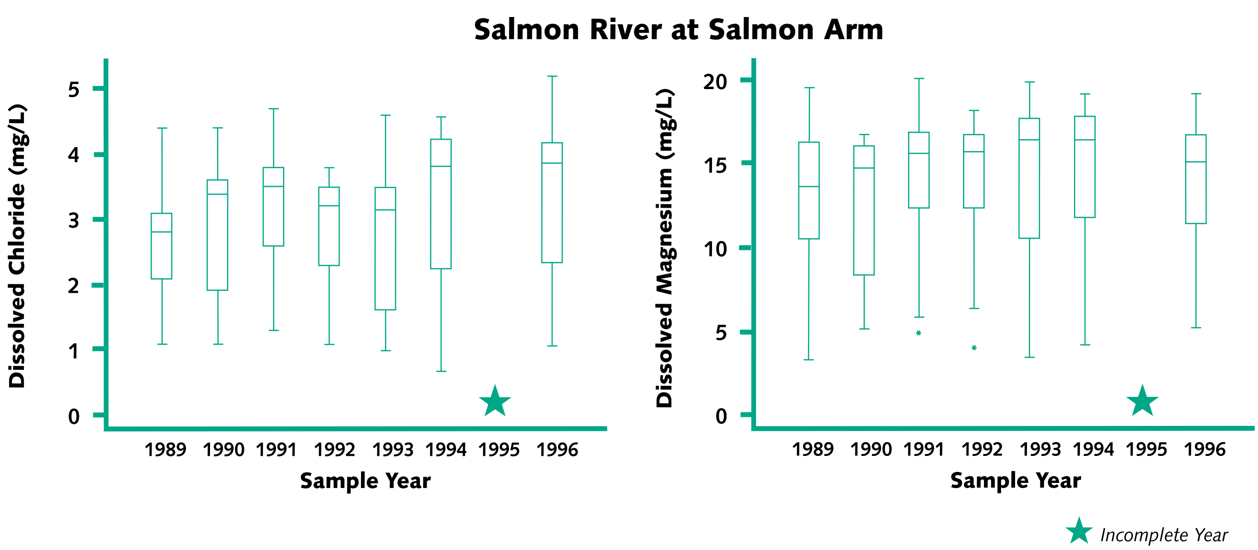


Figure 7. Box and whisker plots of representative dissolved ion measurements from the Salmon River at Salmon Arm. Non-parametric statistical analyses indicated a significant increasing trend in dissolved chloride, but no trend in dissolved magnesium. Regression analyses indicated a significant increasing trend in both.

tion returns to the river through surface runoff. These processes are probably contributing to the overall increase in concentration of most dissolved ions evident in both the trend analyses and summary plots (Figure 7). Accelerated soil erosion and transport to the river, as results from removal of riparian vegetation or from timber harvesting, are probably responsible for the observed increasing trend in turbidity.

These measured increases in dissolved ion concentrations, while not toxic to wildlife or humans and not approaching water quality guideline or criterion levels, are reflecting important and continuing changes in the aquatic ecosystem. Water resource and environmental managers too often confuse "environmental significance" of a water quality trend with proximity of a concentration to an accepted water quality guideline or criterion. Instead, significant changes, however subtle, should be considered as indicative of a process change in some aspect of the ecosystem and deserving of further investigation (Whitfield 1997).

INDUSTRY-SPECIFIC CONTAMINANTS

As part of the FRAP environmental programs, a number of industry-specific contaminants in water have been measured at intervals over the life of FRAP. Pulp and paper-related contaminants including chlorophenolics, dioxins and furans, and resin and fatty acids were measured in the vicinity of pulp and paper mills in the upper basin (Sekela *et al.* 1995; Sylvestre *et al.* 1998a). Water samples were collected on the main stem Fraser River upstream of Prince George (Fraser "reference" area), at Woodpecker (approximately 60 km downstream of Prince George), at Marguerite and in the Thompson River system, upstream and downstream of the pulp mill at Kamloops. Samples were collected as whole-water grab samples as well as through pre-concentration of organic contaminants from centrifuged water with XAD-2 resin columns. A wide range of compounds, including polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and organochlorine pesticides, were measured in addition to pulp and paper mill-related compounds. In addition, occasional samples were collected for nutrient analyses and determination of total and fecal coliform numbers.

Chlorophenolics, resin acids and chlorinated resin acids were elevated downstream of pulp and paper mills on the main stem Fraser River. In the Thompson River system, including the North and South Thompson and Thompson River downstream of Kamloops Lake, values tended to be near or below detection and there were no differences between upstream and downstream sites. Effluent dilution is lowest during the winter low-flow period as reflected clearly in chlorophenol concentrations which varied as much as 40-fold between low- and high-flow sampling periods. Concentrations of these variables in the associated suspended sediment component of the water are presented in Brewer *et al.* (1999). Between 1990 and 1993, mills in the upper Fraser (at Prince George and Quesnel) shifted from 40–60 per cent to 100 per cent chlorine dioxide in the bleaching process. This has had the desired effect of reducing levels of both polychlorinated dioxins and furans (PCDD/F) in the effluents (BC Ministry of Environment, Lands and Parks 1995; Krahn 1995). PCDD/F concentrations in ambient waters were below detection (<1 pg/L) in nearly all analyses, despite sample pre-concentration on resin columns. These frequent non-detections are probably a consequence of both low levels in effluents and very low solubility of these compounds, which will tend to be scavenged from the water column through absorption to suspended sediments.

Dissolved nutrients and microbial variables were also higher in samples downstream of major industrial and municipal effluent sources. While total phosphorus levels at sites downstream of Prince George tend to be higher than at upstream reaches, at least some of the increase is due to natural increases in suspended sediment load. French and Chambers (1995) estimate that only about five (high flow) to 13 per cent (low flow) of the total phosphorus carried in the Fraser River at Marguerite is due to direct effluent sources. However, much (60–95%) of the phosphorus from the major point sources is released in a dissolved form, and more readily available for plant growth than is the sediment-associated phosphorus (French and Chambers 1995).

Cameron *et al.* (1995) identified several other influences of land practices and effluent discharges on water quality. They drew a tentative link between logging activity and elevated dissolved organic carbon levels in waters draining the McGregor and Nechako watersheds, which, as they suggest, requires additional study. They also found very high dissolved carbon dioxide in waters of the Fraser River main stem—levels rivaling those of the Rhine River in Europe. Since high carbon dioxide tension often results from bacterial respiration of organic carbon, a likely cause may be input of organic materials from forestry activities, both timber harvesting and pulp and paper production. Ratios of ^{13}C and ^{12}C in the dissolved organic carbon suggest much of the excess CO_2 in the Fraser may result from organic decomposition in soils and subsequent transport to the river in groundwater. Further work will be necessary to evaluate the significance of these results.

FRASER ESTUARY

Water quality in the lower Fraser Valley is of particular concern, due to the large number and diversity of point and non-point source inputs, coupled with a rapidly increasing population. The population in southwestern B.C. is predicted to grow by 50 per cent over the next 20 years (GVRD 1997), with accompanying increases in the volume of both point and non-point source discharges to the river. Compared to the upper Fraser reaches, the river downstream of Hope has a much lower gradient (*e.g.* Figure 1), receives multiple effluents from both point and non-point sources (such as agricultural runoff), receives a high volume of precipitation seasonally, and is affected strongly by salt water intrusion seaward of New Westminster.

Over the past 20–25 years most data on water quality in the estuary have resulted from short-term, issue-specific sampling campaigns. A compilation of such data from the 1970s (Drinnan and Clark 1980) showed that water quality in the main river channel of the estuary was not degraded. However, in the poorly circulated bottom water of sloughs in the estuary, low oxygen levels, low pH and high organic matter content were common. Although most metals in water were at levels below analytical detection, a few were present at concentrations above existing water quality criteria. Until recently (1993) BC MELP conducted short-term sampling to evaluate attainment of established water quality objectives (BC Ministry of Environment, Lands and Parks 1992; 1993). Water quality objectives were attained in most areas of the lower Fraser River. Exceptions include fecal coliforms in the vicinity of sewage treatment plants in the Main Arm and objectives for dissolved oxygen in some sloughs in the Main Arm. The sloughs are generally poorly flushed and stagnant, and will preferentially accumulate fine particulates and organic materials from the main river flow—the combination of which results in poor water quality.

A recent comprehensive assessment of water quality in the estuary was conducted by the Fraser River Estuary Management Program (FREMP) with funding from FRAP (Drinnan and Humphrey 1997; FREMP 1996). A broad suite of variables was analyzed in this program including physical variables (*e.g.* temperature, conductivity, pH), major dissolved ions, trace metals and a variety of organic compounds. Bi-weekly sampling was conducted for a total of 15 months at three locations; North Arm at Oak Street Bridge, South Arm at Tilbury Island, and in the Main Arm at Mission (Figure 8). Samples were also taken in sloughs on two occasions during low-flow conditions.

Intrusion of marine waters into the estuary during high tide is an important factor in evaluations of water quality in the lower Fraser. The influence is greatest during the winter low-flow period, when the salt water wedge can penetrate as far upstream as New Westminster (Ages 1988; Chapman and Brinkhurst 1981; Drinnan and Clark 1980). In the recent FREMP study, the influence of salt water was evident in all samples collected at low-flow despite measures taken, such as sampling during low slack tide, to avoid this interference.

The three sampling sites in the estuary were similar with respect to most of the physical and inorganic parameters. Significant exceptions were dissolved ion levels and conductivity, which were highest at the Oak St. and Tilbury Island sampling sites due to salt water intrusion. Many trace metals were below limits of

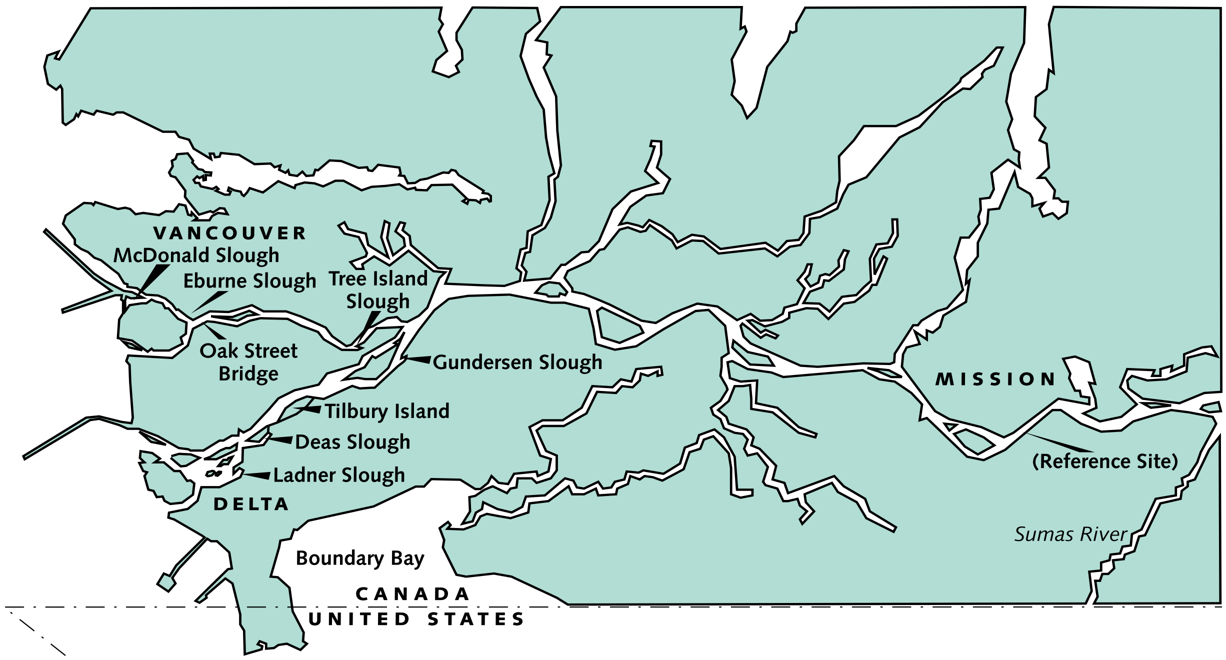


Figure 8. *The Fraser River Estuary area showing sampling sites for the FREMP water quality study (from Drinnan and Humphrey 1994).*

analytical detection, and for those variables that were detected, most were within present British Columbia water quality criteria and Canadian water quality guidelines for the protection of freshwater aquatic life. Concentrations of total aluminum, total copper and total iron did exceed these guidelines frequently, but the measured values were not significantly different from levels in the Fraser River at Hope, upstream of the estuary (Figure 9). This suggests that most of the metals are resulting from upstream erosion of natural bedrock, and that the relative contribution of local effluent sources to the total metal levels in the lower Fraser may be minor. In further support, both total iron and total copper were correlated with suspended sediment concentration, and many of the other trace metals (arsenic, chromium, nickel and zinc) showed peaks in concentration during periods of high flow. Most values did not exceed the established provincial water quality objectives for copper, lead and zinc in the lower Fraser (Swain *et al.* Draft 1995). It should be remembered that the toxicologically important dissolved component of the total metal analysis is generally small relative to particulate metals in natural waters, but may be a large proportion of the total in many industrial and municipal effluents.

Both municipal wastewater treatment plants and stormwater runoff contribute to fecal coliform levels in water in the lower Fraser. Bacterial numbers vary widely over a seasonal cycle, being highest (up to 17,000 fecal coliforms/100ml) in the winter when rainfall is highest and regional treatment plants suspend chlorination of effluent waters (Drinnan and Humphrey 1997). Coliform density in the estuary declines through the summer as a consequence of drier weather and increased disinfection (Figure 10). During this period, levels of fecal coliforms were considerably less than the maximum value for the existing provincial water quality objective (4,000 fecal coliforms/100ml: Swain and Holms 1985) and most single samples were less than the objective for the geometric mean of five samples collected over 30 days (1000 fecal coliforms/100 ml), which apply from April to October. The new objective for the estuary is 200 colony forming units/100 ml (Swain 1998).

Most of the measured organic contaminants were near or below detection and where measurable were many times lower than historical levels. Chlorophenolics, in particular, have dropped nearly an order of magnitude from levels measured in the 1980s (Drinnan *et al.* 1991), attributable to de-registration of chlorophenate

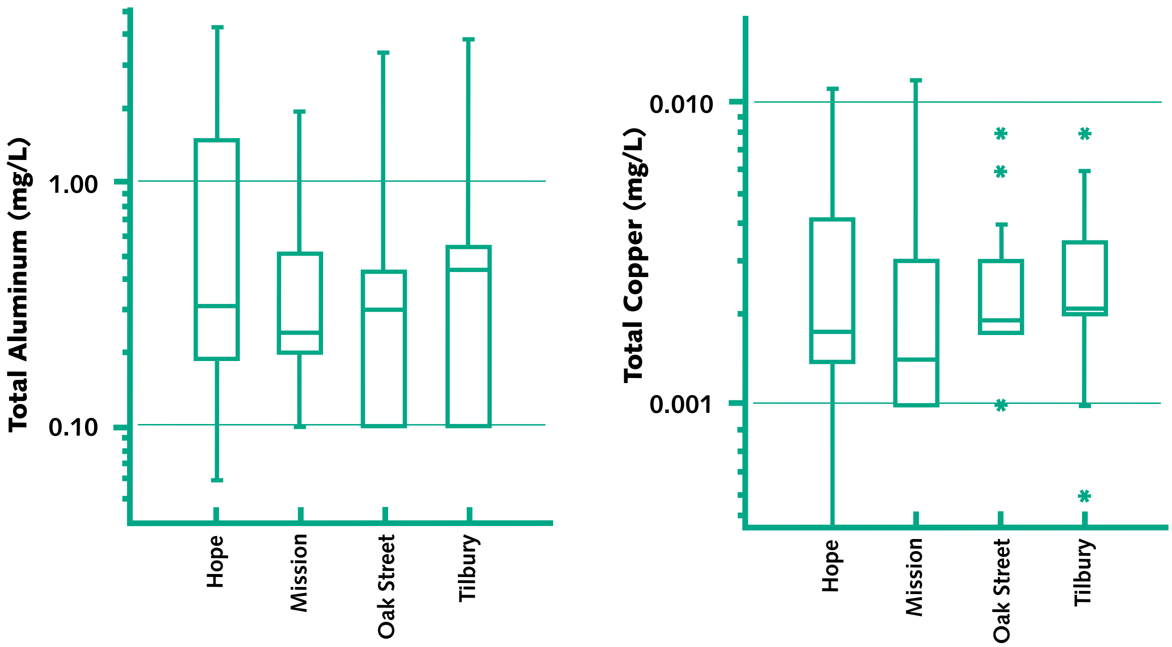
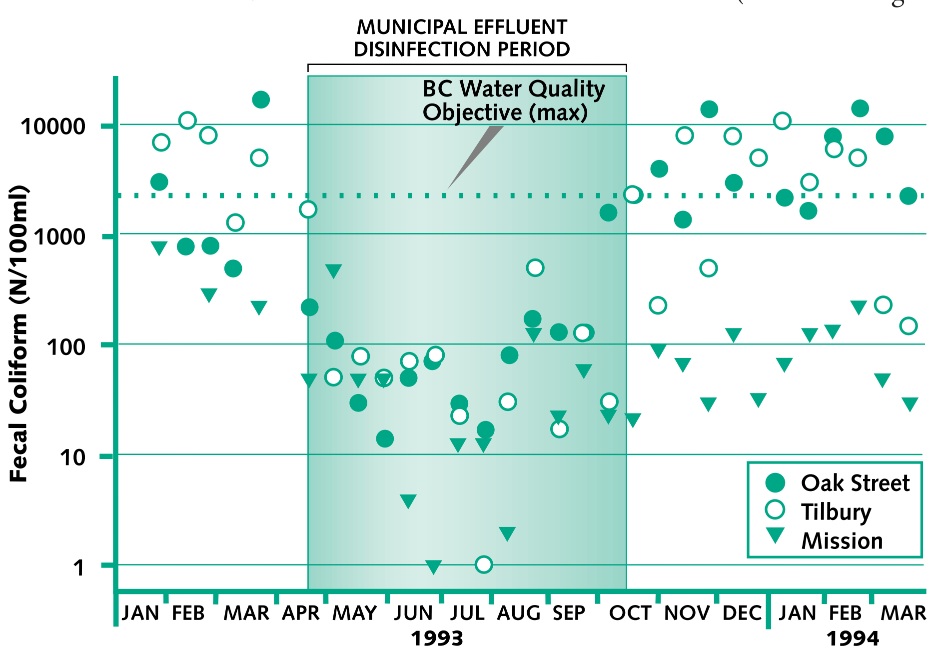


Figure 9. Total aluminum and total copper data summaries for the upstream "control" site at Hope and at the FREMP sampling sites in the estuary.

wood preservatives for general antisapstain treatment. Other compounds continue to be of some concern. For example, concentrations of the PAHs, pyrene, benzo(a)pyrene and phenanthrene exceeded provincial water quality criteria in some sloughs. Elevated levels of PAHs in the estuary, relative to upstream locations, were also measured in bed sediments (Brewer *et al.* 1999) and in fish tissues (Raymond *et al.* 1999)—indicative of point and non-point contaminant sources in this increasingly urbanized estuary. Localized areas of concern remain, particularly in the main stem and Main Arm where levels of total PCBs (0.16–0.26 ng/L) exceeded the B.C. Water Quality criterion of 0.10 ng /L and in the North Arm where measured dioxin TEQs (0.06 ng/L) reached the CCME draft interim water quality guideline for protection of aquatic life (0.06 ng/L) (Sylvestre *et al.* 1998a).

With the successful reduction in environmental releases of bioaccumulative organochlorine compounds, focus has shifted to the effects and levels of other contaminants. An example is nonylphenol, a common component of household chemicals, which causes



The summer depression in numbers reflects a combination of both disinfection of municipal effluents and lower rainfall and stormwater flows.

Figure 10. Seasonal pattern in fecal coliform numbers in the Fraser River near Vancouver.

hormonal disruption in fish (Arukwe *et al.* 1997; Jobling *et al.* 1996; Jobling and Sumpter 1993). Nonylphenol entering the domestic wastewater system in the Vancouver area is released to the environment in sewage effluents in the Fraser River Estuary. Total daily loadings of nonylphenol from the Annacis Island plant alone are in the order of 1.2 (GVRD 1999) to 9 kg/day (Supervisory Coordinating Committee 1987), and ambient levels of 32–130 ng/L and 6.7–7.4 ng/lL have been measured downstream and upstream, respectively, of the effluent discharges (Sylvestre *et al.* 1998b).

CONCLUSION

Trend analyses of water quality at most sites in the basin showed few strong trends. Particularly dramatic increases or decreases were related to changes in the effluent character of major upstream discharges. Subtle trends in particular constituents were seen at a number of sites, and may be an indication of incremental change in some environmental component. Small watersheds, where human activities will have the most effect, are poorly represented in the long-term water quality record.

Water sampling revealed generally low levels of industry-related contaminants. Improvements in industrial wastewater treatment, changes in industrial processes and regulatory action related particularly to wood treatment and preservation have had a favourable effect in reducing organochlorine contamination of waters in the Fraser Basin. Current concentrations of many water-borne contaminants of concern, such as chlorophenolics released from pulp and paper mills and washed from lumber treatment areas, have declined to a fraction of historical levels. The significance of contaminants from other sources, particularly from municipal wastewater treatment plants, will perhaps increase in the future with anticipated population growth throughout the basin.

Despite the relatively high population densities in the Fraser Valley, water quality in the estuary is generally good. Concentrations of a wide range of chlorophenolic compounds, insecticides and other chemical pesticides, and the PAHs are also much lower than historical levels. Areas of some concern do remain, particularly in sloughs near the main channels and in areas of dense effluent and stormwater runoff. Some contaminants, such as the PAHs benzo(a)pyrene and naphthalene associated with urban non-point sources, are of continuing concern in the populated lower reaches of the river. Other less well-known contaminants, such as nonylphenol, prove to be of more concern as information accumulates on both ambient levels and environmental effects.

Future monitoring efforts should address the effects of specific events or activities, including the consequences of active development, on water and general environmental quality in small watersheds. Effects-based monitoring will likely become a trend in the future of water quality assessment in the Fraser River Basin. Concurrent evaluation of both ambient levels of contaminants and effects on biota (through biomarkers, sentinel species or study of aquatic communities) should be a focus of future monitoring programs.

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3.3

SEDIMENT SOURCES, TRANSPORT PROCESSES, AND MODELING APPROACHES FOR THE FRASER RIVER

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A thorough understanding of sediment sources and transport processes in rivers is essential to assessing the impact of pollutants from industrial, agricultural and urban sources on the aquatic ecosystem. Sediments interact with a large number of contaminants and serve as carriers of these contaminants through the river system. This is especially true for fine-grained sediments because of their large specific surface area and high affinity for contaminants. Unlike sand-sized sediments, fine clays and silts are also cohesive, which further complicates their behaviour. Hence, knowledge about the transport and cohesive characteristics of fine sediments is required to better understand their role in contaminant transport and in shaping riverine habitats.

In a number of investigations of contaminant concentrations in Fraser River sediments (*e.g.* Mah *et al.* 1989, Derksen and Mitchell 1994, and Sekela *et al.* 1995), concentrations of a suite of chemicals in suspended and bed sediments, including dioxins, furans, polycyclic aromatic hydrocarbons and chlorophenolics, were observed to be higher in river reaches downstream of pulp mills than those at the reference sites upstream of the pulp mills. The transport of the contaminated sediment, then, determines to a large degree the fate of the contaminants and their interactions with benthic organisms in the riverine environment. For example, deposition of contaminated sediment in sections of the river, where the bed shear stress and turbulence level are low, results in a temporary storage of the contaminants on the riverbed and could expose bottom-dwelling aquatic life and the other organisms connected by the food chain to these contaminants.

Storage of the sediment, and consequently the contaminants, can either be short term or long term depending on the temporal changes in the transport capacity of the river flow. In order to improve our ability to predict the impact of these contaminants on the river ecosystem, it is important that we have a better understanding of the cohesive sediment transport behaviour under different hydraulic conditions of the river.

Predictions of contaminant impacts on the ecosystem of river and other environments are often carried out using contaminant transport models such as WASP5 (Ambrose *et al.* 1991) and EcoFate (Gobas *et al.* 1999). Unfortunately, these models do not include cohesive sediment transport sub-models. However, even if these were available, the data requirements are large and include settling velocity, shear stress and flow relationships, erosion and deposition rates, and critical shear stresses for erosion and deposition of the cohesive sediments. Reliable quantitative estimates of these parameters are not currently available for Fraser River sediments.

For the past two decades, sediment studies in the Fraser River system have been concerned with the transport of cohesionless coarse-grained sediment (see for example: McLean and Mannerstrom 1985; Church *et al.* 1989; Church and MacLean 1994; Kostaschuk *et al.* 1989, 1992; Kostachuk and Church 1993). The work described in this chapter begins to address the data gaps on cohesive sediment and incorporates aspects of coarse-grained sediment dynamics, which influence the processes controlling the erosional and depositional environments for both kinds of sediment. The chapter first considers the sources and annual regime of fine sediment transport in the river. It then reviews historical data collected by Water Survey of Canada on sediment concentrations, load, and some limited characterization of sediment types. It continues with a description of field and laboratory observations of the behaviour of Fraser sediments, and concludes with a discussion of the possibilities for predicting sediment transport and fate.

SEDIMENT SOURCES AND LOADING

The quantity and timing of sediment delivered to a river are determined by the distribution of runoff to the river and the location and character of sediment sources in the drainage basin. As most of the Fraser drainage basin is alpine or plateau country, with elevations near or above 1,000 m, snowmelt in spring gives rise to the major hydrological event, locally called the "freshet." Figure 1 illustrates the dominance of the freshet on the annual discharge pattern, with the river starting to rise in early April and peaking anytime in June. The average sediment concentration tends to peak earlier than the discharge and the loading curve peak occurs between the concentration and discharge peaks.

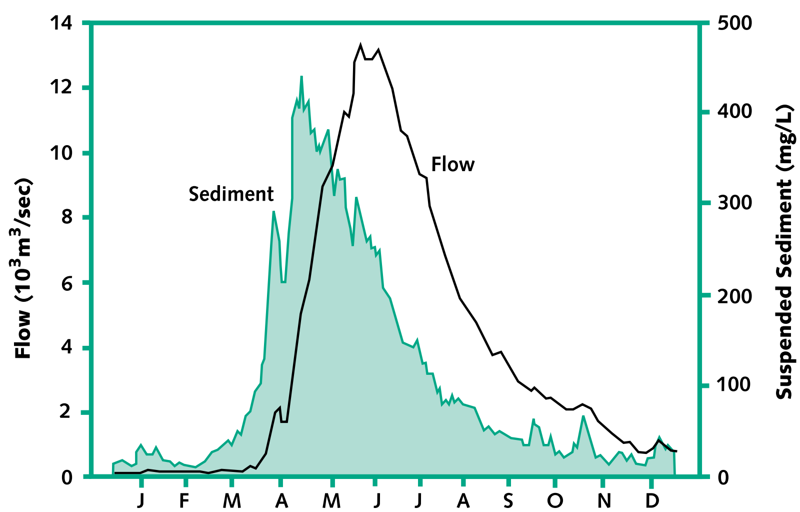

Figure 1. Hydrograph and suspended sediment concentration graphs for Fraser River at Agassiz (Water Survey of Canada Stn. 08MF035), 1972 daily observations. This was an unusually large freshet. The peak of sediment concentration leads the peak flow as a consequence of seasonal exhaustion of readily available sediment along the stream channels as flow increases toward its highest level.

Upland areas in the drainage basin are not prolific sediment sources today. In the high mountains, the main points of sediment production—alpine glaciers, rockfall cliffs and avalanche slopes—are poorly connected with the river channel network. Many alpine streams, which do carry significant sediment loads, drain to lakes in the Fraser headwaters that trap most of the material. In such a large and thinly populated basin, land use is unlikely to have affected overall sediment yield significantly. Thick valley fills of glacial deposits along the Fraser River and its principal tributaries supply the main sediment load of the river directly from the river banks. It is expected, then, that annual sediment yield should reflect the size of the main spring runoff and the amount of bank scour along the main channels, so that long term variations in sediment yield should follow a pattern similar to that for flows. The relation is confirmed by observations at Mission, the lowermost long-term gauge on the river (Fig. 2). The total sediment yield of the basin averages 18.5 million tonnes per year. In comparison with some other large rivers, this is only a modest yield (Table 1). The limited distribution of sources and modest erosion activity are reasons for the low yield.

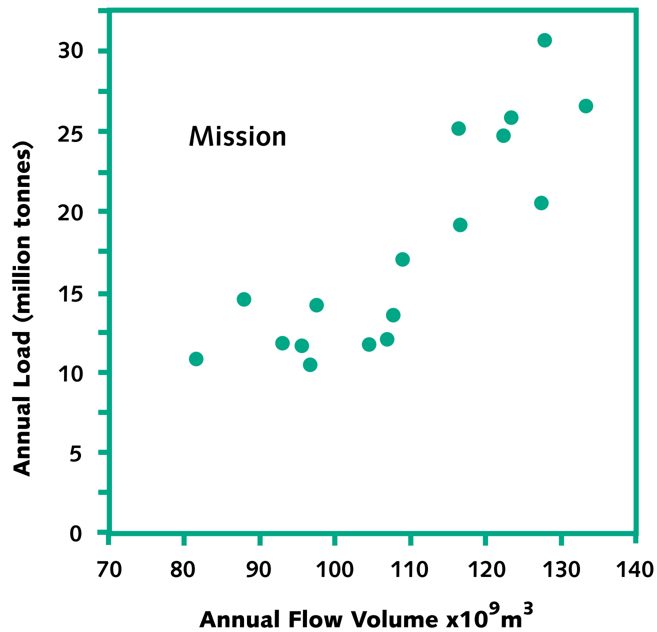

Figure 2. Correlation between annual suspended sediment load and annual flow volume, Fraser River at Mission (Water Survey of Canada Stn. 08MH024).

Table 1. Sediment yield of some large rivers.

RIVER	DRAINAGE AREA (10^3 km^2)	ANNUAL FLOW VOLUME (10^9 m^3/yr)	ANNUAL SEDIMENT YIELD (10^6t/ yr)	SPECIFIC SEDIMENT YIELD (t/km^2.yr)	MEAN SEDIMENT CONCENTRATION (mg/L)
Fraser R. at Mission	214	112	18.5	86.4	165
Chilliwack R.	1.23	2.33	0.130	106	55.8
Columbia R. at Birchbank[1]	88.1	69.5	7.07	80.3	101
Peace R. at Peace River	186	58.6	44.1	237	753
Liard R.	277	75.1	42.5	153	566
Mackenzie R.	1,810	306	100	55.2	327
Rhone R.	90	49	10	111	204
Amur R. (Asiatic Russia)	1,850	325	52	28.1	160
Amazon R.	6,150	6,300	900	146	143
Huang Ho R.	770	49	1,080	1,400	22,000

[1] *Before dam construction.*

The majority of sediment transported in large rivers consists of fine particulate materials suspended in the water column, as opposed to material moving over the bed. The Fraser River is quite typical. At Mission, the lowermost hydrometric station on the river, 99.4 per cent of the sediment load consists of suspended sand, silt and clay, the balance being sand moving on the bed. The division of the load at Mission, in terms of the primary particle sizes, is about 16 per cent clay (very fine material less than 0.002 mm in diameter), 49 per cent silt (material up to 0.063 mm in diameter), and 35 per cent sand. All the gravel and cobbles are deposited in the river before it reaches Mission but constitute only one per cent of the load upstream. The predominance of silt in Fraser River is a bit unusual; most large rivers carry a relatively larger proportion of clay. The reason is the character of the source sediments along the river, which consist of rock grains that were mechanically broken under the influence of freezing and glacial grinding.

Because direct bank erosion constitutes the major source of sediment to the river, its mobilization is related to flows in the river. Hence there is a reason to expect regular behaviour of the sediment regime—perhaps predictably regular. The ability to predict the fine sediment transport in the river would be a substantial advantage in any water quality model.

The set of observations available to study the sediment transport regime of the river consists of regular measurements of suspended sediment concentration undertaken by the Water Survey of Canada at six principal observing stations along the river (Fig. 3) between 1966 and the present day. Records at individual stations vary from 16 to 30 years (Table 2). This represents a remarkably detailed record in comparison with that available for most rivers—the Fraser is one of the best-monitored major rivers of the world. It is the object of the following sections to present an analysis of the relation of flow to sediment transport and to evaluate its potential usefulness in contaminant transport and fate models, such as that developed for the Fraser and Thompson rivers (Gobas *et al.* 1999).

Table 2. *Principal hydrometric and suspended sediment observing stations on the Fraser River, with the period of record for various measurements.*

STATION (WSC[1] NO.)	DRAINAGE AREA (km²)	DISCHARGE RECORDS	SEDIMENT YIELD	SUSPENDED SEDIMENT LOAD PARTICLE SIZE		BED MATERIAL PARTICLE SIZE
				P1	D1	
Hansard (08KA004)	18,000	1952–1996	1972–74	1972–74	1972–74	1973–74
			1976–86	1976–78	1976–81	1979
					1984–86	
Marguerite (08MC018)	114,000	1950–1996	1971–86	1971–79	1971–86	1973–79
				1983–84		
Hope (08MF005)	217,000	1912–49 MC	1965	1965	1965	
		1950–96 RC	1966–69	1967–68	1966–69	
			1970–79	1970–78	1970–79	
Agassiz (08MF035)	217,870	1949–50 MS*				
		1951–55 RC*				
		1956–64 MC*	1966		1966	
		1965 RC*	1970–79	1968	1967–69	
		1966 RS	1970–79	1970–79	1970–78	1970–79
		1967–86 RC	1980–84	1981–84	1980–84	
Mission (08MH024)	228,000	1876–1935 MS*	1965	1965	1965	1965
		1936–64 RC*	1966–71	1966–68	1966–71	1966–71
		1965–96 RC	1972–80	1972–79	1972–80	1972–80
			1981–96	1981–96	1981–96	1982
Port Mann (08MH054)	232,000	1956–71 RC*	1965	1968	1966	
			1966–72	1970–79	1967–78	1965–78
				1981–84	1980–84	

M = manual observations
R = recorded observations
S = seasonal operation
C = continuous operation
** = stage data only*
PI = point-integrated samples
DI = depth-integrated samples
Miscellaneous additional samples have been taken in other years.
[1] WSC denotes Water Survey Canada (Environment Canada)

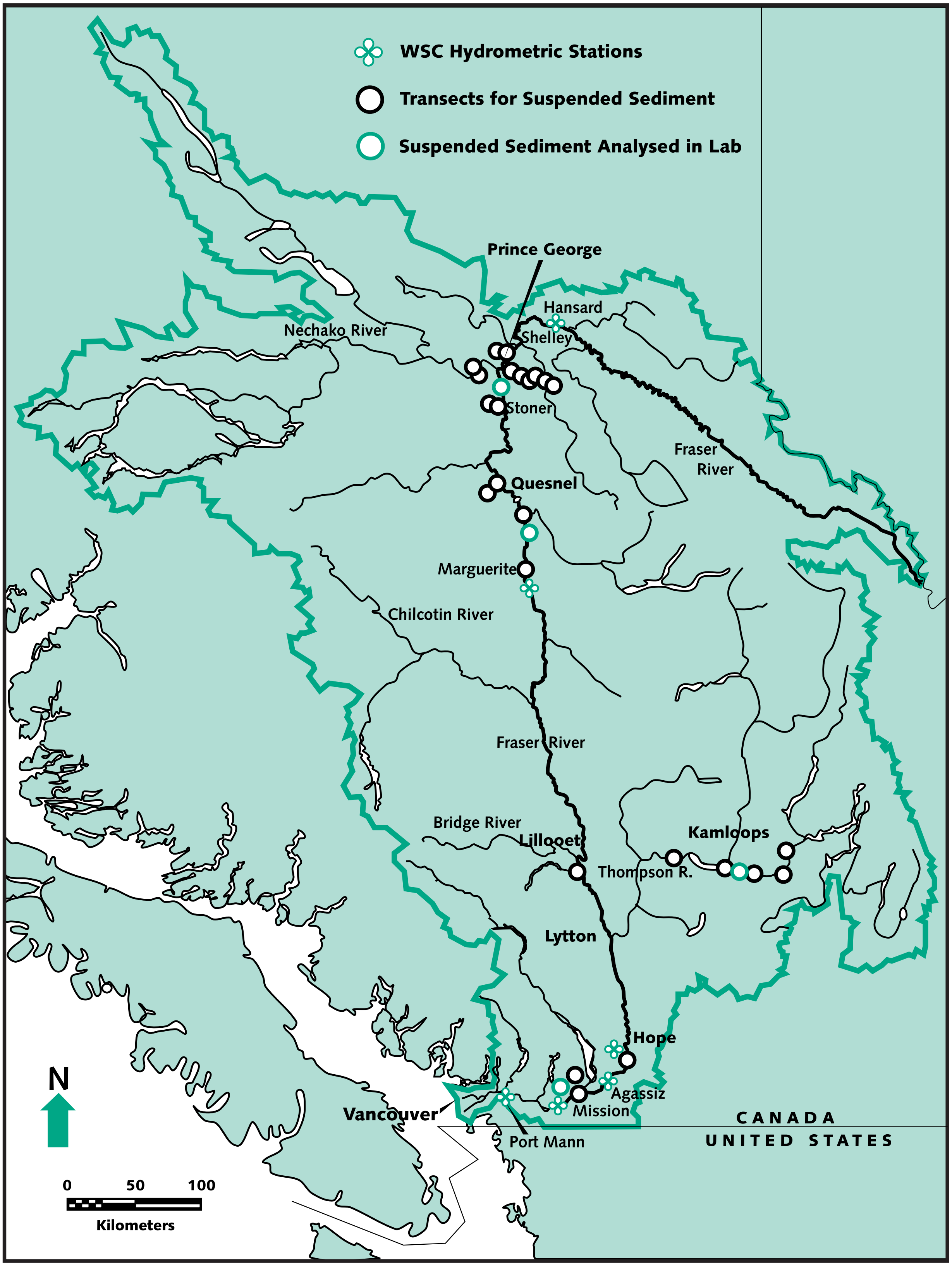

Figure 3. Location of hydrometric stations with suspended sediment monitoring along Fraser River, and of the field observations undertaken for this study.

The Data

Samples were taken from bridges (and from the cable ferry at Marguerite) on a near daily basis through all the months of elevated flows (April through October), and a small number of additional samples were taken during winter low flows. Standard floc-disrupted grain-size analysis was conducted on the sampled material when a sufficient amount was caught, mainly during freshet flows. A daily sample normally consisted of one vertical traverse of the water column at a single location.

Of course, suspended sediment concentration varies across the river channel. On up to 10 or 15 occasions in the year, traverses or multiple point samples were taken in a number of verticals, usually five. The single-vertical observations were subsequently adjusted to represent the average concentration of suspended sediment across the entire channel according to the results of the complete samples. To correct single-vertical samples when no complete sample is available, it is necessary to estimate K, the ratio of the average sediment concentration in the river cross-section to that in the usually measured vertical. This can be accomplished if K varies systematically with some known quantity. An obvious candidate is river discharge, Q. Accordingly, regressions of K on Q were examined.

Figure 4 illustrates a typical result. There is substantial scatter in the values of K, but there is also a systematic trend, and the value of K typically departs from 1.0 (that is, the single-vertical samples are biased). Adjustments were also investigated for size-specific fractions of the suspended sediment load. The pattern of results was similar, although individual regression relations were different. In general, sands are apt to be overestimated by the single-vertical samples, whilst silts and clays are more apt to be underestimated. There is no obvious reason for this pattern other than that the single vertical is customarily located in a part of the river with strong flow, where sand movement is most vigorous.

Predicting the Fine Sediment Load

If measurements are sufficiently frequent, load can be estimated as C_iQ_i where C_i is the observed mean concentration of sediment in the water column and Q_i is the corresponding discharge. Then, either by direct summation of successive measurements, or by interpolation of additional estimates between measurements (usually guided by the hydrograph), the load can be estimated for an arbitrary period. This method was used by the Water Survey of Canada to compute the suspended sediment load in the Fraser River at observing stations during the period of measurements. However, it cannot be used to project estimates beyond the period when measurements are taken.

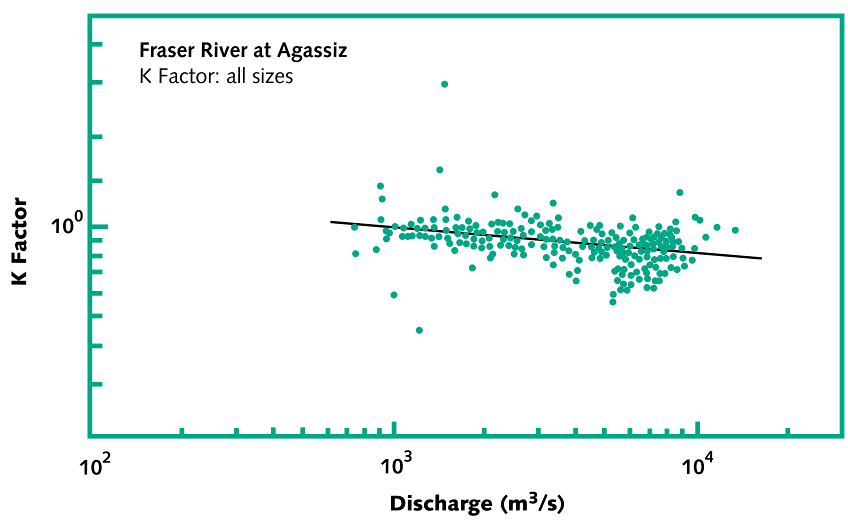

Figure 4. *Variation of the adjustment factor, K (ratio of average sediment concentration in a river cross-section to that measured in a single vertical), with discharge.*

A second approach is to find a predictive functional relation between sediment concentration and some other measured variate, almost always streamflow. So long as the relation—termed a rating curve—remains stable, it can be used to predict sediment load given the continuing record of flows. Our object is to find a prediction model for fine sediment in the Fraser River for continued use beyond the period of measurements.

Suspended sediment rating relations have frequently been presented in the form

$$C_i=C_0Q_i^m$$

in which C_0 is the concentration when Q=1 m^3/s and m is an exponent, often in the range 1.5<m<2.5, which indicates the sensitivity of concentration to changing discharge. The pattern of variation of suspended sediment concentration in the Fraser is more complex than this (Thompson *et al.* 1987). Figure 5 illustrates a typical pattern that exhibits seasonal hysteresis. The best general description of this pattern is

$$C_i=C_0Q_i^{m1}(Q_i/Q_c)^{m2}Q_{i-7}^{m3}$$

in which Q_c is a critical flow level on the rising limb at which the rating sensitivity changes, and Q_{i-7} is the flow seven days before. The term Q_i/Q_c allows us to model the hysteresis; when $Q_i<Q_c$, it is ignored. Physically, the inclusion of this term covers the reduced sensitivity of concentration to changing discharge near the peak of freshet, when the availability of additional fine sediment for entrainment is declining. The term Q_{i-7} reflects the influence of the recent history of flow in determining the continued addition of fine sediment to the water column. The seven-day lag is the time scale of synoptic weather spells.

The rating function was fit to data of individual years at all stations. In addition to data of total suspended sediment concentration, ratings were computed for silt + clay (*i.e.* material finer than 0.063 mm), for fine sand (*i.e.* material with 0.063 mm<D<0.125 mm), and for coarse sand (*i.e.* material coarser than 0.125 mm). The division at 0.125 mm is predicated on the fact that finer material ("wash material"), once entrained, goes immediately into suspension (Sundborg 1967) and may travel a long distance, whereas the coarser material is only intermittently suspended and forms the normal bed material in the lower course of the river (that is, at and below Mission). Ratings were also calculated for all material finer than 0.125 mm, which can be called the "wash load" of the river; it is this material, as will be discussed later, that is apt to form flocs and interact with contaminants.

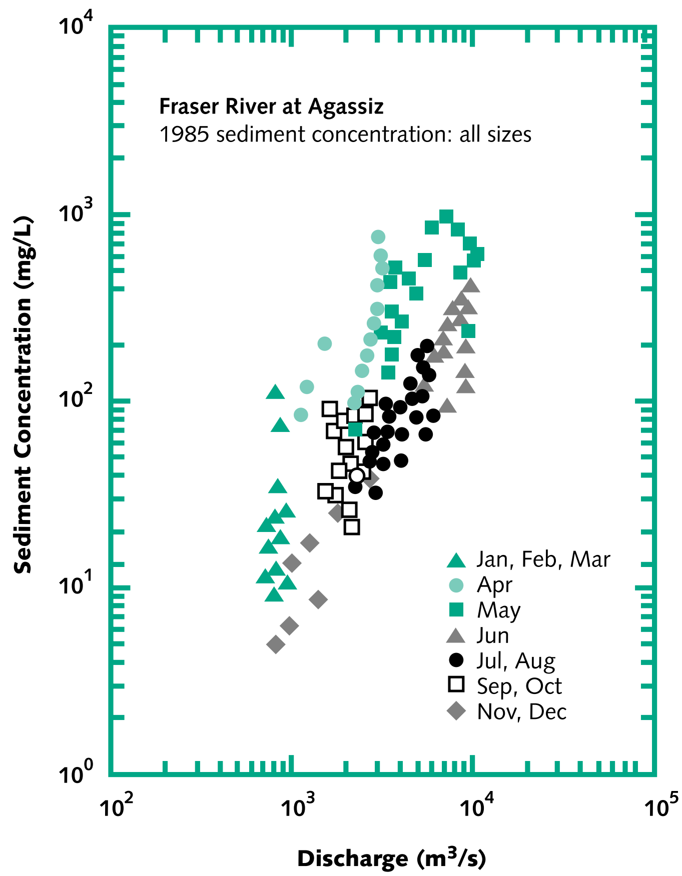

Figure 5. A typical annual rating function of suspended sediment concentration in relation to river discharge, showing the customary hysteresis.

Not all of the terms of the general rating function are always significant. At some stations and in some years, this lack of significance is possibly the consequence of insufficient observations to define the rating function in adequate detail. On the falling limb, neither the m2 nor the m3 terms are significant, so the falling-limb ratings are of classical form. In a few cases, the regression is not significant at all. This outcome usually appears in the silt + clay size range and, occasionally, in fine sand. These are the components of the sediment load that are strongly influenced by sediment supply limitations, so the outcome is not entirely surprising.

More problematically, the rating functions at the same station do not, in general, coincide from year to year. This means that sediment concentration in the Fraser River cannot be predicted on a continuing basis from flow. This is a serious handicap for systematic water quality modeling. Interannual variation presumably is related to the variable character of winter weather, which influences preparation and release of fine materials by weathering along the streambanks, and to the rate of the spring rise in flow, which influences the rate at which material is mobilized. If, as seems probable, a significant amount of these fine materials is stored on the streambed in flocculated form, the history of flows in the preceding autumn and the period of wintertime settling of the material may also be significant. Altogether, these conditions significantly affect the fine sediment mobility.

An implication of this result is that, to assure appropriate input data for a sediment transport and water quality model, observations must be continued at some sites along the river, which will become model input values. To decide how many sites should be monitored will depend upon how well correlated downstream sediment loads are in the short term and how precisely the load needs to be known. Considering the water quality concerns along the river, it would be prudent to maintain one station upstream of significant development on the river, and one upstream and downstream of the Thompson River confluence. Initial choices would be Hansard, Marguerite Ferry and Mission, where historical records already exist.

In comparison with the daily ratings, annual total-suspended-sediment load is well predicted at each gauging station by the annual volume of water passing the gauge. Variance reduction is in the range 83–93 per cent, and the standard error of an annual estimate is about ±10 per cent in the upper river and about ±15 per cent at Agassiz and Mission, but only ±6 per cent at Hope. The high errors in the lower mainland are related to large transient storage of sand in the reach between Hope and Mission.

The change in load between stations is predicted by water volume at the downstream station plus the load of the preceding year at the upstream station. The latter presumably indexes the volume of fine sediment stored in the reach during the low water season. Between Hansard and Marguerite, precision is ±10 per cent, but between Marguerite and Hope it is only ±18 per cent; this probably results from the addition of the large but variable volume of the Thompson River, which carries very little sediment. Between Hope, Agassiz and Mission, predictability is apparently poor because the change in fine sediment load, on average, fluctuates about zero. Absolute precision is of the order ±1 million tonnes, which is comparable with the value in the next reach upriver. This should be compared with a total annual throughput on the order of 18.5 million tonnes.

Sediment Dynamics

On the basis of loads derived from the model equations, changes in sediment transport along the river can be predicted and studied. Average annual-fine-sediment pickup in successive reaches is given in Table 3. Major sediment recruitment occurs in the reach between Hansard and Hope (in fact, nearly all of it upstream of Lytton, where the Thompson River joins the Fraser). There is a near balance of the load below Hope, with deposition of fine sediment occurring along the Hope-Mission reach in most years and net erosion in years with high floods. There are, therefore, substantial swings in the transient storage of sediment, mostly sand, in this reach, but with a balance of accumulation. This is expected in this distal reach of the river, where a substantial floodplain has been constructed within the last ten thousand years. The situation emphasizes the significance of the

Table 3. *Average annual fine sediment recruitment along Fraser River.*

REACH	ANNUAL RECRUITMENT (million tonnes/yr)	STD. DEVIATION
Above Hansard	2.720	0.671
Hansard-Marguerite	7.041	2.820
Marguerite-Hope	7.743	1.640
Hope-Agassiz	-1.095	1.964
Agassiz-Mission	-0.217	1.742
Hope-Mission	-1.261	2.852

Because the rating curves upon which the results are based are optimised at individual stations, there are small discrepancies amongst the various sums that can be formed.

Hope gauge as a reference station for considering sediment transfers. Year-to-year variations in sediment recruitment are shown in Figure 6.

A significant amount of fine sediment is seasonally stored along the river. Most of the storage sites are on the open riverbed or in sandbars along the main channel bank. There are few backwaters for longer term storage. As discussed below, a mechanism exists for flocculated sediment to be seasonally deposited and entrained on the riverbed. The effect is evident by the film of fine sediment which coats the emerging cobble riverbed as the annual flood recedes in late summer (soon washed off exposed rocks by rain).

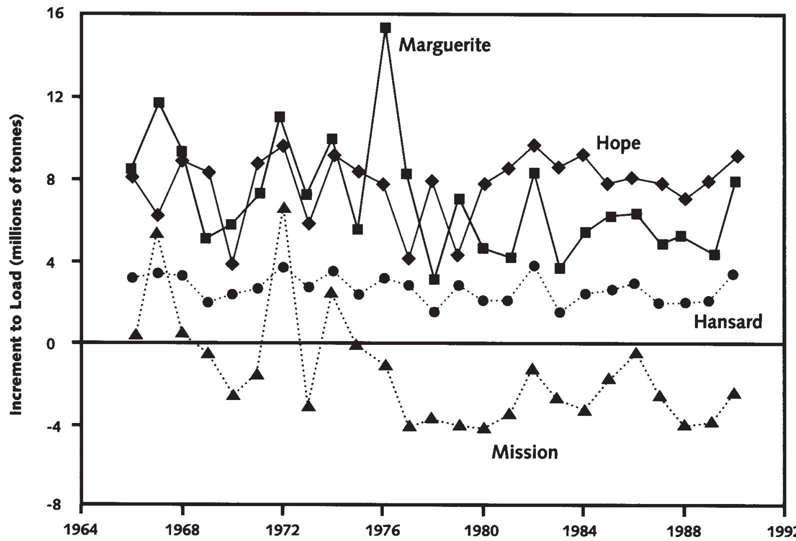

Figure 6. Annual fine sediment recruitment in four major reaches of Fraser River (millions of tonnes). The material is derived from stream banks and from tributary inflows.

Another mechanism augments seasonal storage and creates longer term storage as well. Upstream from Bridge River rapids, much of the river has a cobble bed, which may scour during freshet. Data to study this phenomenon are available only at the Marguerite gauge, where soundings made during flow gauging permit changes in bed elevation to be studied. Hickin (1995; see also Carson 1988) has studied the data and shows up to two metres of bed scour during the highest freshets (see Fig. 7). This creates a scour volume of up to 300 m^3 in the channel bed per metre of channel length. It is probable that such a zone could harbour up to 40 m^3 (or 100 tonnes) of fine sediment (*i.e.* material smaller than 2 mm in diameter), stored in the interstices of the larger material. There is a potential, then, to store as much as 100,000 tonnes per kilometre of scour-prone channel if the Marguerite scour figure is representative. Much of this material would be sequestered for many years, because full scour would occur only occasionally. Furthermore, extensive reaches probably experience much more limited scour. Nonetheless, there appears to be a potentially large capacity here to sequester and release fine sediment, even if only a small fraction of this storage potential is realized.

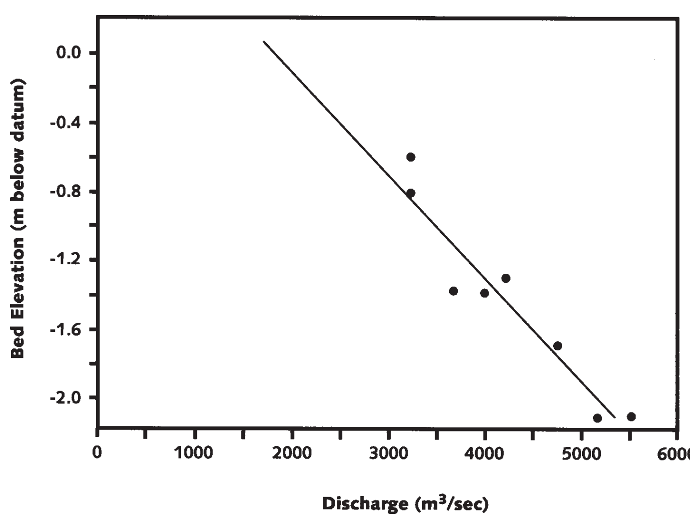

Figure 7. Relation between river discharge and bed elevation at Marguerite, based on the maximum scour observed in each year of record (modified from Hickin 1995; figure 8.6c).

In a small number of subsurface bed material samples from Marguerite analyzed by Carson (1988), about 10 per cent of the material was finer than 0.05 mm. However, almost no material finer than 0.062 mm was found. It is not known whether finer material was lost in the sampling procedure, or was not present. The sequence of years in which significant exchange occurs between the sub-bed storage and the mobile sediment load, juxtaposed on the sequence of transient additions of contaminants to the river, creates the possibility for certain "pools" of contaminated fine sediments, specifically silts and clays, to remain in the riverbed for many years. Episodic release of such stored material would significantly affect the observed variance of contaminant occurrence in the river and could complicate both trends and compliance monitoring.

SEDIMENT TRANSPORT PROCESSES

Transport of the coarse-grained sediments, which behave as individual particles when transported by a river flow, has been studied extensively and a large body of knowledge exists in the literature. With this existing knowledge it is possible to make reasonable predictions of transport parameters such as the critical flow condition for initiation of sediment motion, the sediment transport rate, the characteristics of bed forms such as dunes and ripples and the friction factor of sediment transporting flows. Transport processes of fine sediments, on the other hand, are not very well studied and there is a lack of generally accepted theoretical formulations for treating fine sediment transport in river flows. The reason is that the fine sediments in the size classes of silt and clay, classified as cohesive sediments, exhibit a strong interaction among the sediment grains and form sediment flocs. The interaction depends on the flow turbulence and the physical, chemical and biological properties of the sediment-water mixture. To improve our understanding of fine sediment transport processes in the Fraser and Thompson River system, we initiated a field and a laboratory study to formulate a new fine sediment transport model (FINESED). The main conclusions of the sediment transport studies and the salient features of FINESED are summarized here.

Field Evaluation of Cohesive Sediment Transport

The purpose of the field evaluation was to measure the size distribution of the sediment in suspension in its natural environment and to determine if these sediments were transported in a flocculated form or not. Five field surveys were carried out between September, 1993 and October, 1996. The four fall and one spring dates of these surveys were selected to coincide with low flow periods so that the effects of effluents from pulp mills and other sources on the suspended sediment in the river could be examined (see Krishnappan and Lawrence 1999). Transects were located at 12 stations along the Fraser main stem and on the Nechako and Thompson rivers (Fig 3). Size distributions were measured using an instrument assembled from components of a commercially available laboratory laser particle size analyzer (Krishnappan *et al.* 1992). This instrument was capable of measuring the *in situ* distribution of sediment in suspension without disrupting the flocs, unlike the traditional sampling and analysis methods, which are known to cause floc disruption. To assess the state of flocculation of the suspended sediment, the *in situ* distributions measured in the field were compared with the distributions of the primary particles in concurrently collected samples that were subjected to sonic vibration to ensure total disruption of flocs. The particle size distributions were obtained with a laboratory particle size analyzer operating on the same principle as the field instrument.

A comparison of the median sizes of the particles measured in the field and the primary particles after disruption in the laboratory were made for a number of sampling stations as shown in Figure 8. From this figure, it is evident that at the transect at Shelley (a station upstream of all pulp mills), the median sizes of the *in situ* and primary particles are nearly equal, which implies that the particles at this transect are transported as individual particles rather than as flocs. At a downstream transect, which is located at about 300 m downstream of the Northwood Pulp and Timber Ltd. pulp mill outfall at Prince George, the condition of the particles' state is very different. Here, the median size of the *in situ* particles is higher than that of the primary particle size distribution, which implies that the particles at this transect are transported as flocs. This flocculation phenomenon could have been caused by the presence of the pulp mill effluent. An experimental verification of this hypothesis had been provided by a set of controlled experiments using the

Fraser and the Nechako River water and the Northwood pulp mill effluent (see Krishnappan and Lawrence 1999).

Figure 8 also shows the comparison of the median size of *in situ* and primary particles for the Nechako River, as well as the Fraser River at Stoner, Quesnel, Lillooet and Mission. The Nechako River data show that the particles in this river are also flocculated. The agent responsible for the flocculation of these sediments could be effluent from sewage treatment plants that contain organic matter and bacteria. The presence of bacteria has been found to cause flocculation of sediments by way of secretion of polysaccharides, a glue-like substance that promotes bonding among particles (Van Leussen 1988). The data for the Lillooet transect show that the flow velocities and turbulent shear stresses are high enough in the canyon to break up the flocs into individual particles. Farther downstream at Mission, where the flow becomes much less turbulent, the flocs appear to reform.

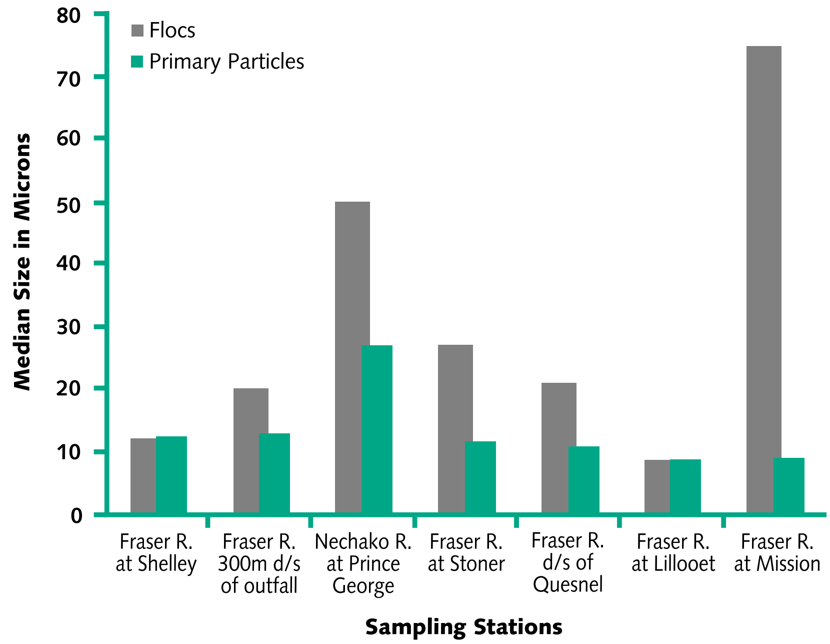

Figure 8. *Floc and primary particle sizes at several locations in the Fraser and at one location in the Nechako River.*

From the field surveys, it became apparent that the suspended sediments of the Fraser River system should be treated as cohesive sediments. Therefore, their transport characteristics cannot be predicted theoretically (Krishnappan and Ongley 1989). With the current state of knowledge on cohesive sediment transport, the transport parameters of cohesive sediment can only be obtained through direct measurements in special flumes such as a rotating circular flume. Such an approach was adopted for the present study.

Laboratory Evaluation of Fraser River Cohesive Sediment Behaviour

The Rotating Circular Flume (RCF) located at the National Water Research Institute was used to evaluate depositional and erosional process parameters of sediment-water mixtures from the Fraser River at Prince George, Quesnel and Mission, the South Thompson River at Kamloops and the Nechako River at Prince George. A brief discussion of the testing procedure and the results are outlined below.

The RCF consists of a circular flume, 5 m in mean diameter, 30 cm wide and 30 cm deep, resting on a rotating platform, 7 m in diameter, with a rotating lid that fits inside the flume with close tolerances. By rotating the flume and the lid in opposite directions at different speeds, it is possible to generate flows with characteristics similar to straight and uniform channel flows. Complete details of the flume can be found in Krishnappan (1993).

The deposition characteristics of Fraser River sediment were studied by placing the sediment-water mixture in the flume and operating the flume at different speeds to simulate different flow conditions. At each speed, the flume was operated for a period of about four hours. Concentrations of sediment in suspension and the size distributions were monitored as a function of time during the course of the experiment. The concentration results from a typical deposition test are shown in Figure 9. This figure shows that for a particular bed shear stress, the concentration drop is steep in the beginning and levels off gradually, leading to an eventual steady state concentration. Earlier studies demonstrate that the attainment of a steady state concentration during the deposition of a cohesive sediment is due to the fragility of the flocs and their inability to penetrate the high-shear-stress region near the riverbed (Partheniades and Kennedy 1966).

The deposition experiments also revealed that the steady state concentration was a function of the initial concentration, and the ratio between these two concentrations was constant for a given shear stress. This implies that when a known amount of cohesive sediment enters the river, a fraction of the sediment will deposit and the remaining sediment will stay in suspension indefinitely. The fraction that stays in suspension indefinitely is a function of the bed shear stress for a particular type of sediment. It is interesting to note here that in the case of cohesionless sediment (sediment that behaves as individual particles), the steady state concentration is a function of only the bed shear stress and does not depend on the initial concentration. This is one of the important differences between the transport characteristics of cohesionless coarse-grained sediment and those of cohesive fine-grained sediment.

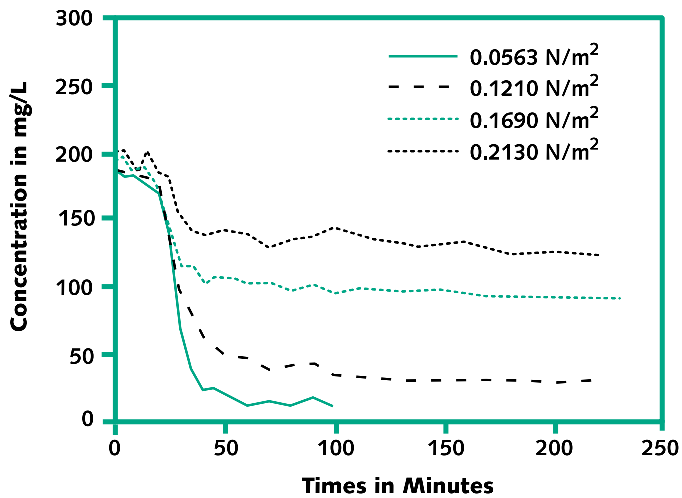

Figure 9. Variation of suspended sediment concentration over Fraser River sediments with different shear stresses. Shear stress expressed as Newtons/m².

The deposition experiments provide quantitative estimates of the amount of sediment that would deposit under a particular bed shear stress given the initial amount and kind of sediment that had entered the river reach. The shear stress at which all of the initially suspended sediment would deposit is termed the critical shear stress for deposition. For Fraser River sediment, an average critical shear stress for deposition of 0.05 Newtons (N)/m^2 was obtained. The variation of this parameter at the different sampling locations was within 15 per cent of the average value. Similar measurements carried out in the Athabasca River near Hinton, Alberta, yielded a value of 0.085 N/m^2 (Krishnappan and Stephens 1995). A lower value of the critical shear stress for deposition means that Fraser River sediment stays in suspension more readily and has lower settling velocities than sediments in some rivers on the east side of the Rocky Mountains.

The re-suspension potential of the deposited sediment was also studied using the rotating flume. For these tests, the sediment was allowed to deposit on the flume bottom over a known period of time at a shear stress slightly below the critical shear stress for deposition, then the erosion characteristics were studied by applying the bed shear stresses in step increments. At each step, the concentration of the eroded sediment and its size distribution were measured as a function of time. A typical result from an erosion test is shown in Figure 10. From such results, we can estimate the shear stress at which the sediment begins to erode, *i.e.* the critical shear stress for erosion. An average critical shear stress for erosion of 0.120 N/m^2 was obtained for Fraser and Thompson River sediments. The variation of this parameter with the sampling locations was within 15 per cent of the average value. In comparison, for Athabasca River sediments, a critical shear stress for erosion of 0.170 N/m^2 was obtained (Krishnappan and Stephens 1995). A lower value for critical shear stress for erosion means that Fraser River sediment is more mobile and can easily be brought back into suspension.

From the deposition and erosion experiments, we observe that the values of critical shear stresses for deposition and erosion are different, which is typical for cohesive sediments. In contrast, for non-cohesive sediments, the two values merge into one and the critical condition for deposition is equal to the critical condition for erosion. A single value for the critical conditions means that these sediments can undergo deposition and erosion simultaneously.

For the Fraser and Thompson River sediments, the ratio between the critical shear stress for erosion and the critical shear stress for deposition is 2.4, which is not too far off from the value of 2 that was obtained for Athabasca River sediments (Krishnappan and Stephens 1995). Nevertheless, the transport characteristics of these two sediments are very different. As pointed out earlier, the Fraser sediments are much more mobile than the Athabasca sediments because the former have lower values for the critical shear stress for deposition and erosion. Further details of the deposition and erosion experiments using the Fraser River sediments can be found in Krishnappan and Engel (1997).

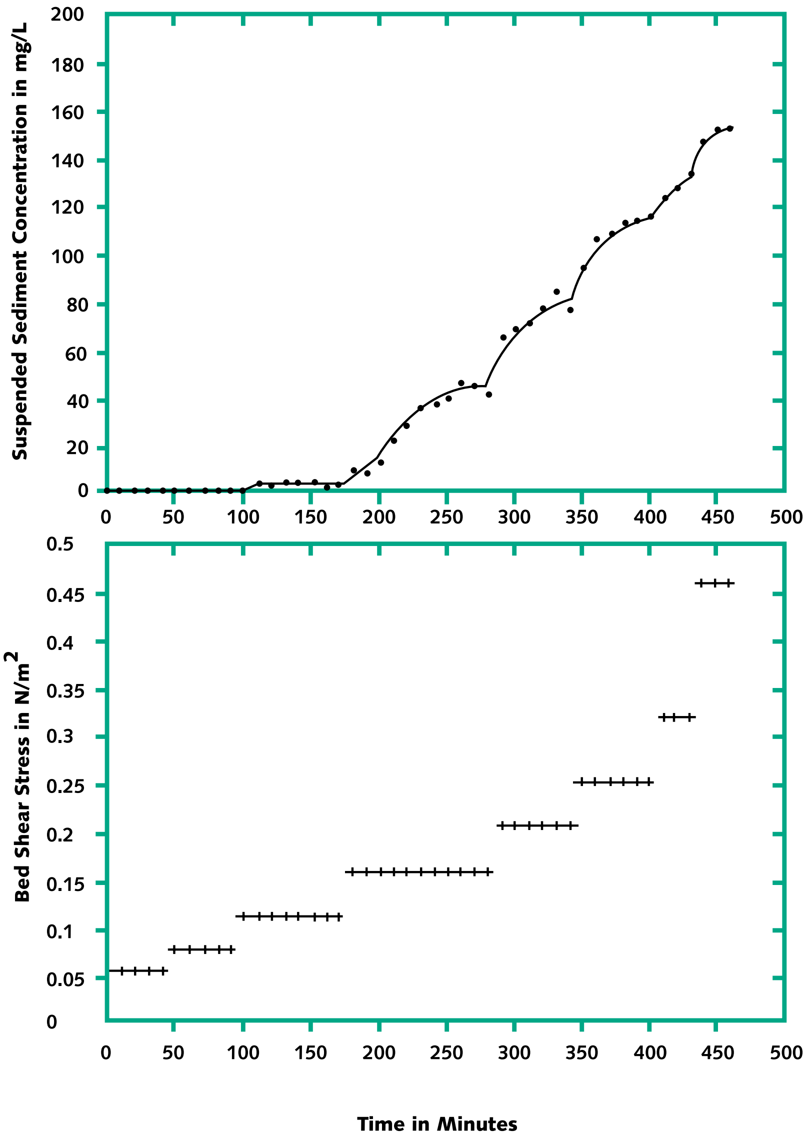

Figure 10. Erosion characteristics of Fraser River fine sediment demonstrated by the increase in suspended sediment concentration with time in response to a series of step-wise shear stress increases starting below the critical shear stress for erosion.

MODELING THE TRANSPORT OF COHESIVE SEDIMENTS IN THE FRASER RIVER SYSTEM

Existing models of cohesive sediment transport (*e.g.* SERATRA and FETRA by Onishi and Thompson [1984], the University of California model by Ziegler and Lick [1986], Finite Element Hydrodynamic and Cohesive Sediment Transport Modeling System by Hayter [1987], TABS-2 by U.S. Army Corps of Engineers [Thomas and McAnally 1985] and WASP5 by US Environmental Protection Agency [Ambrose *et al.* 1991]) assume that the transport characteristics of fine sediment are analogous to those of coarse sediment and treat the fine sediment transport using sediment transport theories developed for coarse-grained sediments.

As indicated above, our study has found a sizeable difference between the shear stress for deposition and erosion for fine sediments in the Fraser suggesting that the theories of coarse-grained sediment behaviour are not adequate for modeling fine sediment transport. These theories give equal critical shear stresses for erosion and deposition. In the case of fine, cohesive sediment, however, the two critical shear stresses demarcate three sedimentary regimes. In the first regime, under low flow conditions below the critical shear stress for deposition, only deposition occurs. In the second regime, under higher flow conditions between the critical stress for deposition and erosion, the fate of a sediment floc depends on whether it is in the bed sediment or suspended in the water and on its specific behaviour with respect to the average critical shear stress for deposition and erosion measured for the whole sediment population. A suspended floc will remain in suspension in this stress regime unless it is denser (a floc with higher settling velocity) than the "average" floc, in which case the floc will deposit at a stress above the average critical stress for deposition. Once attached, however, this floc will not re-suspend until, or if, the critical stress for erosion is exceeded. A floc attached to the bed sediment, on the other hand, will stay attached in this stress regime unless it is more fragile (a floc that is weak and easily broken up) than the average floc in which case it could become suspended below the critical stress for erosion for the average floc. In this regime both erosion and deposition can occur, but they happen at different stresses and to different flocs. In the third regime, under high flow conditions, above the critical shear stress for erosion, only re-suspension occurs.

Accordingly, a true representation of the erosion and deposition behaviour of cohesive sediments at different shear stresses is important for contaminant transport models, as the fate of the adsorbed contaminants will be strongly influenced by the shear stresses occurring over the calculation period. A model that assumes erosion and deposition are occurring simultaneously, although at different rates depending on the shear stress, will predict accumulation and dispersion outcomes that differ from a model that accounts for the behaviour of the sediments in the three shear stress regimes discussed above. Hence, for stresses below the critical stress for deposition, a model incorporating the cohesive sediment assumption will predict deposition only, while a model incorporating a non-cohesive sediment assumption will predict both deposition and erosion. The cohesive model would predict a higher contaminant concentration in the bed sediment than the non-cohesive model if the suspended sediment load was more contaminated than the bed sediment, or vice versa if it was less contaminated.

For stresses between the critical levels for deposition and erosion two general outcomes are possible. For example, the cohesive model would predict no dispersion of bed sediments and greater dispersion of suspended sediments than the non-cohesive model as the river flow and associated stress increased past the critical stress for deposition. However, as a high river flow declined past the critical stress for erosion the cohesive model would predict a cessation of bed sediment dispersion and a higher dispersion of suspended sediments. The impact on contaminant concentrations in the suspended and bed sediments under these two scenarios would vary dramatically depending on the modeled inventory of accumulated bed sediment and contaminant burden in the bed and suspended sediments.

Lastly, at stresses above the critical level for erosion, the cohesive model would predict no deposition and the beginning of the dispersion of contaminated sediments deposited earlier which, depending on their contaminant content, could decrease or increase the contaminant concentration in suspended sediments and water. The non-cohesive model, on the other hand, would still predict exchange of contaminated sediments between the water column and the bed sediments, which would effectively slow down the rate of bed sediment dispersion even though the model would undoubtedly predict a net erosion of the accumulated sediments.

While the theoretical implications of these different model outcomes have been stated here, there is a need to examine the practical implications of this modeling approach over different time and spatial scales as well as hydrological regimes. To fulfill this need, a new sediment transport model called FINESED was formulated for Fraser River sediment based on the results of the laboratory experiments.

The deposition and erosion experiments in the rotating flume provide quantitative estimates of the fraction of sediment that would deposit and the fraction of the deposited sediment that would re-suspend under a particular bed shear stress. From these experiments, empirical relationships were developed to quantify these fractions in terms of the ratio between the bed shear stress and the critical shear stress for deposition. These relationships formed the basis of the FINESED model. According to the relationship developed for the deposition process, when the ratio between the bed shear stress and the critical shear stress for deposition is less than unity, all of the initially suspended sediment would deposit. When the ratio is between 1.0 and 10.5, only a fraction of the initially suspended sediment would deposit and the amount of the deposited sediment is given by the *deposition function* derived from the laboratory experiments. When the bed shear stress is in excess of 10.5 times the critical shear stress for deposition, none of the initially suspended sediment would deposit. According to the relationship developed for the erosion process, the re-suspension of the deposited sediment was also expressed in terms of the ratio between the bed shear stress and the critical shear stress for deposition. When this ratio is less than 2.4, none of the deposited sediment will re-suspend. When the ratio is greater than 2.4 and less than 25, a fraction of the deposited sediment will re-suspend and the amount of re-suspended sediment can be calculated from the *erosion function* that was determined from the erosion experiments. When the bed shear stress is in excess of 25 times the critical shear stress for deposition, all of the deposited sediment will re-suspend. For complete details of the FINESED model refer to Krishnappan (1997).

To apply the model, the bed shear stresses in each reach to be modeled have to be determined or estimated. This can be done in a number of ways depending on the required precision and the availability of river geometry and hydraulic data.

In the case of the Athabasca River study where a similar model was developed, Golder Associates Ltd. (1996) used a set of "Regime Equations" that related flow velocity, flow depth and slope of the energy grade line to the flow rate. A better approach would be to use a hydrodynamic flow model such as MOBED, which works for non-cohesive sediments (Krishnappan 1981) to calculate the bed shear stresses as a function of distance along the river and time for different flow hydrographs. The spatial and temporal variation of bed shear stress will give rise to different modes of fine sediment transport in different parts of the river. For example, near the banks of the river where the bed shear stress value is close to zero (because of shallower depths), deposition of fine sediment will occur for all flows. Under low flows during winter months, when the flow may be covered with ice, the bed shear stress can be in the range for which partial or full deposition of sediment could occur over the whole width of the river. During high flows in spring, when the bed shear stress can exceed the critical shear stress for erosion, partial or full erosion of the previously deposited sediment could occur. Depending on the magnitude of the flood event, complete erosion of the deposited sediment and subsequent transport of sediment in suspension through the river system is a possibility.

Other required input parameters for the model are the suspended sediment concentration for a chosen time interval at the upstream boundary of the modeled reach, the lateral input of fine sediment from tributary inflows, and the bank erosion in the reach being modeled. As discussed earlier, concentrations must be synthesized from real data and interpolated for the reach in question as no consistent relationship was found between present flow and suspended sediment load. In the absence of real time data, scenarios of expected sediment concentrations could be generated using sediment and flow data from a year with similar hydrograph characteristics to provide the sediment input parameter for modeling contaminant transport.

Recommendation for a Sediment and Contaminant Modeling Strategy in the Fraser River

Since the development of stable sediment rating relationships that would serve as the boundary conditions for the fine sediment transport model was not possible for the Fraser River, an alternate modeling strategy is recommended. This strategy would make use of existing coarse-grained sediment transport models such as MOBED (Krishnappan 1981), the new fine grained sediment transport model, FINESED, developed as part of the Fraser River Action Plan, and continued sediment sampling at a station forming the upstream boundary for the reach. For example, the upstream boundary for the reach from above Prince George to Marguerite would be at Hansard, which also has the benefit of an historical data set of sediment loads.

MOBED solves the flow and sediment mass balance equations and, thus, is capable of calculating the bed shear stresses. It also calculates the transport rates of fine sand, coarse sand and gravel fractions and changes in riverbed elevations due to erosion and deposition of sediment as a function of time and distance along the river for a specified flow hydrograph at the upstream and downstream boundaries. Using the results of the coarse-grained sediment model and the measured sediment data at Hansard as the upstream boundary condition, the new fine sediment transport model can be run to predict fine sediment concentration as a function of time and distance along the modeled river reach. Such predictions can then be used to improve the accuracy of EcoFate (a contaminant fate model developed for the Fraser and Thompson rivers; see Gobas *et al*. 1999). Because the EcoFate model assumes simultaneous erosion and deposition of sediment for all flows, it is possible that it might under-predict the sediment deposition in low flows and over-predict the same in high flows.

By adopting our recommended modeling strategy, a number of issues that were identified in this chapter could be addressed. For example, MOBED allows us to predict the long term storage or erosion of fine sediment due to the aggradation process of the streambed and bank erosion. FINESED, on the other hand, can account for the short term storage of fine sediment due to flocculation and settling on the riverbed during low flow periods. Finally, the lack of a sediment rating relationship can be overcome by using the measured sediment data from an upstream sediment monitoring station; data used should be those acquired in years with hydrological conditions similar to the one being modeled.

SUMMARY AND CONCLUSIONS

The Fraser River is not as “muddy” or sediment-laden as generally believed, especially in comparison to other northwestern rivers. In addition, the suspended sediments are dominated by silt-sized primary particles which, in combination with the relatively low sediment load, may mean that these sediments are not as important in contaminant transport as formerly thought. However, the dynamics of coarse-grained sediment deposition and erosion could provide temporary “refugia” for the fine sediments that become contaminated as they pass point sources or that originate from contaminated sources (*e.g.* particulates in effluents).

The analysis of the existing data showed that fine sediment concentrations cannot be predicted from flow rate in the Fraser River system. There are many possible explanations for the lack of a relationship, chief among them being that the amount of sediment stored on the riverbed during autumn and winter is not determined by the flow rate in the subsequent spring and summer.

New data, which consist of size distributions of fine sediment in the river and transport characteristics of fine sediment measured in a laboratory flume show that fine sediment in the Fraser is transported in a flocculated form and is likely to deposit on the riverbed during low flow periods. These data allow the formulation of a fine sediment transport model that is more realistic than existing non-cohesive sediment transport models. Unfortunately, this model could not be applied during this study because of the lack of predictability of the sediment load and the requirement of reach-specific shear-stress-flow relationships.

A new modeling strategy for the future is proposed that would involve the use of MOBED, an existing coarse grained sediment transport model, and the new fine grained sediment transport model, FINESED, developed during the course of the present study. These models, along with the sediment data from continuous sediment monitoring stations, can be used to predict the fine sediment concentrations in specific reaches. In addition to monitoring suspended sediments in the Fraser main stem at Hansard, Marguerite and Mission, suspended sediment measurements would be required on principal tributaries that deliver significant loads of fine sediment to the main stem.

These results, in turn, can then modify the calculated partition of contaminants between the water column, suspended sediments and bed sediments, which is needed to assess and predict contaminant exposure of fish and benthic invertebrates. Such a modeling strategy will improve the predictive capability of the EcoFate model developed for the Fraser and Thompson River system. Unfortunately, as there are no historical suspended sediment measurements on the major tributaries and monitoring at the main stem stations has recently been terminated, implementing this strategy cannot proceed without new resources.

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3.4

SEDIMENT TRANSPORT PATTERNS IN THE LOWER FRASER RIVER AND FRASER DELTA

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The Fraser River delivers an estimated load of 17×10^6 t of sediment to the Fraser River delta and Strait of Georgia each year (McLean and Tassone 1991). This sediment load and its distribution are important in sustaining the ecology of the delta and estuary. For example, the tidal flats of Sturgeon and Roberts banks, historically formed from Fraser River sediments, have international ecological importance, particularly as bird and fish habitat (Fraser River Estuary Management Program 1994; Vermeer and Butler 1987). These deposited sediments also provide an indicator of this environment's condition through the concentration of contaminants in the sediment (Brewer *et al.* 1998; Swain and Walton 1991, 1993, 1994). Although important for its sustenance of fish and wildlife habitat, sediment deposition presents a concern for maintaining shipping access to the lower river. Thus, since the turn of the century, there has been an ongoing program to maintain a navigable shipping channel in the estuary through the removal of sediment from the river channels by dredging (Neu 1966).

An understanding of sediment transport and its distribution is important in managing the estuary and delta. Therefore, we conducted a study to investigate net sediment transport patterns in the Fraser River Estuary and delta foreshore. The specific objectives were: (1) to establish the pattern of sand movement, for guidance in determining optimum areas for dredging and for depositing dredged sands; and (2) to determine the best sampling sites for monitoring contamination in Fraser Estuary sediments by developing knowledge of the movement and deposition areas of fine sediments. These sediments are preferred for contaminant sampling because of the greater surface area for contaminant adsorption (Karickhoff *et al.* 1978).

In this study, net sediment transport patterns were established using a technique known as a Sediment Trend Analysis (STA™). First described in McLaren and Bowles (1985), this approach measures relative changes in grain-size distribution of the existing sediments. The derived patterns of transport are, in effect, an integration of all processes responsible for the erosion, transport and deposition of sediments over the time period required to form the deposits. In addition, the analysis estimates the probability for transport of each grain size, thus describing the behaviour of the sediments in the environment.

This largely qualitative approach (it is unable to establish rates of transport or deposition) has been used elsewhere to: (1) evaluate and direct numerical models (Van Heuvel *et al*. 1993; McLaren *et al*. 1993a); (2) determine the behaviour of sediments at dredged material disposal sites (McLaren and Powys 1989); (3) predict the build-up and dispersal of contaminated sediments (McLaren *et al*. 1993b; Little and McLaren 1989); and (4) understand the sediment and process interrelationships among natural marine and coastal environments (De Meyer and Wartel 1988).

METHODS

In the pre-freshet period between February 9 and April 7, 1993, 1,488 surface sediment samples were collected from the Fraser River, from its confluence with the Pitt River (including 1 km up the Pitt River) to the 200-metre bathymetric contour in the Strait of Georgia. The northern and southern boundaries of sampling were Point Grey and Point Roberts. A Shipek Grab was used to sample the top 10 to 15 cm of surface sediment. The intertidal flats (Roberts and Sturgeon banks) were sampled with a trowel from a hovercraft. A representative sub-sample (about 100 g) from the grab was stored in a sealed plastic bag and shipped to the GeoSea (UK) office in Cambridge for grain-size analysis.

In the river, sampling was carried out on a series of transects spaced 500 m apart. On each transect, collection sites were equally spaced at about 100 m apart. Where the river was narrow, a sample was collected in the centre of the main channel and midway from the centre to the bank on either side. On the intertidal banks, samples were collected on a regular grid of 500 m spacings; in the offshore the interval was increased to 1 km.

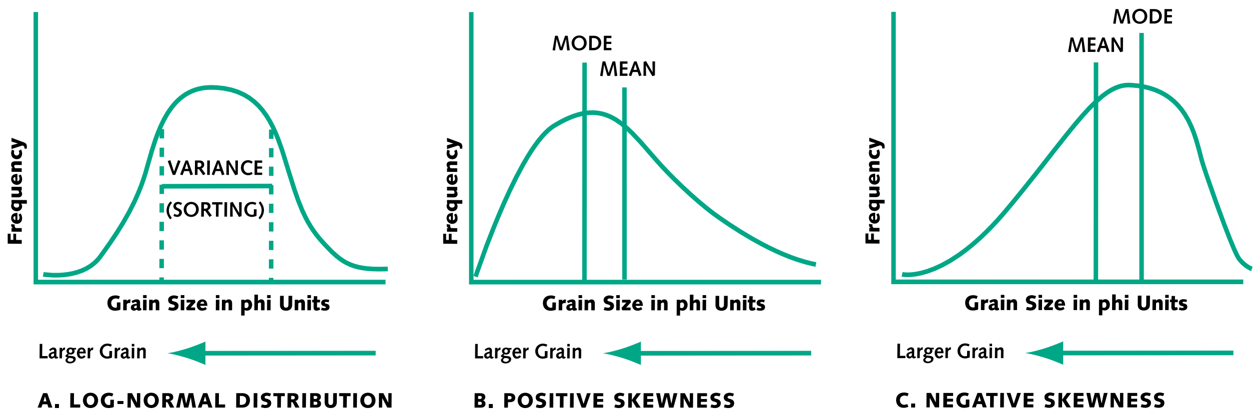
The grain-size distribution of each sample was determined by sieving sediment (ranging from -2 phi [ϕ] to 0ϕ [4 mm to 1 mm] grain-size diameter) at 0.5ϕ intervals. A Malvern laser particle-size analyzer was used to determine the particle size for the range between -0.85ϕ to 10.0ϕ (1.8 mm to 1.0 μm) grain-size diameter following a standardized technique developed by GeoSea Consulting Ltd. (McLaren and Ren 1995). Where both size fractions were present, the sieve and laser data were merged to form one complete distribution.

Sediment Trend Analysis

Details of the theory are described in McLaren and Bowles (1985) and McLaren and Ren (1995). The approach is summarized here.

The Sediment Trend Analysis examines the relationship in grain-size distribution among adjoining sediment samples. Grain sizes of most sediment deposits are characterized by a log-normal distribution, as illustrated in Figure 1a. Deviations from this distribution are caused by erosion, transport and deposition and result in skewness from the log-normal distribution (Figure 1b, 1c). The Sediment Trend Analysis model shows that, as sediment erodes, the selective removal of fine-sized particles occurs in such a manner that the grain-size distribution of the lag, or remaining deposit, is coarser, better sorted and more positively skewed (*i.e.*, the median particle size is larger than the mean) than the original deposit.

The model builds a relationship between two hypothetical sediment samples (D_1 and D_2) which are taken sequentially in a known transport direction (for example from a riverbed where D_1 is the up-current sample and D_2 is the down-current sample). The theory shows that the sediment distribution of D_2 may become



Phi units are based on a negative log scale.

Figure 1. *Illustration of grain-size distribution of sediment deposits.*

finer or coarser than D_1. If it becomes finer, the skewness of the distribution must become more negative. If D_2 is coarser than D_1, the skewness must become more positive. Over time, sediment at both sites will be sorted into more uniform particle sizes. If either of these two trends is observed, we can infer that sediment transport is occurring from D_1 to D_2. However, if D_2 is finer, better sorted and more positively skewed than D_1, transport cannot be from D_1 to D_2 and we cannot suppose that transport between the two samples has taken place. Under this scenario D_2 must be more negatively skewed than D_1 for transport to have occurred from D_1 to D_2.

In the above example, where we are already sure of the transport direction, $D_2(s)$ can be related to $D_1(s)$ by a function X(s) where 's' is the grain size. The distribution of X(s) may be determined by:

$$X(s) = D_2(s)/D_1(s)$$

X(s) denotes the statistical relationship between the two deposits and its distribution defines the relative probability of each particular grain size being eroded, transported and deposited from D_1 to D_2.

Initially, a trend is easily determined using a statistical approach whereby, instead of searching for "perfect" changes in a sample sequence, all possible pairs contained in the sequence are assessed for possible transport direction. When one of the trends exceeds random probability within the sample sequence, we infer the direction of transport and calculate X(s).

To analyze for sediment transport directions over two dimensions, a grid of samples is required. Each sample is analyzed for its complete grain-size distribution and these are entered into a computer equipped with appropriate software to search for statistically acceptable trends.

Empirical examination of X-distributions from a large number of different environments has shown that four basic shapes are most common when compared to the D_1 and D_2 distributions. These are as follows:

(1) Dynamic Equilibrium: The shape of the X-distribution closely resembles the D_1 and D_2 distributions. The relative probability of grains being transported, therefore, produces a similar distribution to the actual deposits. This suggests that the probability of finding a particular grain in the deposit is equal to the probability of its transport and redeposition (*i.e.*, there is a grain-by-grain replacement along the transport path). The bed is neither accreting nor eroding and is, therefore, in dynamic equilibrium (Figure 2a).

(2) Net Accretion: The shapes of the three distributions are similar, but the mode of X is finer than the modes of D_1 and D_2. The particle size becomes finer in the direction of transport; however, more fine grains are deposited along the transport path than are eroded, with the result that the bed, though mobile, is accreting (Figure 2b).

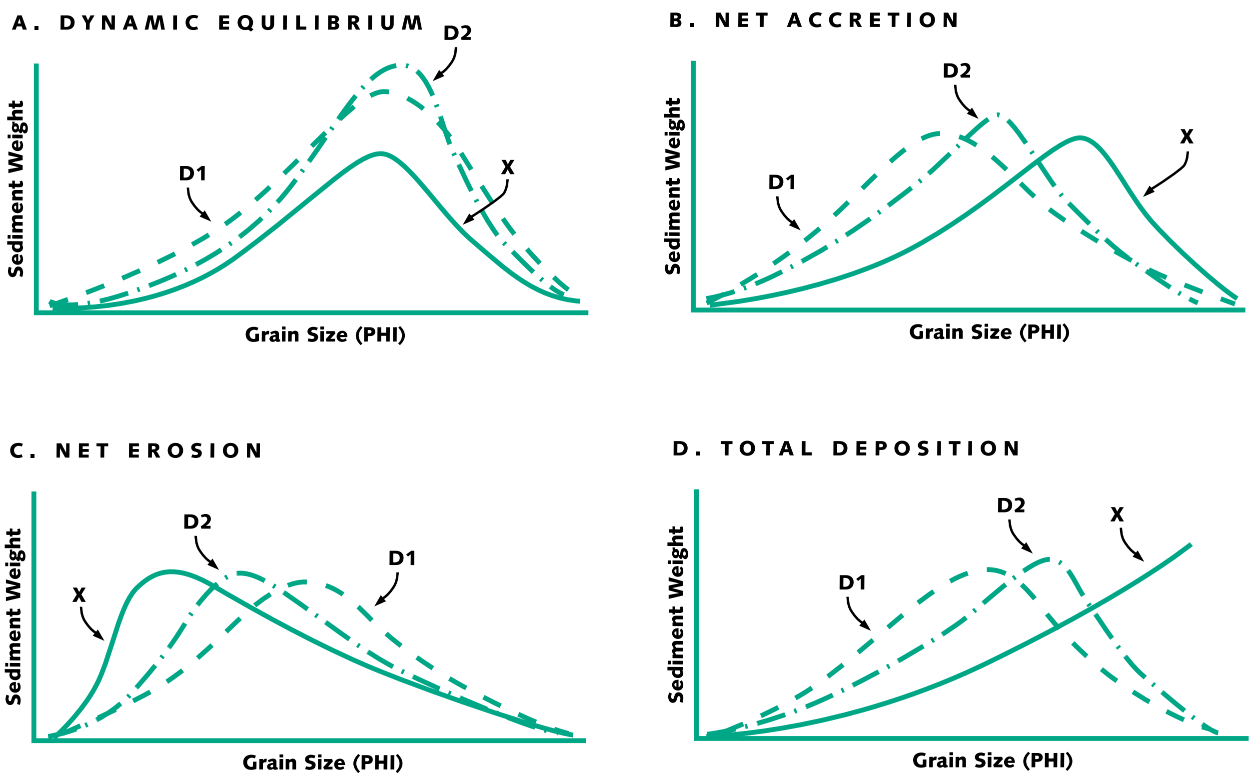


Figure 2. *Summary of interpretations given to shapes of X-distributions relative to the D_1 and D_2 deposits.*

(3) Net Erosion: Again the shapes of the three distributions are similar, but the mode of X is coarser than the D_1 and D_2 modes. Sediment coarsens along the transport path, more grains are eroded than deposited and the bed is undergoing net erosion (Figure 2c).

(4) Total Deposition: Regardless of the shapes of D_1 and D_2, the X-distribution more or less increases monotonically over the complete size range of the deposits. The particle size must become finer in the direction of transport; however, the bed is no longer mobile. Rather, it is accreting under a “rain” of sediment that becomes finer with distance from source. Once deposited, there is no further transport (Figure 2d).

A multiple correlation coefficient (R^2), defining the relationship among the mean, sorting and skewness in the sample sequence, was calculated. The R^2 gives a relative indication of how well the samples are related by transport. If a given sample sequence follows a transport path perfectly, R^2 will approach 1.0 (*i.e.*, the sediment samples are perfectly related by transport).

RESULTS AND DISCUSSION

Separate transport trends were established for two sediment types: sand (samples composed of 50% or more sand[1]) and mud (samples composed of 50% or more mud; mud[2] is defined as silt + clay). The sand comprises the majority of the data base (65% of the samples) and is found mainly in the river and on the tidal flats (Figure 3).

[1] ***Sand:*** *Particle Diameter from 4ϕ to 0ϕ (62.5 µm to 1 mm)*

[2] ***Mud:*** *Particle Diameter >4ϕ (<62.5 µm)*

Figure 3. Sediment types in the lower Fraser River and Delta.

A total of 175 transport lines were selected to describe the pattern of sand transport and 67 transport lines were used for describing mud transport. Details of sand and mud trends are presented in McLaren and Ren (1995). Figure 4 presents a summary of the major transport patterns detected for sand. Figure 5 presents the trends for mud.

River Sections

In the Fraser River, the derived sediment transport patterns show general net movement of both sand and mud from the upstream end of the study area to all the exits into the Strait of Georgia. This follows the classic delta morphology associated with the river. Exceptions are upstream transport, or reversals in a number of channels (Table 1). Upstream movement of sediments towards Pitt River is shown in the Coquitlam/Pitt River area. The evidence for such movement is supplied by the formation of a "negative delta" at the south end of Pitt Lake, the result of flood tide currents that are stronger than the combined ebb stream and river flow (Thomson 1981). No specific evidence in the literature could be found to directly support any of the other reversals.

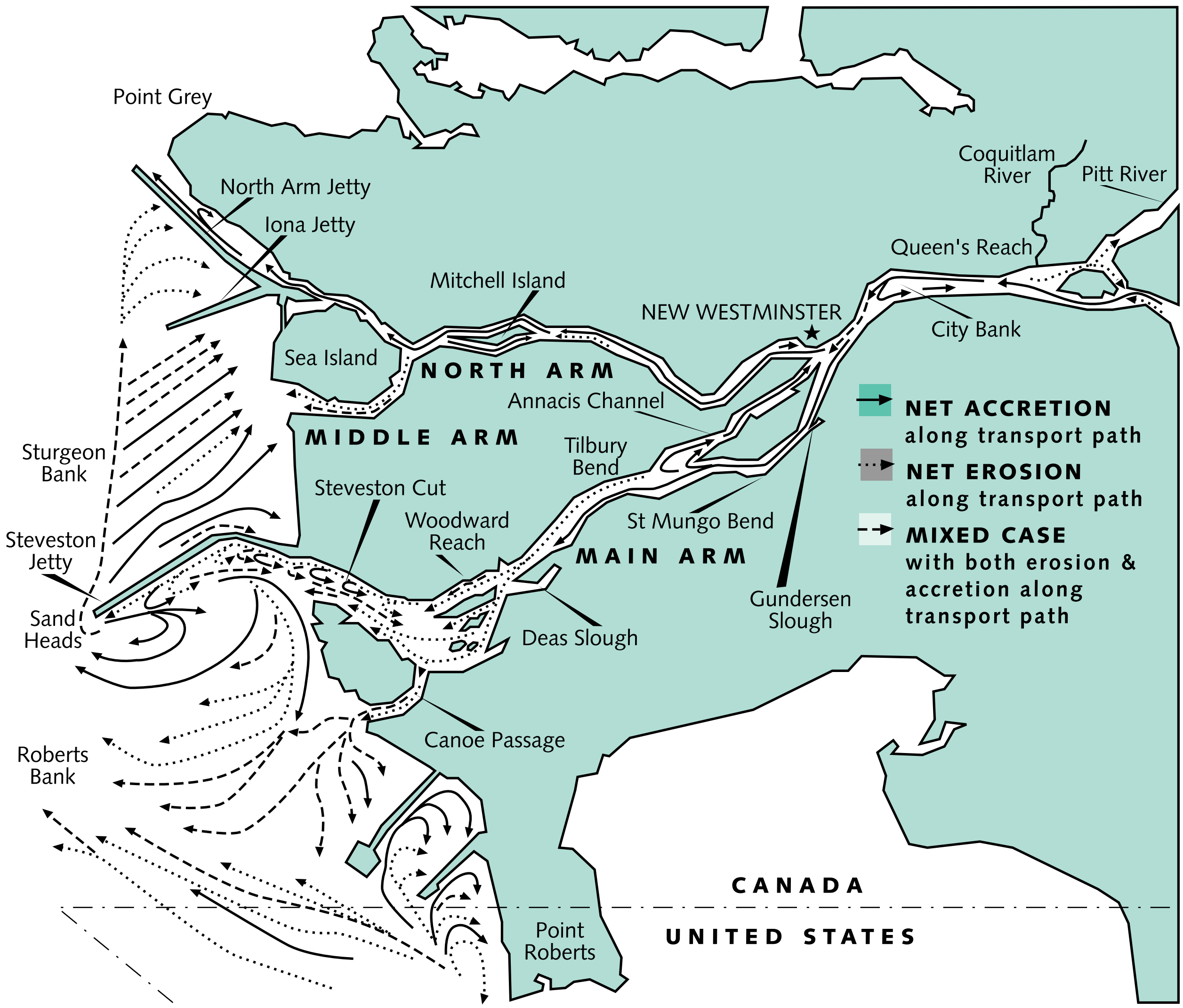


Figure 4. *Sediment Transport Patterns for Sand in the lower Fraser River and Delta.*

In some instances, the observed reversals coincide with areas requiring major dredging. Ferguson (1991) has identified the North Arm immediately east of Mitchell Island as one such region. Here, the South Mitchell Island upstream transport regime meets the eastern North Arm downstream transport regime, providing conditions that should increase sedimentation. In Steveston Cut, there is a flow reversal that meets the Woodward Reach transport regime. When river transport dominates over flood transport, high deposition could be expected in Steveston Cut. When the situation is reversed, the deposition would shift to Woodward Reach. Prior to the late 1960s, extensive dredging was required to address channel instability in the trifurcation area at New Westminster. In this region, downriver transport bifurcates into the North and Main Arms, but is “hampered” by an opposing flow out of Annacis Channel. Since the construction of training walls, this area is now largely self-scouring.

Only in Queen’s Reach, southeast of City Bank, did the delineation of a flow reversal fail to coincide with an area of high deposition. There is a dredge disposal site in this part of the river (Ferguson 1991) and apparently this is an area used extensively for borrow dredging (Stepchuk 1993, pers. comm.). Such activities could have two possible effects on the sediment trends. The first is that dredging and dumping have

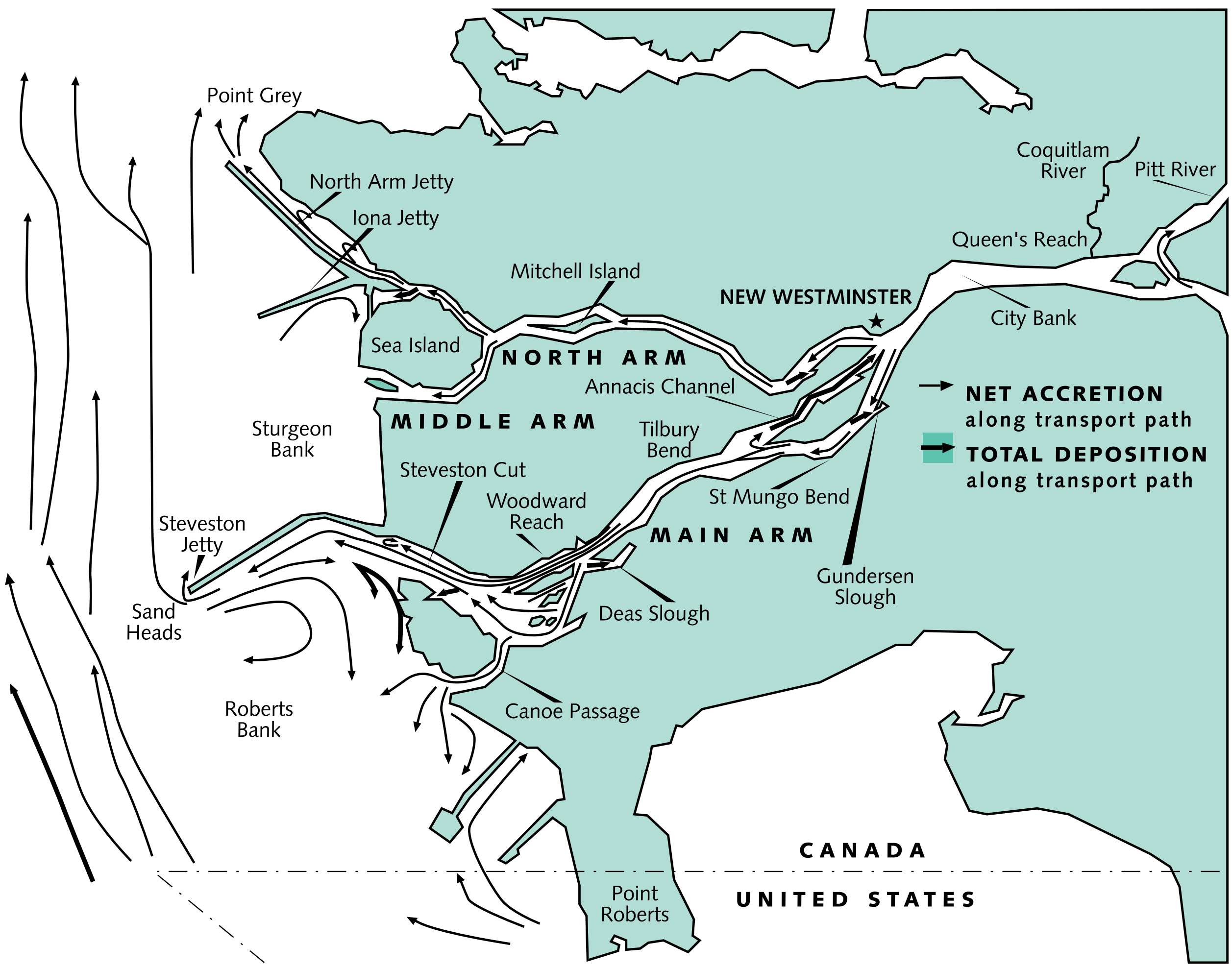


Figure 5. Sediment Transport Patterns for mud in the lower Fraser River and Delta.

Table 1. Lower Fraser River reaches demonstrating upriver sediment transport patterns as detected by the Sediment Transport Analysis. R^2 denotes the multiple correlation coefficient defining the relationship among the mean, sorting and skewness in the sample sequence. SD denotes standard deviation.

SITE	SAND			MUD		
	# of Sample transport lines	R^2 value		# of Sample transport lines	R^2 value	
		Mean	SD		Mean	SD
Coquitlam River/Pitt River	12	0.87	0.09	1	0.85	
Queen's Reach, south of City Bank	5	0.69	0.09	none		
Annacis Channel	2	0.76	0.03	2	0.94	0.04
Channel south of Mitchell Island	2	0.56	0.44	none		
Inside Steveston Cut—out to Jetty	30	0.92	0.05	none		
Deas Slough	none			1	0.87	
Gundersen Slough	none			1	1	
Cannery Channel[1]	none			2	0.86	0.007

[1] *located by Steveston Cut*

changed the natural sediment distributions sufficiently to produce apparent trends that are erroneous. The second is that the activities, particularly borrow dredging, have caused a transport reversal in the main channel south of City Bank. The relatively strong transport trend (mean R^2 = 0.69; Table 1), as detected by the sediment transport analysis used in this study, favours the second explanation.

The region of greatest conflict between known areas of sand deposition and the transport paths is in a stretch of river between Sand Heads and Woodward Reach. Seaward of Woodward Reach many of the trends were mixed, showing valid statistics for both erosion and deposition (accretion), a few were depositional and still others, particularly those adjacent to the Steveston Jetty, were erosional. In Woodward Reach, all the trends were erosional, with the exception of some mixed case trends along the right bank. The confluence of opposing transport regimes should result in areas of high deposition, as was evidenced in the North Arm east of Mitchell Island (discussed earlier). The lack of depositional trends for sand cannot easily be explained although the trends do show evidence for preferred erosion on the right bank inside of Steveston Jetty (Steveston Bend) and accretion on the left, a finding substantiated by Kostachuk and Luternauer (1987). A combination of extensive dredging in these channels as well as the presence of a dredge disposal site (PWC-3; Ferguson 1991) in Woodward Reach may have created erroneous X-distributions, despite the fact that transport trend statistics, particularly those along Steveston Jetty (mean R^2 = 0.92; Table 1), are excellent.

In comparison to sand, mud plays a minor role in the main river. Mud is generally found on the sides or banks of the main channels and the mud transport trends most often are associated with reversals into backwaters, such as sloughs, where total deposition (*i.e.*, once deposited, there is no further transport) is observed. Such sites have been shown to accumulate contaminants associated with fine particles (Young *et al*. 1985). The identified sloughs and backwaters have been used to assess contaminant exposure in the Fraser Estuary (Brewer *et al*. 1998; Swain and Walton 1991; 1993; 1994).

Tidal Flats

Of the four tributaries that enter the Strait of Georgia, only two, Middle Arm and Canoe Passage, have a relatively unobstructed flow onto the tidal flats. Canoe Passage accounts for 14 per cent of the total Fraser River discharge (Acres 1984; Hay and Co. 1988), and the pathways for sand produce a dendritic pattern of transport that could be considered "typical" of a meandering transport regime across a delta tidal flat. Through Canoe Passage itself, the trends for sand are indicative of high energy, erosive transport and on the delta flat nearly all the trends for sand produced a mixed case (*i.e.*, both deposition [accretion] and erosion). The statistics nearly all favour deposition, but the mix of the two cases and the eroding trends in Canoe Passage provide a signal that there may be a limited quantity of sediment reaching this area. Should sediment supply decrease in the future, the tidal flat in this area might easily change to net erosion.

Middle Arm accounts for a much smaller volume of discharge (about 3% or 4%). The amount of sand may be small as eroding trends occur inside Middle Arm; however there is accretion of mud. There appears to be no transport of sediment (sand or mud) out of the arm onto Sturgeon Bank.

The remaining two tributaries, North Arm and Main Arm, have been completely channelized across the tidal flats, with the result that the dendritic transport patterns for sand observed out of Canoe Passage cannot exist (Monahan *et al*. [1993] make a similar conclusion). In both arms, sand is carried to the extreme seaward edge of the tidal flats. In the Main Arm, which carries 70 per cent of the Fraser River discharge (Hay and Co. 1988), the sediment trends indicate three routes for sand to take. The first is a return system down the south side of Steveston Channel (on the inside of Steveston Jetty) which suggests that there is a constant recycling of sand in this region. The second route follows a clockwise gyre over northern Roberts Bank and the third transports sand northeast across Sturgeon Bank. Patterns of megaripples on Sturgeon Bank provide some support for the latter transport direction (Medley and Luternauer 1976).

There are distinct similarities in the transport of sediment on north Roberts Bank and Sturgeon Bank. For the transport pathways close to the Sand Heads source, the trends show net accretion. Farther away from source, the lines become either mixed case, or produce X-distributions indicative of net erosion. This suggests that the amount of sediment available for deposition is small. Accretion is occurring near source, but the trends soon show evidence for erosion.

There may be a fourth route for sand to be removed from the Sand Heads region, although it was not apparent from the samples obtained for the trend analysis. Evoy *et al*. (1993) described downslope slumping and gravity flow processes as a mechanism for sand to bypass the delta slope and to be deposited directly in prodelta and basinal environments, the deeper water environments associated with the Strait of Georgia. Sand lost from the river mouth in this way will no longer be available for deposition on the tidal flats.

Similar to the Main Arm distributary, sand out of the North Arm does not appear to be transported onto the delta foreshore (no sand samples were found in the sampling program). In this case, trends terminated at the Point Grey breakwater, and no further transport could be determined from the North Arm onto any of the tidal flats.

Transport pathways in Sturgeon Bank stopped at the Iona Jetty. A new transport regime started north of the Iona Jetty and extended to the North Arm Jetty. The trends in this area show net erosion that is consistent with the distance from the main source of sand at Sand Heads.

Transport pathways in southern Roberts Bank, in the vicinity of the Westshore Terminals Causeway and the Tsawwassen Ferry Terminal, appear to have no relationship with sediments derived from the Fraser River. The transport patterns show clockwise gyres with a sediment source from a northwest-trending nearshore regime. The trends for lines close to the low-water line indicate net erosion suggesting, once again, that there is little sediment available to maintain the present intertidal width.

Mud is relatively rare on the intertidal flats, being confined principally to the landward margins. Where transport trends could be determined, the derived patterns closely follow those for sand.

Delta Front and Offshore

All trends on the delta foreslope show transport to the north, parallel to the bathymetry, a finding in complete agreement with known processes. North flood currents are stronger than the ebb currents (Pickard 1956), and even surface currents affecting dispersal of the Fraser River plume favour northward transport (Thomson 1981). According to Luternauer *et al*. (1978), bottom currents on the foreslope are strong enough to transport sand-sized material and hydraulic bedforms have been observed on the sandy southwestern Roberts Banks slope. The sediment trends for these sands show net erosion which is supported by observations made in a PISCES submersible revealing erosional ledges and lag deposits (Luternauer and Finn 1983). Kostaschuk and Luternauer (1993) point out that there is no obvious source of sand replenishment and warn of slope failure. The trends indicate that the sand source must be the erosion of earlier deposits of delta foreslope sands, and it appears that such a process could have serious consequences for the stability of the Tsawwassen and Coal Port terminals. The rate of detected processes, such as erosion, cannot be determined by this analysis. As a result of this finding, a subsequent investigation was undertaken where it was found that the eroding trends may have formed in deposits disposed in the area during the construction of the Coal Port (Hay and Co. 1996).

The presence of the Fraser River and its associated plume are undoubtedly responsible for the mud deposits on the foreslope in depths greater than 50 m. It is not, however, primarily the cause for the northward direction of transport. Rather, particulate matter in the plume is carried both north and south of the outflow source at Sand Heads. On settling through the pycnocline, these particles become deposited from the transport regime that is under the influence of the dominant northward currents.

CONCLUSIONS AND RECOMMENDATIONS

In the Fraser River Estuary, the predominant pattern for sand and mud movement is from upstream to downstream; however, several reversals were detected. A close correspondence with known sites for sediment accumulation was noted.

The sediment trend analysis suggests that sand deposition over most of the intertidal flats is no longer the result of natural deltaic processes. Two of the four distributaries are channelized to the seaward edge of the intertidal flats (North and South Arms) and are restricted from meandering. Only through Canoe Passage do the trends follow the expected transport paths indicative of a delta formation. Here, as well as elsewhere on the banks, many of the trends show either a mixed case (erosion and accretion) or net erosion only, a finding that occurs when sediment supply is small. The evident paucity of sand on the tidal flats is attributed principally to channelization and removal of river sands by dredging, although the various causeways and jetties crossing the banks may also be a contributing factor. It is recommended that studies addressing the sediment budget and transport rates be undertaken to establish rates for processes observed in this study. Such studies would provide guidance regarding the impact of sediment removal on the tidal flat environment.

The evidence for a sediment return system along the inside of the Steveston Jetty (Steveston Bend) and in Steveston Cut may, in part, be caused by dredging activities in the river throughout the study area, which has lowered bed levels in recent years. In many estuaries it has been found that over-deepening increases the tidal range resulting in stronger flood currents (Jenson and Sieffert 1994). It is suggested that future engineering work along the inside of Steveston Jetty designed to decrease dredging requirements should take a landward return of sediments into account. It is not known if the same transport patterns take place during freshet.

The findings show erosion on the foreslope, which may be endangering the stability of the Roberts Bank Coal Port and Tsawwassen Ferry Terminal and are probably the result of the channelization of the Main Arm which effectively diverts sediment replenishment from this area to the Strait of Georgia. The method used in this study cannot predict the rate of erosion; thus it is recommended that the sediment transport processes be confirmed and their rates established for this area of Roberts Bank through a quantitative study.

The analysis shows that areas of total deposition for mud were observed in backwaters and sloughs in the river and at depths greater than 50 m in the foreslope. The river sites correspond to sites currently used by researchers for monitoring contaminant exposure in sediment. The results from this study confirm the suitability of using these sites for contaminant monitoring. A caveat is that dredging activities in some sloughs (Ferguson 1991) may disturb the distribution of contaminants associated with natural sedimentation.

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3.5

SEDIMENT QUALITY IN THE FRASER RIVER BASIN

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Many pollutants entering the aquatic environment are extremely water insoluble and, as a result, readily bind to sediment particles in the water column. There is abundant evidence of environmental degradation in areas where water quality criteria are not exceeded, yet organisms in or near sediments are adversely affected (Chapman 1989; US EPA 1994). Contaminated sediments may be directly toxic to aquatic life (Swartz *et al.* 1985) or, through bioaccumulation and biomagnification, can cause long term chronic effects. As the Fraser River carries a relatively high sediment load (8–26 million tons/year) (Stewart and Tassone 1989), sediments make an ideal medium for the geographical and temporal characterization of contaminants in the basin.

Prior to the Fraser River Action Plan (FRAP), studies of contaminants associated with sediments in the Fraser Basin were relatively few and of limited geographical coverage. In 1990 and 1991, Derksen and Mitchell (Draft 1994) measured dioxins, furans and chlorophenolics in suspended sediments from Marguerite and Lillooet on the main stem of the Fraser River. Bed sediments from the upper Fraser River and Thompson sub-basin were measured for dioxins and furans in 1988 (Mah *et al.* 1989) and for a number of trace organics associated with pulp mill effluents from 1989 to 1991 (Dwernychuk 1990; Dwernychuk *et al.* 1991). In the Fraser estuary, trace organics and metals were measured in bed sediments between 1985 and 1992 (FREMP 1996).

FRAP sediment assessments were conducted between 1992 and 1996 in order to determine the current status of the basin and to establish a baseline of trace organic and metal concentrations in suspended and bed sediments, upstream and downstream of pulp mills and major cities. Contaminants associated with suspended sediments represent one route of exposure to organisms in the water column, whereas contaminants associated with bed sediments represent a route of exposure to benthic and bottom feeding organisms. The principal advantage of suspended sediment sampling is that it provides an integrated sample over a known period of time with a high degree of reproducibility (Sekela *et al.* 1994; 1995), whereas bed sediment sampling is a cost effective method for characterizing contaminant exposure over a longer period of time (weeks to years).

METHODS

Suspended sediments were sampled under varying flow conditions in three consecutive years between 1992 and 1994 (Sekela *et al.* 1995) and in the spring and fall of 1996 (Sylvestre *et al.* 1998a; 1998b). Bed sediments were sampled in three consecutive years between 1994 and 1996 (Brewer *et al.* 1998). Suspended sediments were sampled using continuous flow centrifuges; clarified water from the centrifuge was sampled using solid phase extraction. Ekman grabs were employed for bed sediment sampling. Sampling locations were chosen to coincide as much as possible with previous studies, as well as with those used in the FRAP resident fish health assessment study (Raymond *et al.* 1999) and the FRAP benthic community study (Reynoldson and Rosenberg 1999). Refer to Figure 1 for sampling locations for bed and suspended sediment sampling and to Table 1 for sample site descriptions and sample sizes.

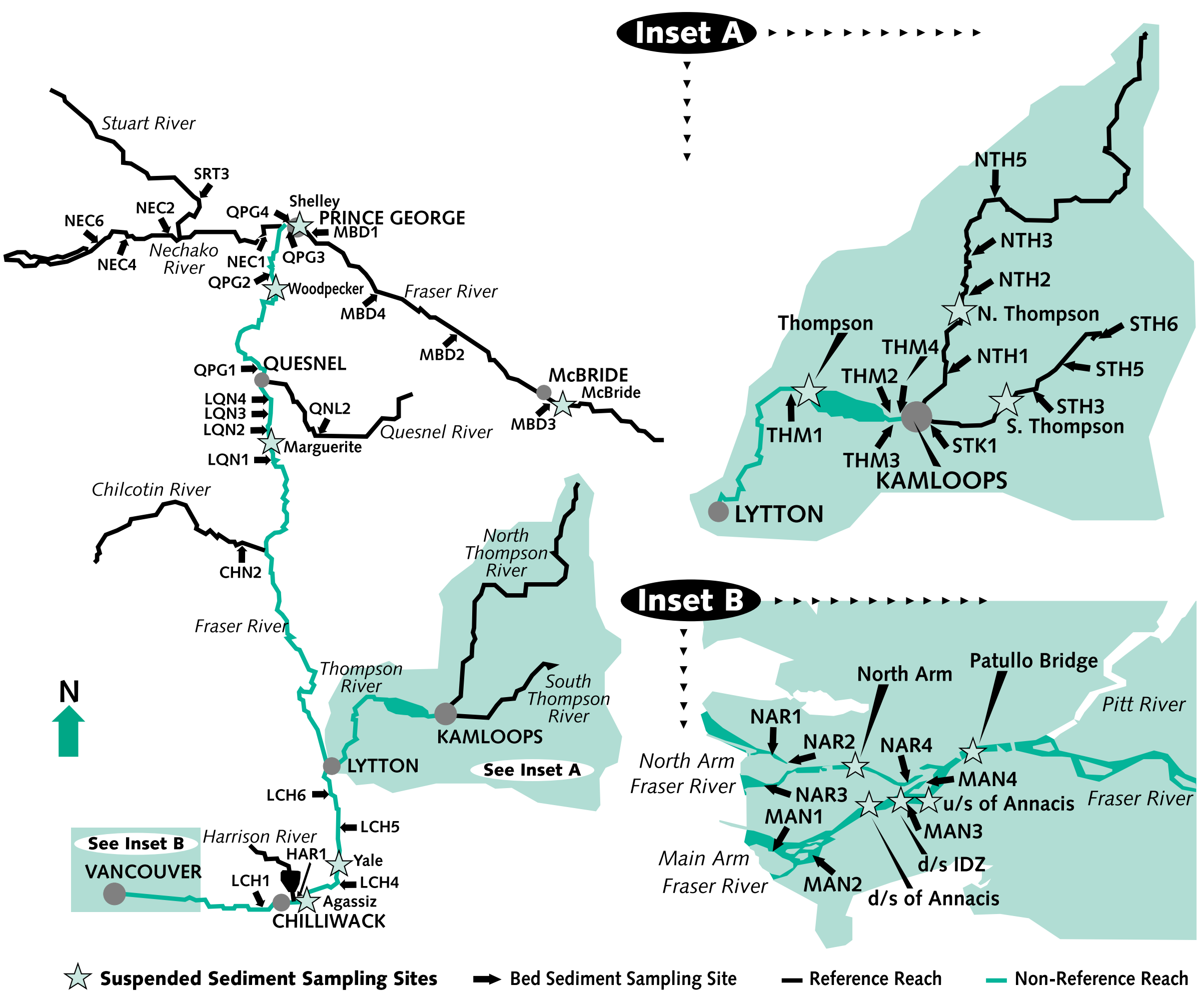


Figure 1. Fraser River Basin sediment sampling sites.

Table 1. Sediment sampling site/reach descriptions and sample sizes.

SAMPLE TYPE	REACH ID	GEOGRAPHICAL REGION	REACH DESCRIPTION	SITE/REACH TYPE	SAMPLE SIZE*
suspended	McBride	Upper Fraser	Fraser River 2 km upstream of McBride	reference	1
suspended	Shelley	Upper Fraser	Fraser River at Shelley	reference	4
suspended	Woodpecker	Upper Fraser	Fraser River at Woodpecker	non-reference	4
suspended	Marguerite	Upper Fraser	Fraser River at Marguerite ferry	non-reference	5
suspended	N. Thompson	Thompson sub-basin	N. Thompson River at McLure ferry	reference	3
suspended	S. Thompson	Thompson sub-basin	S. Thompson River 3 km upstream of McGreggor Creek at Lafarge Cement Plant	reference	1
suspended	Thompson	Thompson sub-basin	Thompson River at Steelhead Park near Savona	non-reference	4
suspended	Yale	Lower Fraser	Fraser River at Yale	non-reference	4
suspended	Agassiz	Lower Fraser	Fraser River at Agassiz Bridge	reference	1
suspended	Patullo Bridge	Lower Fraser	Fraser River Main Stem at Patullo Bridge	non-reference	1
suspended	Upstream of Annacis	Lower Fraser	Fraser River Main Arm at Alex Fraser Bridge upstream of Annacis STP	non-reference	1
suspended	Downstream of IDZ	Lower Fraser	Fraser River Main Arm immediately downstream of Annacis STP Initial Dilution Zone (IDZ)	non-reference	1
suspended	Downstream of Annacis	Lower Fraser	Fraser River Main Arm, 6 km downstream of Annacis STP	non-reference	1
suspended	North Arm	Lower Fraser	Fraser River North Arm at foot of Kerr Road	non-reference	1
bed	NEC	Upper Fraser	Nechako River	reference	4
bed	MBD	Upper Fraser	Fraser River from McBride to Prince George	reference	4
bed	QPG	Upper Fraser	Fraser River from Prince George to Quesnel	non-reference	4
bed	LQN	Upper Fraser	Fraser River from Quesnel to Lytton	non-reference	4
bed	SRT	Upper Fraser	Stuart River	reference	1
bed	CHN	Upper Fraser	Chilcotin River	reference	1
bed	QNL	Upper Fraser	Quesnel River	reference	1
bed	NTH	Thompson sub-basin	North Thompson River	reference	4
bed	THM	Thompson sub-basin	Thompson River	non-reference	4
bed	STH	Thompson sub-basin	South Thompson River	reference	3
bed	STK	Thompson sub-basin	South Thompson River at Kamloops	non-reference	1
bed	LCH	Lower Fraser	Fraser River from Lytton To Chilliwack	reference	4
bed	HAR	Lower Fraser	Harrison River	reference	1
bed	NAR	Lower Fraser	Fraser River North Arm	non-reference	4
bed	MAN	Lower Fraser	Fraser River Main Arm	non-reference	4

** sample size for suspended sediment samples denotes total number of samplings at each site from 1992–1996*
sample size for bed sediment samples denotes number of replicate samples collected per reach per year sampled

Sediments were sampled for several classes of contaminants associated with known sources (Table 2). Analysis of variance (ANOVA) was performed on bed sediment data to determine if significant differences exist between reaches. Organic parameter data were log transformed and analyzed by ANOVA followed by multiple comparison tests. If the assumptions for parametric tests were not satisfied, the data were analyzed by Kruskal-Wallis analysis of variance on ranks. Trace metals data were analyzed by analysis of covariance (ANCOVA), with the silt/clay fraction as the covariate. Principal components analysis (using the covariance matrix) was applied to per cent normalized data to identify geographical patterns within the data set.

Table 2. Contaminants measured in suspended and bed sediments—sources and effects.

CONTAMINANTS	MAJOR SOURCES	EFFECTS
Dioxins and furans	- pulp and paper mills using chlorine bleaching - incinerators - commercial chemicals (PCBs, pentachlorophenol, 2,4-D) - wood and fossil fuel combustion - sewage treatment plant effluents	- teratogenic - carcinogenic - acutely toxic - endocrine disrupting - bioaccumulative
Chlorophenolics	- pulp and paper mills using chlorine bleaching - wood treatment facilities/treated wood products - incinerators - chlorinated pesticides - sewage treatment plant effluents	- immunotoxic - fetotoxic - embryotoxic - fish tainting
Polycyclic Aromatic Hydrocarbons (PAHs)	- wood and fossil fuel combustion - creosote treated products - spills of petroleum products - slash burning - plant material - natural oil deposits	- carcinogenic - bioaccumulative
Chlorinated Pesticides	- agriculture - sewage treatment plant effluents - industrial effluents - global transport and deposition	- carcinogenic - endocrine disrupting - bioaccumulative
Polychlorinated Biphenyls (PCBs)	- transformers - lamp ballasts (pre-1980) - global transport and deposition - sewage treatment plant effluents - pulp and paper mill effluents	- immunotoxic - endocrine disrupting - bioaccumulative
Nonylphenol	- pulp and paper mills - textile processing and manufacturing - plastics manufacturing - leather processing - household cleaners - sewage treatment plant effluents	- acutely toxic - estrogenic - bioaccumulative
Trace Metals*	- mining and metallurgy - paints and dyes - electrical and electronic manufacturing - cleaning and duplicating - electroplating/finishing	- acutely toxic - endocrine disrupting - bioaccumulative

** some trace metals (e.g. iron, copper, zinc, manganese, selenium, cobalt, magnesuim) are required for normal biological function*

Insufficient replication did not allow for statistical analysis of the suspended sediment data. Organic parameter data for both bed and suspended sediments were normalized to one per cent organic carbon for the purposes of comparisons to guidelines, criteria and objectives. Toxicity equivalents (TEQs) for 2,3,7,8-tetrachlorodibenzo-para-dioxin (TCDD) (CEPA 1990) and PCBs (Ahlborg *et al.* 1994) were calculated with detection limits set at zero.

RESULTS AND DISCUSSION

Dioxins and Furans

Chlorinated dioxin and furan concentrations and 2,3,7,8-TCDD TEQs measured in suspended and bed sediments in the Fraser River Basin were generally elevated downstream of pulp mills and urban centres compared to reference sites. TEQs in suspended sediments were approximately double in the Fraser River Estuary, relative to the Agassiz reference site upstream of Vancouver, but similar to those in the Thompson River downstream of the pulp mill in Kamloops. TEQs in bed sediments were higher in the North Arm of the Fraser River in comparison to all other reaches in the basin, likely due to the large amounts of stormwater entering this reach of the river. No differences were found in dioxin concentrations measured in suspended sediments upstream and downstream of the Annacis Island Sewage Treatment Plant (STP), indicating that the STP is not likely a significant source of dioxins and furans to the Main Arm of the Fraser River.

Congener profiles in bed and suspended sediments were dominated by hepta- and octa-chlorinated dioxins and furans suggesting combustion and/or pentachlorophenol (PCP) sources. Since combustion and PCP source profiles are very similar, both are possible sources. However, the most pristine reference reaches (Chilcotin and Quesnel), which were least likely to have been affected by historical PCP use, exhibited a congener profile composed solely of dioxin congeners. Such a profile has not been associated with a PCP source (Czuczwa and Hites 1996). The South Thompson River site at Kamloops had a congener profile consistent with PCB contamination (Grundy *et al.* 1997). Dioxin and furan TEQs measured in bed sediments exceeded the interim federal sediment quality guideline of 0.25 pg/g TEQ at 29 of 44 bed sediment sites sampled from the basin, ranging from 0.002–8.75 pg/g TEQ, with the greatest exceedances occurring in the Thompson River and in the North Arm of the Fraser River (Table 3). The most toxic dioxin congener, 2,3,7,8-TCDD, exceeded these guidelines at three locations in the Fraser Basin: the Thompson River downstream of the pulp mill and in two sloughs in the North Arm of the Fraser River.

The exceedance of TEQ guidelines at reference locations, where non-point source combustion is a likely source of dioxins and furans, suggests that it may not be possible to meet the current interim federal guideline for 2,3,7,8-TCDD TEQ (CCME, Draft 1995). With the exception of the North Arm, TEQ concentrations in bed and suspended sediments were similar to those found in the Columbia River (Bortleson *et al.* 1994) and the northern rivers of Alberta (Crosley 1996a; 1996b; Pastershank and Muir 1995).

Dioxin and furan concentrations measured in suspended sediments from the upper Fraser and Thompson rivers varied seasonally, with higher concentrations in the winter base-flow period relative to the fall low-flow period. In the Fraser River, this concentration peak was attributed to a higher proportion of contaminated sediments in the river during base flow period when levels of natural erosion-derived suspended sediment are at their yearly lowest. In the Thompson River, this pattern was associated with inverse thermal stratification in Kamloops Lake during the limnological winter.

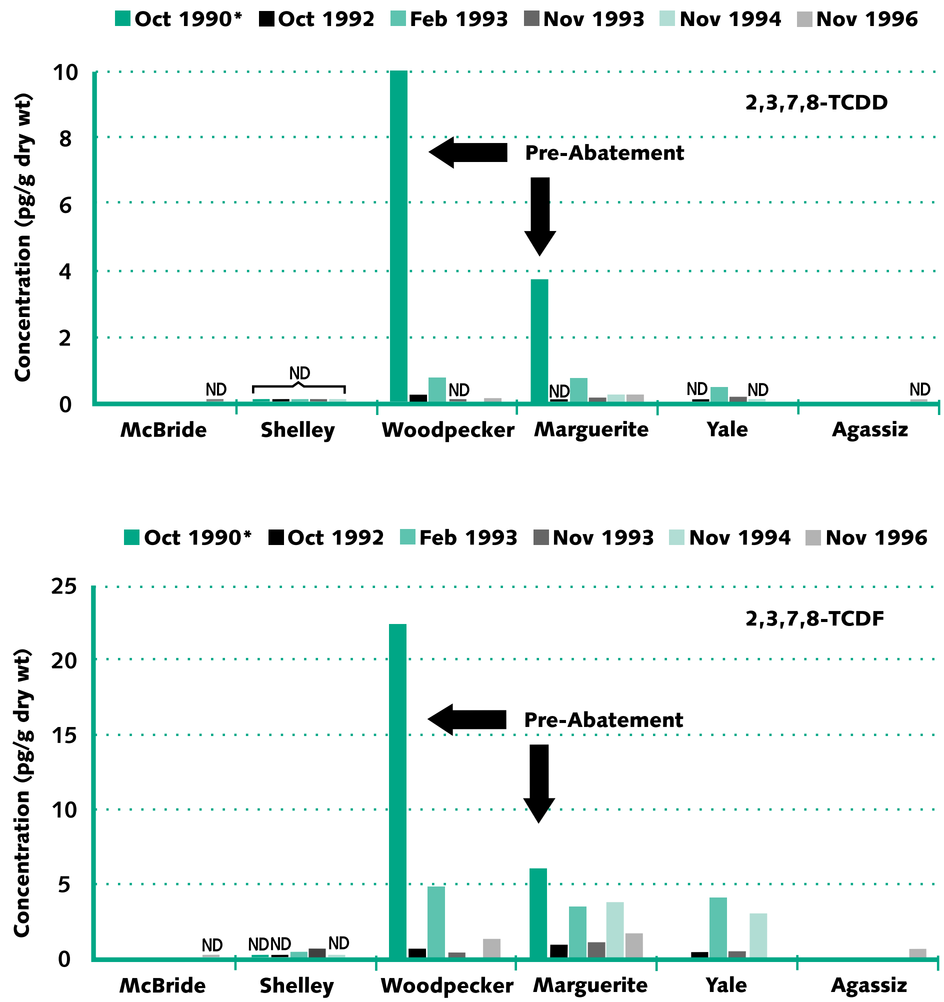
Temporally, concentrations in the upper Fraser River remained relatively constant between 1992 and 1996. A notable reduction in dioxin and furan concentrations was measured in the Thompson River at Savona in 1996 in comparison to 1993, when a failure in the effluent containment pond at the pulp mill in Kamloops resulted in a measured 2,3,7,8-TCDD TEQ of 7.6 pg/g at this site. However, the most dramatic reductions in dioxin and furan concentrations in suspended sediments occurred after the implementation of pulp mill

Table 3. Organic contaminants in bed sediments exceeding guidelines, criteria or objectives for the protection of aquatic life (1994–1996).

PARAMETER	NECHAKO RIVER	MCBRIDE TO PRINCE GEORGE	PRINCE GEORGE TO QUESNEL	QUESNEL TO LYTTON	STUART RIVER	CHILCOTIN RIVER	QUESNEL RIVER	NORTH THOMPSON RIVER	SOUTH THOMPSON RIVER	SOUTH THOMPSON RIVER AT KAMLOOPS	THOMPSON RIVER	LYTTON TO CHILLIWACK	HARRISON RIVER	MAIN ARM	NORTH ARM
Dioxins & Furans															
TEQs	F	F	F	F		F		F	F	F	F	F	F	F	F
2,3,7,8-TCDD											F				F
PAHs															
Naphthalene	P			P			P	P		P	P	P		P	P,O
Phenanthrene									F,P	F,P	F,P			F,P,O	F,P,O
Fluoranthene										F				F	F
Pyrene										F	F			F	F,O
Benz(a)anthracene									F					F,O	F,O
Chrysene										F	F			F	F
Benzo(a)pyrene										F,P	F			F,P,O	F
Dibenz(ah)anthracene														O	
PCBs															
Total PCBs				P						P					O
Aroclor 1254										P					
Aroclor 1260										P					P
Pesticides															
γ-HCH	F	F	F	F,P								F,P		F	F
β-HCH											P				
Total HCH			P	P											
Total Chlordane	P		P												
p,p'-DDE											F,P				
Total DDT					F,P									F,P	
Total DDT + metabolites					P						P			P	P

F indicates exceedence of Interim Federal Guideline (CCME, Draft 1995; Smith et al. 1996)
P indicates exceedence of Provincial Criteria (BC MELP 1995)
O indicates exceedence of Regional Objectives (Swain et al., Draft, 1996)

abatement measures consisting of the substitution of molecular chlorine with chlorine dioxide. Concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF were approximately 95–99 per cent lower between 1992 and 1996, relative to those measured in 1990 prior to the initiation of chlorine dioxide substitution by the mills (Fig. 2). Similarly, concentrations of furans in bed sediments were approximately three orders of magnitude lower than those measured downstream of pulp mills in 1988 in the upper Fraser and Thompson rivers (Mah *et al.* 1989). Declines in TCDD and TCDF concentrations relative to pre-abatement levels were also observed in whitefish muscle tissue in the upper Fraser and Thompson basins (Raymond *et al.* 1999).



* October 1990 data provided by George Derksen, Environment Canada   ND Denotes Below Detection Limit

Figure 2. 2,3,7,8-TCDD and 2,3,7,8-TCDF in Fraser River suspended sediment (1990–1996).

Dioxin and furan partitioning to the suspended solid phase was highly variable and was strongly influenced by the sediment organic carbon (TOC) content. Log K_{ocs}, derived from field measurements, ranged between 4.3 to 8.4, and were generally in the range of published values (Mackay *et al.* 1992).

Chlorophenolics

Chlorophenolics measured in suspended and bed sediments were higher in concentration downstream of pulp mills relative to reference sites throughout the basin. Chloroguaiacols, chlorocatechols and chlorovanillins, all associated with pulp mill effluents, were found at higher concentrations than chlorophenols, traditionally associated with wood preservation. Generally, total chlorocatechols and chlorovanillins were higher in reaches downstream of pulp mills relative to reference reaches in the upper Fraser River, while total chloroguaiacols and chlorovanillins were elevated in the Thompson River downstream of the pulp mill at Kamloops.

All classes of chlorophenolics measured in bed sediments, including the chlorophenols, were elevated in the North Arm of the Fraser River relative to the Main Arm and the Lytton to Chilliwack reach upstream of the metropolitan Vancouver area. Total chlorophenolics measured in the Stuart River were similar to levels measured in the urbanized North Arm of the Fraser River, indicating that chlorophenolic sources possibly related to wood preservation may exist in this reach. The Annacis Island STP did not contribute substantially to the chlorophenolic concentrations measured in suspended sediments in the Main Arm of the lower Fraser River, as evidenced by similar concentrations measured upstream and downstream of the STP.

Concentrations of chlorophenolics measured in bed sediments from the Fraser River Basin were generally lower than those reported in the Columbia River (Bortleson *et al.* 1994), in the northern rivers in Alberta (Crosley 1996a) and in the Apalachicola-Chattahoochee-Flint River system in Florida (USGS NAWQA 1997). No sediment quality guidelines or criteria for the protection of aquatic life exist for chlorophenolics.

As was the case for dioxins and furans, chlorophenolic concentrations in suspended sediments varied seasonally with higher concentrations during the winter base-flow period relative to the fall low-flow period. Temporally, chlorophenolics measured in suspended sediments were detected in fewer numbers and at lower concentrations in 1996 relative to 1992–1994, indicating that the switch to 100 per cent chlorine dioxide substitution (implemented at all kraft mills in the basin between 1994 and 1996) was effective in reducing levels of these compounds. Pentachlorophenol measured in bed sediments was more than an order of magnitude lower than that detected in the North Arm of the Fraser River in 1987 (Swain and Walton 1988), prior to the deregistration of this compound as an antisapstain chemical in sawmills. Similarly, pulp mill related chlorophenolics have decreased from levels measured by Dwernychuk (1990) prior to the initiation of pulp mill abatement measures.

Generally, chlorophenolics were associated with the sediment phase in the range of 1–20 per cent in the Fraser River, whereas in the Thompson River <1 per cent were associated with this fraction. A lower suspended sediment concentration in the Thompson River (<5 mg/L) in comparison to that of the Fraser River (0.9–226 mg/L) is believed to account for the lower partitioning to sediments in the Thompson River. Log K_{ocs}, derived from field measurements, ranged between 2.8 to 7.0 and were generally higher than theoretical values (Karickhoff 1981), indicating that chlorophenolics are partitioning to the suspended solid phase in greater proportions than expected.

Resin and Fatty Acids

Resin acids measured in suspended and bed sediments were elevated downstream of pulp mills in the upper Fraser River, when compared to reference sites. The presence of five pulp mills and numerous sawmills in the Prince George to Quesnel reach of the Fraser River likely accounts for the observed elevated levels of these compounds. Resin acids in bed sediments were significantly higher ($p<0.05$) in the Fraser estuary than at reference sites upstream of the greater Vancouver area, likely due to numerous sources including wood processing industries, log boom storage and hog fuels. Chlorinated resin acids were detected only at sites located downstream of pulp mills. Dehydroabietic and abietic acids, both major components of pulp mill effluents (Fox 1977), comprised the greatest proportion of total resin acids. Total resin acids in suspended sediment from the Fraser Basin exceeded 100 µg/g, which is 10 times higher than the total resin acid concentration reported for suspended sediment from the upper Athabasca River (Crosley 1996b).

Total fatty acids in suspended sediments from the upper Fraser Basin were three to four times higher at sites downstream of pulp mills than at reference sites. In contrast, few differences were observed in total fatty acid concentrations measured in bed sediments throughout the Fraser Basin, and occasional upward deviations were attributed to local decomposition of organic matter. Palmitic acid comprised the largest proportion of the total fatty acids measured in both suspended and bed sediments. Fatty acid concentrations measured in the Fraser Basin were generally several fold higher than in bed sediments measured in the Columbia and Athabasca rivers (Bortleson *et al.* 1994; Crosley 1996b).

PAHs

Polycyclic aromatic hydrocarbons (PAHs) in suspended and bed sediments were higher downstream of urban centres relative to reference sites in both the Fraser River main stem and Thompson sub-basin. Total parent PAH concentrations in suspended sediments were approximately six times higher in the Fraser River Estuary relative to the Agassiz reference site. In bed sediments, total parent PAHs were twice as high in the heavily industrialized North Arm as in the Main Arm of the Fraser River, reflecting the large number of stormwater inputs to this reach. PAH source signatures (Yunker and Macdonald 1995) indicate that reference reaches were generally dominated by petroleum PAH sources, likely of natural origin, while non-reference reaches were characterized by petroleum and pyrogenic sources. Retene and other plant-derived PAHs in suspended sediments were elevated in the Fraser estuary where log boom storage, sawmill leachate and hog fuels are probable sources. In the fall of 1996, total parent PAHs measured immediately downstream of the Annacis Island STP initial dilution zone (IDZ) were over twice as high as those from upstream of the plant.

A number of PAHs measured in bed sediments exceeded federal and/or provincial sediment quality guidelines and criteria and/or draft regional objectives for the protection of aquatic life (Table 3). The relatively large number of exceedances in the Fraser estuary reaches and at the Thompson and South Thompson urban sites suggests that aquatic life may be affected by these contaminants in the more urbanized parts of the basin. Nevertheless, total parent PAHs, ranging from 7–1,000 ng/g, were several orders of magnitude lower than those measured in the northern rivers of Alberta (Crosley 1996a) and rivers in the Great Lakes Areas of Concern (Bolattino 1993).

PAH concentrations in suspended sediments varied seasonally in the Thompson River with higher concentrations in the winter base-flow period relative to the fall low-flow period (Fig. 3). Higher ratios of perylene to the remainder of the parent PAHs, measured throughout the Fraser River Basin during fall low flow, compared to winter base-flow, points to a mostly sedimentary PAH source in the fall season compared to a largely anthropogenic source in the winter season. PAH concentrations did not vary temporally in either bed or suspended sediment between 1992 and 1996. PAHs in bed sediments from the Fraser estuary were similar in concentration to those measured by Swain and Walton between 1989 and 1992 (Swain and Walton 1990; 1991; 1993).

Phase partitioning measurements indicated that, generally, <20 per cent of low molecular weight PAHs (LPAHs) and 20–80 per cent of high molecular weight PAHs (HPAHs) were found in the suspended solid phase. The high affinity of LPAHs for the water phase appears to increase their bioavailability, based on the predominance of LPAHs in peamouth chub muscle from the lower Fraser River (Raymond *et al.* 1999). Field estimates of log K_{ocs}, ranging between 3.0–7.3, were higher than published log K_{ocs} for LPAHs, but similar to published values for HPAHs (Mackay *et al.* 1992). Phase partitioning was influenced by site specific environmental factors such as the total organic carbon content of sediments and the suspended solids concentration.

Pesticides

Organochlorine pesticides measured in suspended and bed sediments were detected in trace levels (<10 ng/g) throughout the Fraser River Basin. Among those most commonly detected were DDT, DDE, HCB, α and γ HCH, chlordane, mirex and endosulphan sulphate. Similar levels of organochlorine pesticides were detected in rivers in the Great Lakes Areas of Concern (Bolattino 1993) and in river basins studied in the USA (Ott 1997; Stephens and Deacon 1997; Tate and Heiny 1997; Tornes *et al.* 1996; USGS NAWQA 1997). Both suspended and bed sediment data, in general, did not show a consistent pattern in pesticide concentrations between reference and non-reference sites in the upper Fraser and Thompson regions.

DDE in suspended sediments was detected in the Thompson River in concentrations four times higher than in the upper Fraser River. The same DDE pattern was observed by Raymond *et al.* (1999) in muscle from peamouth chub and mountain whitefish sampled in the same reaches. DDE+DDD/DDT metabolite ratios (Sanchez *et al.* 1993) indicated that in the upper Fraser reaches the DDT source may be more recent, perhaps originating from atmospheric deposition of globally transported pesticides. A less recent source was indicated for the Thompson reaches, where it may be linked to historical use in the region.

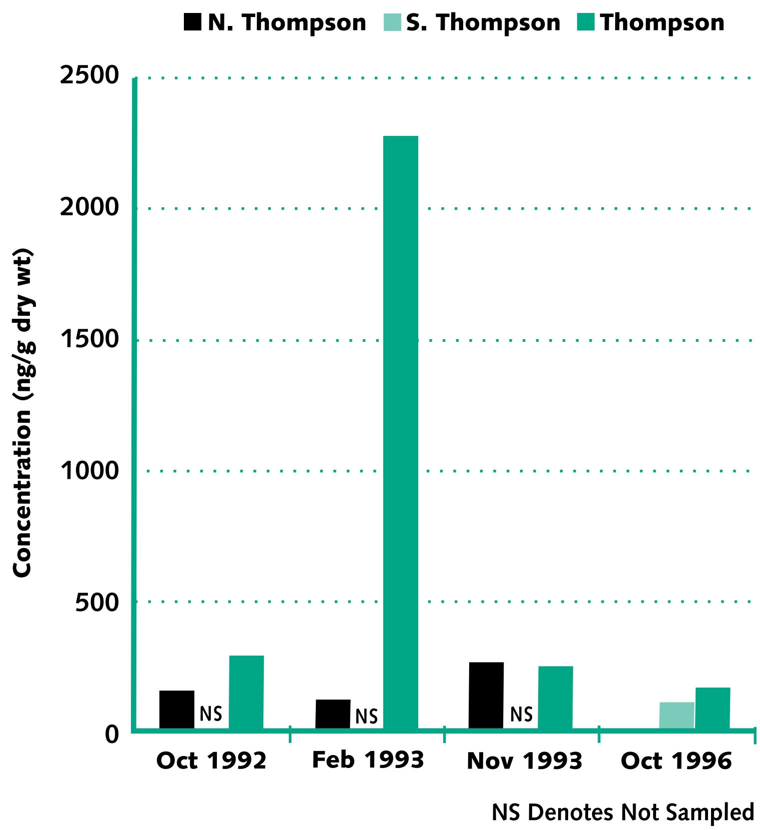


Figure 3. *Total parent PAHs in North Thompson, South Thompson and Thompson River suspended sediment (1992–1996).*

The highest levels of pesticides were generally detected in the lower Fraser River, with the exception of p,p'-DDT, measured in suspended sediments, which was approximately four times higher in concentration at the McBride and the South Thompson River reference sites than in the Fraser River Estuary. The Annacis Island STP did not contribute significantly to organochlorine pesticide concentrations in the Main Arm of the Fraser River. DDD and DDE concentrations measured in bed sediments from the North and Main Arms of the lower Fraser River were approximately an order of magnitude lower than those detected in 1989 by Swain and Walton (1990). A number of pesticides measured in bed sediments exceeded interim federal guidelines and/or provincial criteria for the protection of aquatic life (Table 3).

PCBs

Polychlorinated biphenyls (PCBs) measured in suspended and bed sediments were detected in trace levels in the upper Fraser and Thompson basins. Total PCB congeners in suspended sediments did not exceed 1.0 ng/g and generally did not differ between reference and non-reference sites. The suspended sediment data from the upper Fraser River did not indicate seasonal or temporal variability from 1992 to 1996. Two bed sediment sites had elevated total PCB concentrations exceeding provincial sediment quality criteria for the protection of aquatic life: a rural site on the Fraser River in the Quesnel to Lytton reach (26.4 ng/g) and an urban site on the South Thompson River in Kamloops (1,367.0 ng/g) (Table 3). Aroclors 1254 and 1260 also exceeded severe effects level criteria for the protection of aquatic life at the latter site which is affected by urban runoff.

Total PCBs measured in suspended sediments from the Fraser estuary were over 150 times higher than levels measured at the Agassiz reference site, upstream of the metropolitan Vancouver area, with the highest concentrations measured in the Main Arm of the Fraser River. Total PCB congeners were approximately 10 times higher, and TEQs were over 13 times higher in the Main Arm compared to the North Arm of the Fraser River. The presence of PCBs in sewage treatment plant effluent (Derksen, draft 1997) and elevated concentrations of total PCBs downstream of the Annacis Island STP suggest that the STP may be a source of PCBs to the Main Arm of the Fraser River.

In contrast, bed sediments from the North Arm of the Fraser River had higher PCB concentrations than those of the Main Arm. Sediments from the North Arm of the Fraser River exceeded the provincial sediment quality criterion for the protection of aquatic life for Aroclor 1260 and the regional sediment quality objective for total PCB congeners (Table 3). The elevated levels of PCBs observed in the Fraser River Estuary likely reflect contributions from urban runoff. Peamouth chub liver, collected from the lower Fraser River, similarly had higher total PCB levels relative to the rest of the Fraser Basin (Raymond *et al.* 1999). PCB concentrations in bed sediments were four orders of magnitude lower than those in the Great Lakes Areas of Concern (Bolattino 1993) but generally similar to those measured in the northern rivers of Alberta (Crosley 1996a).

Nonylphenol

Total 4-nonylphenol measured in suspended and bed sediments was elevated at sites in or downstream of major cities in the Fraser River Basin. Total 4-nonylphenol in suspended sediments from Woodpecker and Marguerite was approximately double the concentration measured at the reference reach upstream of Prince George. Similarly, 4-nonylphenol was eight times higher in the Thompson River at Savona (56 ng/g) than in the South Thompson River upstream of Kamloops. However, the highest 4-nonylphenol levels were measured in the Main Arm of the Fraser River (mean=62 ng/g), downstream of the Annacis Island STP, where concentrations were double those measured at a location just upstream of the STP and approximately five times higher than at Agassiz. The detection of relatively high levels of 4-nonylphenol in suspended solids from STP effluent (GVRD, Draft 1996) and the elevated concentrations measured downstream of the plant suggest that the STP may be an important source of this compound to the Main Arm of the Fraser River.

With the exception of a single sample in the Stuart River reach, which had a 4-nonylphenol concentration of 570 ng/g, the highest 4-nonylphenol levels in bed sediments were measured in the North Arm of the Fraser River (7–64 ng/g). Concentrations in the heavily industrialized North Arm reach were three times higher than in the Lytton to Chilliwack reach upstream of metropolitan Vancouver. Concentrations in the North Arm of the Fraser River were approximately double those measured in the Main Arm of the Fraser River. However, levels in the North Arm were from 4–2,000 times lower than those measured in bottom sediments from heavily industrialized sites in the St. Lawrence River in Ontario (Bennie *et al*. 1996). Guidelines or criteria do not presently exist for 4-nonylphenol.

Total 4-nonylphenol partitioned to the suspended solid phase in the range of 0.8–9.3 per cent, indicating that this compound is largely found associated with the water phase. The calculated log K_{oc} was in the range of 4.7–5.7, which was higher than the theoretical log K_{oc} of 4.9 (Karickhoff 1981).

Metals

Elevated levels of copper, manganese and selenium were measured in suspended sediments from Savona relative to the South Thompson, upper Fraser and lower Fraser sites, while the highest levels of lead were measured at the McBride reference site, upstream of Prince George. Whereas natural sources are believed to contribute to the metal concentrations measured at McBride, metal enrichment from Kamloops City may account for the elevated levels measured at Savona. Trace metal concentrations in the lower Fraser River were similar to those found in the upper Fraser River, with the exception of arsenic. Arsenic concentrations were approximately double to those measured in the upper Fraser and Thompson rivers, likely due to contributions from wood preservation facilities in the lower Fraser River using copper arsenate-based wood preservatives.

In addition to arsenic, copper, zinc and lead measured in bed sediments were generally elevated in the Main and North Arms of the lower Fraser River relative to the upper Fraser River. However, concentrations measured in the Chilcotin and Quesnel reference reaches often exceeded those measured in the Fraser estuary.

These relatively high levels in the reference reaches are believed to originate from natural mineral deposits and historical mining activity (K. Andrews 1998, pers. comm.). Elevated levels of some metals measured in the Harrison River reach may be related to atmospheric deposition of trace metals from the greater Vancouver area (W. Belzer 1998, pers. comm.). As with suspended sediments, lead concentrations were highest in the McBride reference reach, indicating that these levels represent natural background concentrations for the basin. Lead concentrations in the lower Fraser River were similar to those measured at the same sites in 1990 and 1992 (Swain and Walton 1991; 1993) suggesting that levels of this metal may be leveling off after an initial sharp decrease associated with the banning of leaded gasoline in 1990.

Chromium, manganese, iron, nickel and copper exceeded provincial criteria (BC MELP 1995) at all sites in the Fraser Basin, indicating that these metals are naturally high in the sediments. Chromium also exceeded the draft sediment quality objective (<26 µg/g) for the Main Arm of the Fraser River (Swain *et al.,* draft 1996). Arsenic exceeded the provincial criterion of 6 µg/g in the Prince George to Quesnel reach, the Nechako, Stuart, Quesnel tributaries and the Fraser estuary. Zinc exceeded the provincial criterion of 120 µg/g in the North Arm of the Fraser River. Previous studies in the Fraser River Estuary have similarly reported metal concentrations exceeding provincial sediment quality criteria for many of the same metals measured in the present study (Swain and Walton 1990; 1991; 1993). In spite of these exceedences, most metals measured in the Fraser Basin were lower in concentration than levels found in the Columbia and Mississippi rivers (Bortleson *et al*. 1994; Garbarino, *et al*. 1995), the Great Lakes Areas of Concern (Bolattino 1993) and many rivers in the USA (Tornes *et al*. 1996; Ott 1997; USGS NAWQA 1997).

CONCLUSIONS

Dioxins, furans, chlorophenolics, resin and fatty acids, PAHs and 4-nonylphenol were measured in higher concentrations downstream of pulp mills and urban centres relative to reference locations throughout the Fraser River Basin. In contrast, organochlorine pesticides and PCBs were elevated relative to reference locations generally only in the urbanized lower Fraser estuary. The North Arm of the Fraser River generally had the highest levels of contaminants measured in bed sediments.

Urban runoff is the likely source of higher chlorinated dioxins, organochlorine pesticides, PCBs and PAHs, whereas pulp mills are the likely source of the lower chlorinated dioxins and furans, chlorophenolics and resin acids to the basin. The Annacis Island STP appears to be an important source of PCBs, PAHs and 4-nonylphenol to the Main Arm of the Fraser River. While metals were found to be naturally high in sediments throughout the Fraser Basin, elevated levels of some metals measured in urban areas may originate from anthropogenic sources. Seasonal fluctuations in contaminant concentrations in suspended sediment were measured for dioxins, furans, chlorophenolics and PAHs. Reductions in concentrations of dioxins, furans, chlorophenolics, pesticides and lead were noted relative to levels measured prior to 1991, due to regulations implemented in the 1980s and early 1990s. Dioxins, furans and HPAHs were generally associated with the suspended sediment phase, while chlorophenolics, LPAHs and 4-nonylphenol were found in higher proportions in the water phase. Exceedences of federal guidelines, provincial criteria and regional objectives for the protection of aquatic life in bed sediments were measured for dioxins and furans, PAHs, PCBs, pesticides and trace metals.

Geographically, urban areas such as the Fraser River Estuary and the Thompson River in and downstream of Kamloops were the most heavily impacted areas in the basin. The upper Fraser Basin was comparatively less impacted, although elevated levels of contaminants associated with pulp mills and municipal STPs were measured downstream of urban centres. On a larger scale, contaminants were generally similar or lower than those measured in other large river systems throughout North America, indicating that the basin is in a relatively good environmental state. Although the environmental quality of sediments from the basin is considered to be generally good, the Thompson River and Fraser estuary should continue to be monitored to ensure that the environmental quality does not further degrade as a result of increasing stress from population growth.

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3.6

BENTHIC INVERTEBRATE COMMUNITY STRUCTURE

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Benthic (bottom-dwelling) macroinvertebrates are the basis for most biomonitoring programs currently in operation worldwide (Rosenberg and Resh 1993), and the reasons underlying the use of these organisms are compelling (Rosenberg and Resh 1996). In the United States, regional programs tend to be local modifications of national programs (*e.g.* Plafkin *et al.* 1989), and national programs in both developed (*e.g.* Australia: Parsons and Norris 1996) and developing countries (*e.g.* India: Sivaramakrishnan *et al.* 1996) may be modifications of ones developed elsewhere.

Fundamental to the scientific method is the use of controls or control conditions against which results obtained under test conditions are compared. In field studies, all variables cannot be controlled and replicates cannot be randomly assigned to treatments, so an attempt is made to choose test and control conditions (often represented by different sites) that are as similar as possible; the variable of interest is then manipulated, and uncontrolled variables are assumed to fluctuate similarly. Traditionally, this problem has been solved in aquatic studies by choosing adjacent sites in streams (*i.e.* upstream and downstream comparisons, Norris *et al.* 1982), by dividing lakes into halves (Schindler 1974), or by using mesocosms (Graney *et al.* 1984). Such approaches have several problems (Cooper and Barmuta 1993); a major issue in streams is confounded designs (Eberhardt 1978), often called “pseudoreplication” (Hurlbert 1984).

A recent development in water quality monitoring has been the attempt to describe **reference conditions** (Reynoldson *et al.* 1997) based on pre-established criteria that exist at a wide range of sites rather than relying on information from one or a few control sites. These reference conditions then serve as the control against which test-site conditions are compared. The notion of reference condition is really one of **best available condition** and it is represented by information from numerous sites.

The concept of a reference condition is a critical element in approaches now being developed for biomonitoring and bioassessment of aquatic resources. For example, the reference condition is central to currently accepted ideas of "biocriteria" being developed by the US Environmental Protection Agency (EPA) (Davis and Simon 1995). The same approach has been used in the United Kingdom for river classification and water quality assessment (Wright 1995). It is currently being used in Canada to develop biological sediment guidelines for the Great Lakes (Reynoldson *et al.* 1995) and is the basis for the National River Health Program in Australia (Parsons and Norris 1996).

The reference-condition approach using benthic invertebrates is well-suited to large-scale biomonitoring programs because sites serve as replicates, and reference sites can be scattered throughout a catchment (Reynoldson *et al.* 1997). Local knowledge, published information or simple reconnaissance trips can be used to identify sites that represent best available condition for use in building the reference-site models. Several approaches to establishing reference-site conditions exist (Figure 1); based on comparisons of these methods (Reynoldson *et al.* 1997) we are currently using the BEAST (BEnthic Assessment of SedimenT), which was developed in Canada (Reynoldson *et al*. 1995). However, the efficacy of the other approaches will be assessed further.

We used the reference-condition approach to develop a permanent database for biomonitoring water quality in the Fraser River catchment. In this report, we discuss key elements of a regional benthic monitoring programme developed for the entire catchment (234,000 km^2) of the Fraser River.

METHODS

Site selection

Our intent was to sample approximately 250 reference sites over three field seasons (1994–1996) to characterize variability and to build the appropriate predictive models. In addition, we included a smaller number of test sites (*i.e.*

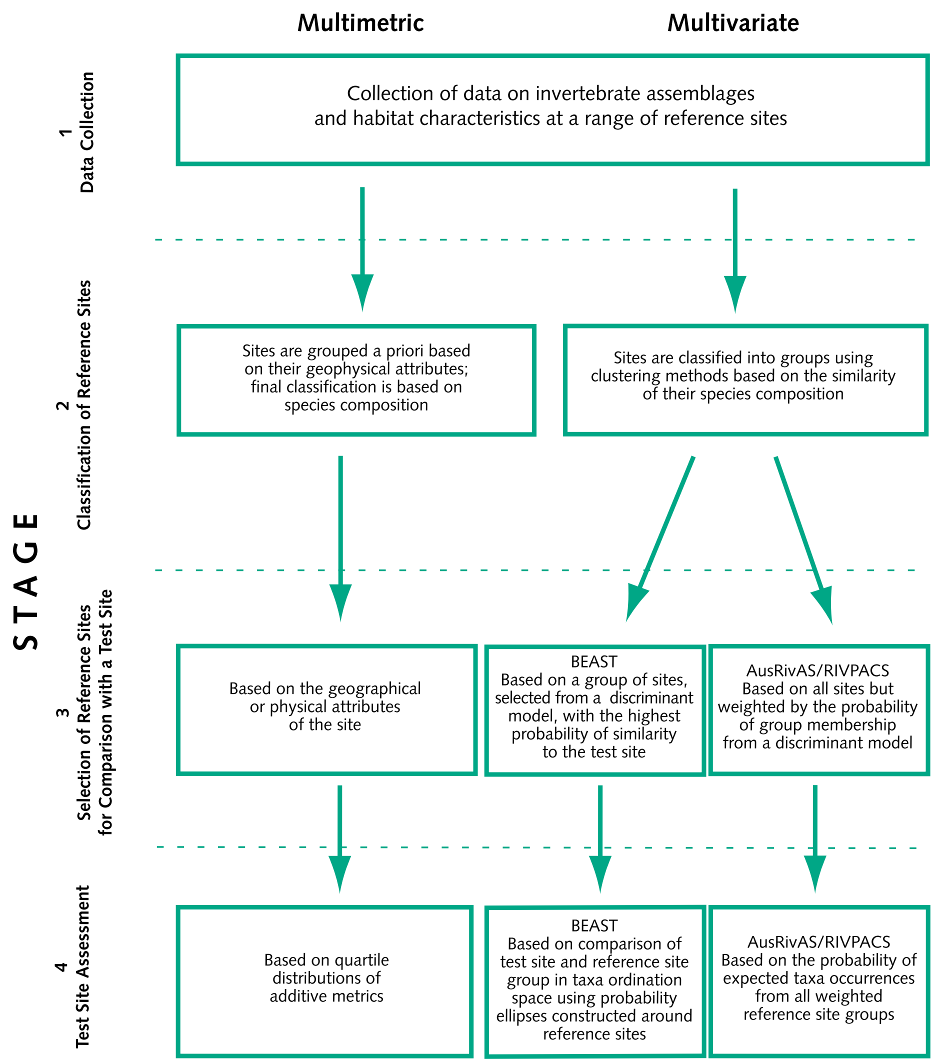

Figure 1. *Flowchart of assessment methods using multimetric and multivariate approaches.*

AUSRIVAS = AUStralian RIVer Assessment Scheme; BEAST = BEnthic Assessment of SedimenT; RIVPACS = River InVertebrate Prediction And Classification System (from Reynoldson et al. *1997).*

those potentially impacted by human activity) to verify performance of the reference-site model during its development.

Site selection was based on two spatial scales (ecoregion and stream order) and two temporal scales (annual and seasonal).

Spatial scales

It is important to capture the range of conditions within the study boundaries. The diversity of sites was maximized by stratifying the basin in terms of both ecoregions and stream orders. The ecoregion scale ensured the inclusion of different climatic and landscape conditions, whereas the stream-order scale ensured the inclusion of a range of hydraulic conditions. Sites were located on up to seven stream orders (at a map scale of 1:250,000) in 23 subcatchments representing 11 ecoregions. In the selected subcatchments approximately equal numbers of sites were sampled for each stream order.

A series of workshops with local experts identified impacted and unimpacted subcatchments. Unimpacted catchments were those judged to have no or minimal human activity, such as logging, mining or agriculture. Reference and test sites were randomly selected from each sampled subcatchment.

Temporal scales

We examined how annual and seasonal variability affected the accuracy of the predictive models. We sampled nine sites in all three years of the field research to measure annual variability; the results are unavailable at this time. Seasonal variability was investigated by P. Dymond (Dymond *et al.* 1997). Analysis of data from eight sites sampled seasonally showed that there was considerable variation of the communities at the species level from the reference condition based on fall sampling. Using models based on family level taxonomic information it was observed that the seasonal samples frequently fell within the range of variation described by sites sampled only in the fall. However, at this time we would recommend that, where possible, site assessments be conducted based on fall sampling.

Environmental variables

Environmental variables were used in two ways: (1) to relate habitat conditions to subsets of reference sites grouped by similarities in invertebrate communities and (2) to build the predictive models for matching new sites to the appropriate subset of reference sites.

An optimum set of predictor variables cannot be determined *a priori*, so a maximum number of likely variables was chosen before sampling began. We compiled these variables from published information and reviewed and amended them at an initial workshop. Table 1 shows the final list for which data were collected.

Sites sampled

Figure 2 shows the distribution of reference and test sites sampled over the three years of the study. We sampled 37 reference sites and nine test sites in 1994, 90 reference and 11 test sites in 1995, and 97 reference and 26 test sites in 1996 for a total of 224 reference sites and 46 test sites. In addition, 21 sites were used for quality assurance at which additional samples were taken. At reference sites where additional quality assurance samples were collected, the first sample was used in the reference database, whereas the additional samples were used as test samples.

Benthic macroinvertebrate collection

We collected benthic invertebrates at erosional habitats (*i.e.* riffles) using a 38 x 38 x 38 cm kick net (400-mm mesh). We took one sample of three minutes duration per site. Two-hundred-organism subsamples were removed using a Marchant box (Marchant 1989). Details of the calibration study and procedures are given in Rosenberg *et al*. (in preparation). The leaf-pack assemblage was also sampled concurrently to

Table 1. Environmental variables measured.

MAP	SITE	CHANNEL	WATER
Latitude	Date	Wetted width	pH
Longitude	Velocity	Mean depth	Dissolved oxygen
Altitude	Discharge	Max. depth	Conductivity
Ecoregion	Flow	Bankfull width	Temperature
Stream order	Macrophyte cover	Slope	Total phosphorus
	Riparian vegetation	Substrate framework[1]	Nitrate
		Substrate matrix[2]	Alkalinity
		Substrate embeddedness[3]	Total suspended solids
		Mean particle size	
		Max. particle size	
		Periphyton biomass	
		Periphyton chlorophyll *a*	

[1] *Estimated size of the dominant particle*
[2] *Estimated size of the surrounding material*
[3] *The degree to which the framework material is buried within the matrix*

determine if the benthic invertebrate assemblage from this habitat provided additional information on the biological condition of a site. This study was the subject of a M.Sc. thesis by S. Sylvestre at the University of Western Ontario (Sylvestre 1997).

We were concerned about the need for non-specialists to identify macroinvertebrates, so we compared the efficacy of family-level versus lower-taxon (mostly genus) level identifications in the determination of site groups and in model development (see below).

Reference condition statistics

We used the four-stage procedure (Fig. 1) described in Reynoldson *et al*. (1997) for the Fraser River catchment data. The stages are as follows: (1) assemble data from multiple reference sites; (2) use cluster analysis (unweighted pair mean group average) to develop reference-site groups based on the invertebrate assemblage data; (3) use discriminant function analysis to relate the structure observed in invertebrate assemblages to environmental variables, and derive an

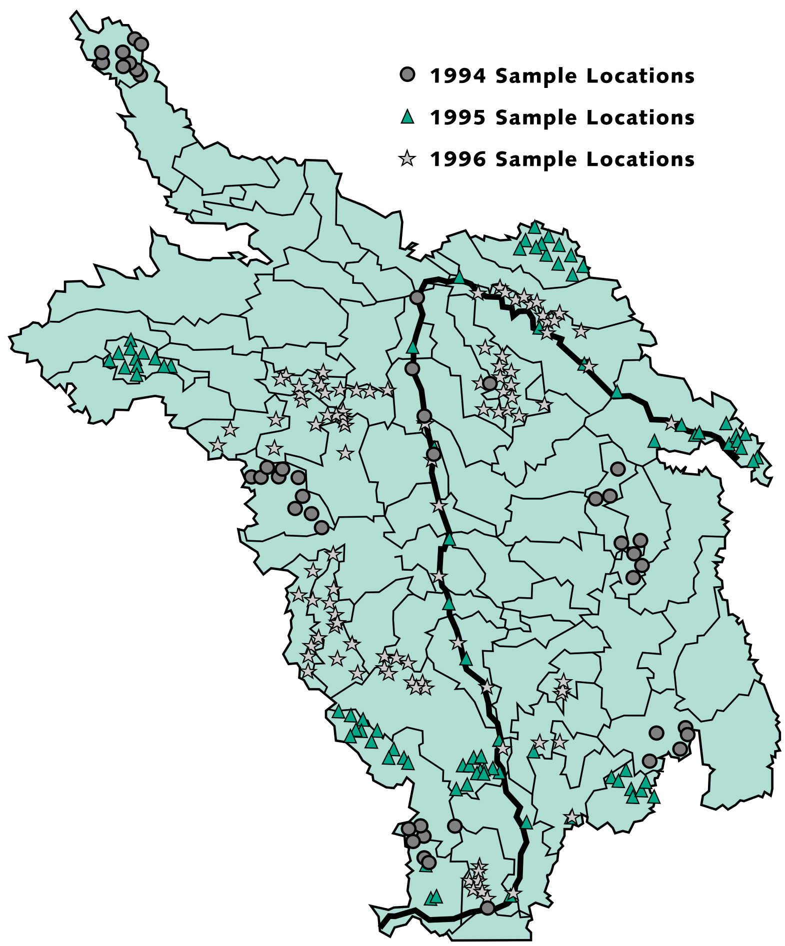

Figure 2. Fraser River Basin sampling locations, 1994 (circles), 1995 (triangles) and 1996 (stars), with the Fraser River and sub-basin boundaries indicated.

optimal set of variables to predict group membership; and (4) use hybrid multi-dimensional scaling ordination to assess the biological condition of new (test) sites in comparison to the reference group or groups that has/have similar values for predictor variables.

The following results describe models developed from a preliminary analysis of the data. Slight modifications in model performance may result from further analysis once the taxonomic data have been verified and final data analysis is complete. Furthermore, as new reference sites are added, the models will be re-calibrated on a regular basis.

RESULTS AND DISCUSSION

Site selection

Stage 1

The database currently consists of 322 separate records, each representing a sample taken from a discrete physical location and having complementary data on the benthic fauna and habitat descriptors. Each site has been categorized as either reference or test (Table 2).

Table 2. *Reference and test sites included in the Fraser River Basin database, 1994–1996.*

YEAR	REFERENCE	QUALITY ASSURANCE	TEST	
			POSSIBLE IMPACT	MAIN STEM
1994	37	[a]5 (20 samples)	5	4
1995	90	[b]7 (14 samples)	0	11
1996	97	[b]9 (18 samples)	15	11
Total	224	21 (52 samples)	20	26

[a] 5 replicated samples taken per site; 1 used as a reference site, 4 used as test sites.
[b] 3 replicated samples taken per site; 1 used as a reference site, 2 used as test sites.

Model building at the family level

Stage 2

Of 224 potential reference sites, 220 were included in the initial model building. Four sites were excluded because they were apparent outliers; they will be re-examined for later inclusion in the database. The cluster analysis and ordination diagrams show six macroinvertebrate groups (Fig. 3), formed from 74 families. Group determination is subjective and based upon a minimum group size of five sites. The six groups showed little geographic pattern (*i.e.* they were scattered among the subcatchments sampled) except for Group 6 sites, which were largely located on the western side of the basin.

Stage 3

Principal axis correlation, a multiple regression method, was used to relate the habitat descriptor matrix to the taxa matrix. Twenty habitat variables were identified to be significantly related ($P<0.01$) to the ordination structure of the benthic fauna at the family level: ecoregion, altitude, latitude and longitude (map variables); water velocity (maximum and average), per cent grasses, and macrophytes (site variables); bankfull width, channel width, channel depth (average and maximum), and three substrate measures (channel variables); and Kjeldahl nitrogen, nitrate-nitrite, total phosphorus, pH, and alkalinity (water variables).

Stepwise discriminant analysis selected nine variables to describe the six groups identified by cluster analyses (Fig. 3) of the family-level invertebrate data. The relationship of these nine variables in ordination space

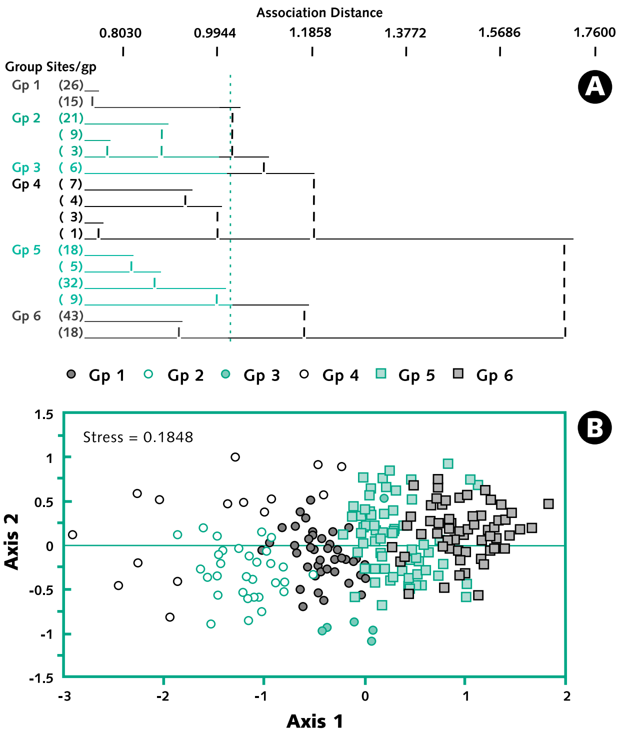

Figure 3. Family-level classification of 220 reference sites in the Fraser River Basin, 1994–1996.

(A) Cluster diagram. Vertical dotted line indicates cut-off point for group formation. (B) Ordination of six groups.

to the centroids of each of the site groups is shown in Figure 4, and the mean values of these environmental variables for each group are shown in Table 3.

Using discriminant analysis (Table 4), a number of predictive models were tested. The performance of different models was tested by their ability to classify sites, using a set of habitat variables, to the group they had been originally assigned to by cluster analysis of the invertebrate assemblage. The error rate for each model was calculated as the average of the percentage of sites per group incorrectly assigned. Three models appeared to perform well. The optimal model used 27 variables (error rate = 34.5%), the stepwise model using nine variables had an error rate of 42.9 per cent, and a model based on stepwise discriminant analysis using 19 variables performed similarly to the optimal model (error rate = 36.4%).

The same analysis is being done at the lower-taxon level. At the species/genus level preliminary analysis has produced nine groups with an error rate of 32.7 per cent using 27 environmental variables. Details of the lower-taxon level analysis for the complete

Table 3. Mean values of predictor variables in six reference groups.

PREDICTOR VARIABLES	GROUP 1	2	3	4	5	6
Grasses present (% sites in group)	22	6	17	7	48	69
Alkalinity (mg/L)	32.3	26.0	23.9	45.6	44.0	62.3
Channel width (m)	23.8	23.9	15.3	77.4	12.3	15.0
Bankfull width (m)	57.1	59.8	95.8	126.3	26.6	30.2
Framework (cm)[1]	11.3	22.1	24.3	4.8	7.7	3.6
Average velocity (cm/s)	0.50	0.59	0.43	0.36	0.41	0.37
Maximum channel depth (cm)	45.9	49.7	33.0	37.3	34.7	31.8
pH	7.6	7.3	7.3	7.7	7.7	7.7
Embeddedness (%)[1]	30.0	33.2	29.2	46.8	28.0	26.8

[1] *Defined in Table 1*

Table 4. Prediction of family groups from three discriminant models.

	TO GP 1	TO GP 2	TO GP 3	TO GP 4	TO GP 5	TO GP 6	AVERAGE ERROR RATE
Reference sites per group	41	33	6	15	64	61	
Sites correctly predicted by:							
Stepwise model (9 variables)	16	20	5	8	28	38	42.9%
Enhanced stepwise model (19 variables)	15	21	6	11	26	41	36.4%
Optimal model (27 variables)	15	18	6	11	35	45	34.5%

data set will be presented in Reynoldson *et al.* (in prep.).

These models were developed with the taxonomic database before final revision and with species level data for 1994 and 1995 only. Finalised models for three taxonomic levels (species, genus and family) are presented in the final technical report (Rosenberg *et al.* 1999).

Stage 4

Next, we assessed test sites against the reference-site groups. We used a multivariate method that incorporates the entire invertebrate assemblage. In this method, the reference and matching test sites are ordinated and plotted in ordination space using hybrid multi-dimensional scaling. The distribution of the reference sites provides the normal range of variation in unimpaired communities. The expected assemblage at a test site is determined by predicting the site to one of the reference-site groups using the predictive models constructed in stage 3. The actual assemblage at a test site can then be compared to this normal variability with a given probability of belonging to the reference group using probability ellipses built around the reference sites. The greater the departure from reference state, as measured in ordination space, then the greater the impact. However, determining actual degree of impact and what departure from the reference state defines an unacceptable impact is ultimately a subjective decision.

An extensive river water quality survey was conducted in the United Kingdom in 1990. The survey provided the impetus for the development of methods to circumscribe the continuum of responses into a series of bands representing grades of biological quality (Wright *et al.* 1991). The method is a simplification of what is a description of the continuum of site responses ranging from good to poor biological quality. The method is appropriate for obtaining a simple statement of biological quality allowing broad comparisons in either space or time that would be useful for management purposes.

We used a similar approach based on probability ellipses (Figure 5) constructed around a ref-

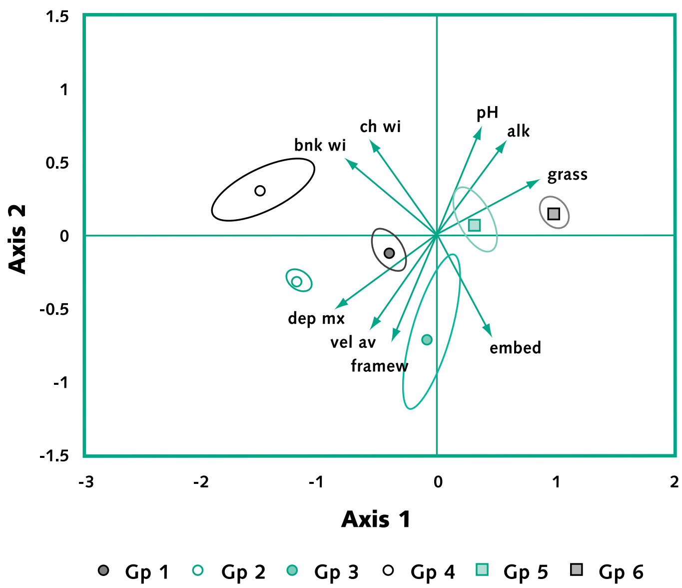

Figure 4. Principal axis correlation vectors for predictor variables and their relationship to six family-level groups for Fraser River Basin reference-site data, 1994 to 1996.

Ninety per cent confidence ellipses represent the ordination space occupied by the centroid of each group. (alk. = alkalinity, bnk.wi. = bankfull width, ch.wi. = channel width, embed. = embeddedness, dep. mx. = maximum channel depth, framew. = framework, grass = presence of grasses in riparian zone, vel. av. = average velocity).

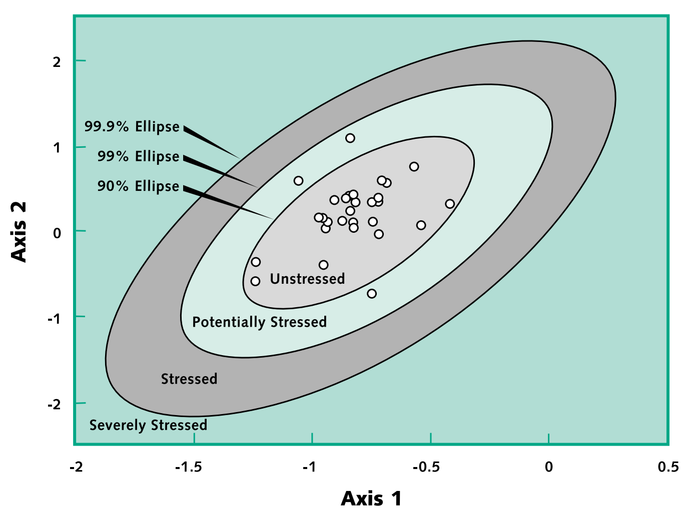

Figure 5. Example of probability ellipses calculated for reference sites to indicate potentially stressed (90%), stressed (99%) and severely stressed (99.9%) invertebrate assemblages at test sites.

Note that some reference sites fall naturally outside the ellipses.

erence-site group. Test sites within the smallest ellipse (90% probability) would be considered **unstressed**; sites between the smallest and next ellipse (99% probability) would be considered **potentially stressed**; sites between the 99 per cent probability and the largest ellipse (99.9% probability) would be considered **stressed**; and sites located outside the 99.9 per cent ellipse would be designated as **severely stressed**.

A test site is assigned to a reference group using a discriminant model, and the assemblage at the test site is compared to the assemblage of the matching reference sites. A data file is constructed that includes the taxa counts for the appropriate reference sites and the test site. The data are ordinated so that a matrix is calculated for both reference and test sites. The sites can then be plotted in ordination space showing the ordination dimensions that synthesize the biological attributes of the sites (Fig. 5). Probability ellipses are calculated for the reference sites only. The location of the test sites relative to the reference sites can then be determined. The assessment of stress is based on site locations on the ordination axes, and the overall assessment is based on the most stressed band to which a site belongs. In using such a probability based method there is a likelihood that some reference sites will naturally fall outside the ellipses (*e.g.* 10% of the reference sites are likely to be outside the 90% ellipse). In Figure 5 this can be seen for three of the 29 reference sites.

To demonstrate this approach, a selection of sites representing potential impacts from agriculture, logging and mining were sampled (Fig. 6). The observed responses are summarized below. Water quality in the Fraser River main stem was also assessed to determine if there were large spatial responses to urban activities and cumulative impacts on the river from basin activities.

Detecting impacts from agriculture, logging and mining

The methods described above were used to assess the ability of the reference-condition approach to detect and differentiate between different types of impact. To assign test sites to a reference-site group we tested all three models described above (Table 4) and found high concordance (87.2%) among the groups to which a test site was predicted. To illustrate the method for assessing individual test sites we have presented detailed results from those assessed sites predicted to belong to one reference group (Group 6, Fig. 7). A complete summary for all test sites is shown in Table 5.

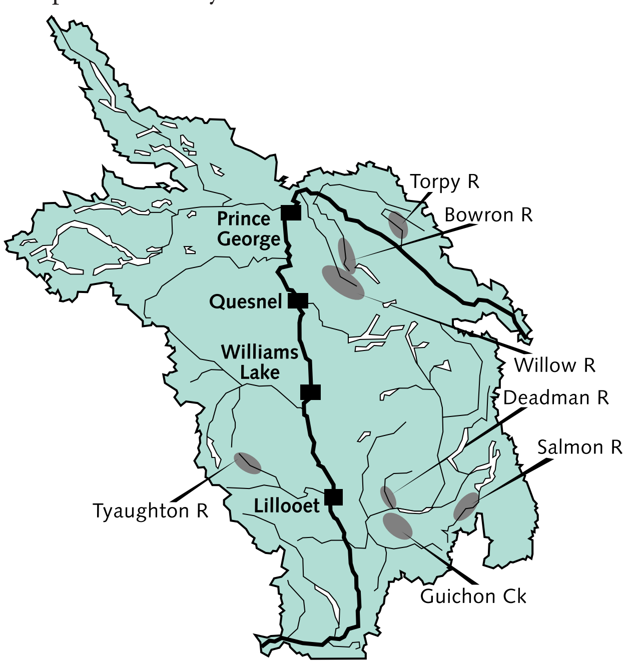

Figure 6. *General location of sites in the Fraser River catchment used to test for stress caused by human activity.*

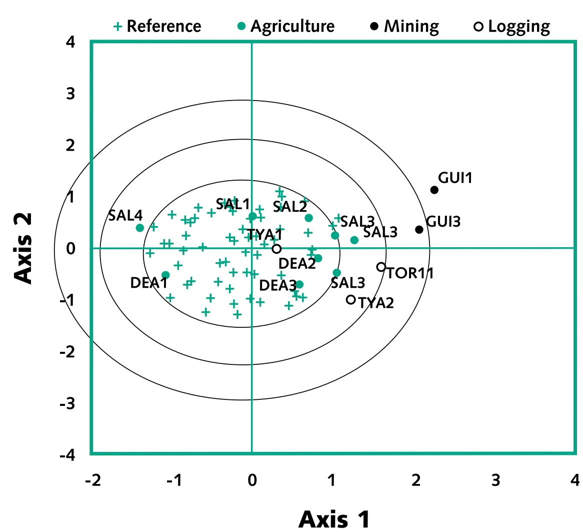

Figure 7. *Assessment of macroinvertebrate assemblage structure at test sites predicted to Group 6 using the BEAST model from the Fraser River Basin.*

Sites were located in the Deadman (DEA), Salmon (SAL), Torpy (TOR), Tyaughton (TYA) and Guichon (GUI) catchments.

Table 5. Assessment of test sites exposed to either agricultural (A), mining (M) or logging (L) activities. Sites were located in the Deadman (DEA), Salmon (SAL), Bowron (BOW), Torpy (TOR), Tyaughton (TYA), Willow (WIL) and Guichon (GUI) catchments.

SITE	POTENTIAL STRESS	PREDICTED TO GROUP	PROBABILITY OF BELONGING TO PREDICTED GROUP	ASSESSMENT
DEA1	A	6	0.90	Unstressed
DEA2	A	6	0.89	Unstressed
DEA3	A	6	0.92	Unstressed
SAL1	A	6	0.98	Unstressed
SAL2	A	6	0.75	Unstressed
SAL3	A	6	0.93	Unstressed
SAL3	A	6	0.93	Potentially stressed
SAL3	A	6	0.93	Potentially stressed
SAL4	A	6	1.00	Potentially stressed
BOW13	L	5	0.74	Unstressed
BOW14	L	1	0.43	Potentially stressed
TOR10	L	1	0.51	Unstressed
TOR11	L	6	0.74	Potentially stressed
TYA1	L	6	0.79	Unstressed
TYA2	L	6	0.92	Potentially stressed
TYA5	L	5	0.55	Unstressed
WIL1	L	5	0.62	Unstressed
WIL4	L	5	0.64	Unstressed
GUI1	M	2/6	0.68/0.090	Severely stressed/severely stressed
GUI3	M	2/6	0.81/0.94	Stressed/stressed
WIL2	M	5	0.46	Unstressed
WIL3	M	5	0.52	Unstressed

Sites potentially impacted by agriculture were sampled on the Deadman (DEA1–3) and Salmon (SAL1–4) rivers (Fig. 6). At one of these sites (SAL3), three replicates were taken for quality assurance purposes and were assessed individually. The sites on the Deadman River showed no effects of agricultural activity. They were within the 90 per cent probability ellipse for the reference sites (Figure 7), and are therefore assessed as being unstressed (equivalent to reference). Field notes and photographs taken from these three sites at the time of sampling showed little evidence of physical disturbance, grazing or other agricultural activity. On the Salmon River, only the two downstream sites showed indications of potential stress (SAL3 and SAL4 in Fig. 7). The effects on the invertebrate assemblage are different at SAL3 and SAL4 based on their location in ordination space (Fig. 7). All three samples from SAL3 were located to the right of the reference group. This site showed evidence of physical disturbance and bank instability, and the response in the assemblage was one of reduced abundance of organisms. Site SAL4 moved in the opposite direction in ordination space because of increases in numbers of organisms. This site was located adjacent to a small paddock with grazing stock and demonstrates the effects of nutrient enrichment.

Four streams were sampled that had, based on visual inspection, varying degrees of logging activity. Two sites were sampled on the Bowron River (BOW13, 14), two sites on the Torpy River (TOR10, 11), three sites on the Tyaughton River (TYA1, 2, 5), and two sites on the Willow River (WIL1, 4). Although at each of these sites logging had occurred over the past few years, in most cases there was a well-developed riparian buffer zone. Only three sites indicated potential stress (Table 5, Figure 7): BOW14, TOR11, and TYA2 (BOW14 is not shown in Figure 7 because it was a Group 1 site). All three sites had generally lower numbers and/or taxa than the reference sites, which indicates physical disturbance. Site TOR11 had been channelized and TYA2 was just upstream of a large culvert.

Guichon Creek and Willow River were sampled for potential mining impacts. One Guichon site (GUI1) was a drainage ditch from a tailings pond, and the creek channel had no flow. The second site (GUI3) was a residual channel in the original water course; however, it was fed by a small pipe and the substrate had extensive mineral encrustation. Both sites were identified as impaired: GUI1 was severely stressed and GUI3 was stressed. The two sites on the Willow (WIL2 and WIL3) were located downstream of Jack of Clubs Lake, which has received extensive tailings from historic mining activities in the Barkerville area (Mudroch *et al.* 1993). Site WIL3 was approximately two km from the lake outlet and WIL2 was a further 12 km downstream. Neither site showed any visual evidence of impairment and both had well-developed riparian zones. Both sites were assessed as unstressed.

From these results, it is evident that the reference-condition approach can detect invertebrate assemblage responses to all three stress categories. It is also noteworthy that the type of stress can be inferred by the trajectory of the site away from the reference-site swarm (Fig. 7). Responses to physical disturbance and enrichment resulted in sites moving in different directions away from the reference swarm.

Changes along the main stem of the Fraser River

A subsidiary objective of the study was to examine changes in invertebrate communities along the length of the Fraser River from the headwaters to the lower boundary of the study area at Agassiz. A total of 28 sites were located approximately every 50 km along the river. Four sites were sampled in 1994, and alternate sites were sampled in 1995 (even-numbered) and 1996 (odd-numbered). In addition, three sites were sampled in each of the study years (14, 16 and 28) and two sites (6 and 20) were replicated in 1995, to examine annual and site-scale variability.

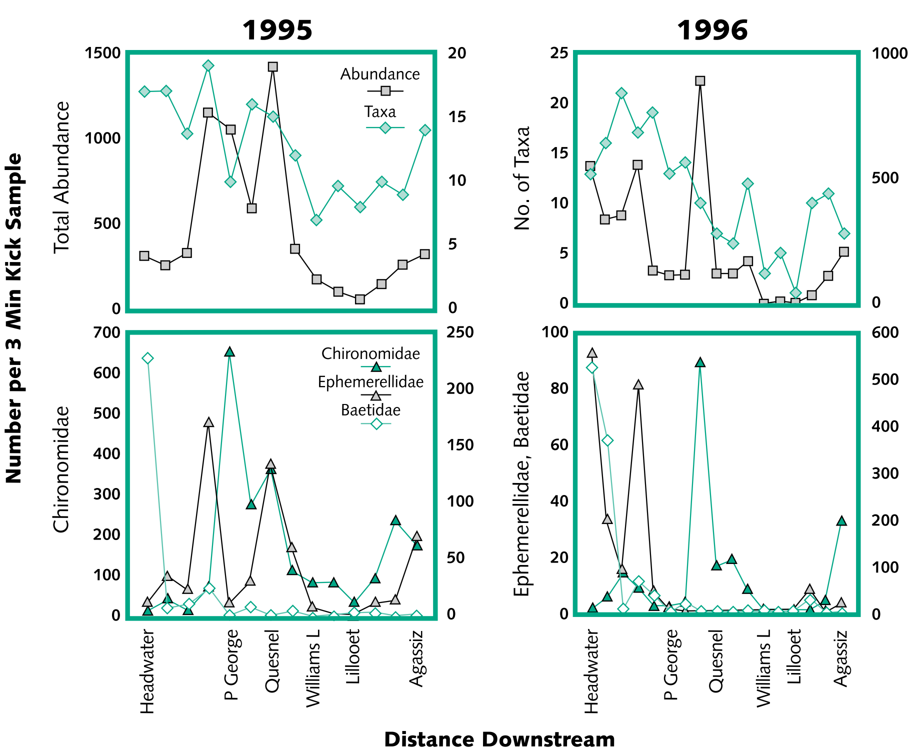

Figure 8. Downstream changes in abundance of invertebrates in the Fraser River main stem, 1995 and 1996.

There are similar trends in 1995 and 1996 in overall abundance and in the major families (Fig. 8). There is a general trend of increasing abundance from the headwaters to Quesnel (Site 14), followed by a region of low abundance to Lillooet (Site 23) and then a gradual increase in abundance to the last site (28). Diversity (number of taxa) decreases from the headwaters to the lower reaches. The changes in abundance and diversity are partially reflected by the major families. The upstream sites, particularly in the headwaters, are dominated by mayfly families.

The baetid mayflies occur primarily in the headwaters and the ephemerellid mayflies occur in the headwater and upstream sections (Figure 8). Increased total abundance between Prince George and Quesnel is primarily the result of increased abundance of chironomid midges (Fig. 8).

We compared the main stem invertebrate assemblages to the reference-site database to determine if the above pattern of change in invertebrates indicates changes other than those associated with natural variation. The main stem sites were separated into those included as part of the reference data set used for model building and those tested to determine if there were any impacts occurring in the river from point or non-point sources (Fig. 9).

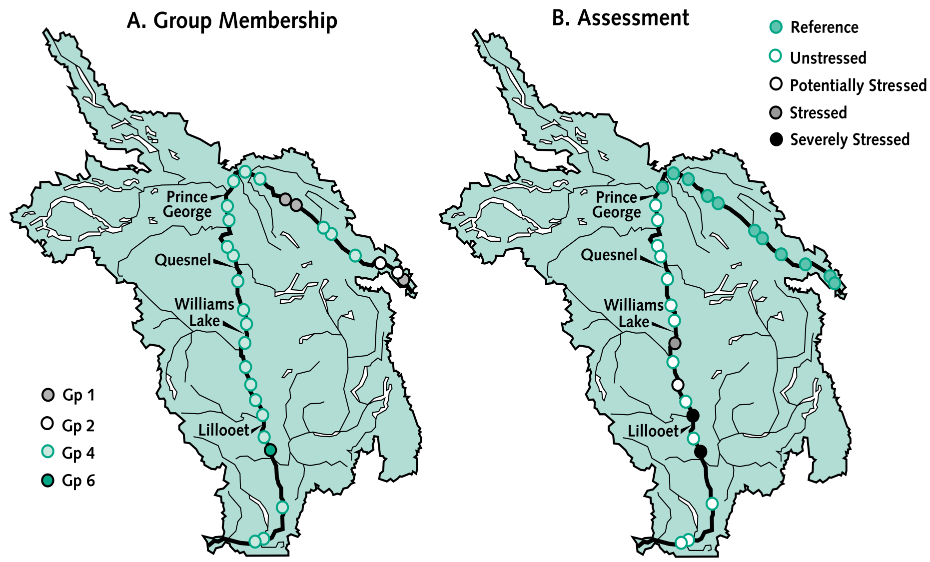

Figure 9. *Location of 28 sites on the main channel of the Fraser River.*
(A) Group membership of reference and test sites. (B) Assessment of test sites.

Sites upstream of Prince George (FRA1–11) were considered to be part of the reference data set because of insignificant anthropogenic activity in the area. Sites downstream of Prince George (FRA12–28) were tested to determine whether or not they could be considered as equivalent to reference. The results are summarized in Table 6.

Most sites on the main stem are either members of Group 4 (Fig. 9A) or are predicted to be members of Group 4. Of the sites downstream of Prince George, most were assessed as unstressed (Fig. 9B). In 1996, four sites from downstream of Williams Lake to immediately downstream of Lillooet diverged from the reference state. There are no significant industrial discharges to the river in this area, and in both study years total abundance and number of taxa were low in this river reach (Fig. 8). There are three possible explanations for the assessment of these sites as being stressed in 1996. First, there is some stressor other than point source discharges to which the assemblages are responding. Second, the low numbers observed in 1996 are at the extreme of the normal range and thus, by chance, are outside the probability ellipse. Third, the reference-site group is an inappropriate match for these test sites. We suspect that the third explanation is the most probable one, because the four sites in question are in a canyon with quite different characteristics compared to other main stem sites. Furthermore, reference Group 4 is small (15 sites) and includes mainly main stem sites upstream of Prince George. The applicability of the reference-site database to this reach of the river will be investigated further.

The assessment shows no response in the invertebrate assemblages in the vicinity of the pulp and paper discharges at Prince George and Quesnel. The stimulation of invertebrate growth as a result of exposure to pulp mill effluent from Prince George, as observed under experimental conditions (Culp and Lowell 1999), is not sufficient to distinguish these sites from the normal variation found in the ambient river environment. The effect may be too small to be detected by our reference-condition approach but it should be noted that the closest sites were 17 km downstream of the discharges.

Table 6. Assessment of Fraser River main stem sites, 1994–1996.

SITE	PROBABILITY OF BELONGING TO PREDICTED GROUP	ASSESSMENT
FRA6-1995	1.00	Unstressed
FRA6-1995	1.00	Unstressed
FRA12-1995	1.00	Unstressed
FRA13-1994	1.00	Unstressed
FRA13-1996	1.00	Unstressed
FRA14-1994	1.00	Unstressed
FRA14-1995	1.00	Unstressed
FRA14-1996	1.00	Unstressed
FRA15-1996	0.95	Unstressed
FRA16-1994	1.00	Unstressed
FRA16-1995	1.00	Unstressed
FRA16-1996	1.00	Unstressed
FRA17-1996	0.99	Unstressed
FRA18-1995	1.00	Unstressed
FRA19-1996	1.00	Stressed
FRA20-1995	0.84	Unstressed
FRA20-1995	0.98	Unstressed
FRA20-1995	1.00	Unstressed
FRA21-1996	0.99	Potential stress
FRA22-1995	1.00	Unstressed
FRA23-1996	0.81	Severe stress
FRA24-1995	1.00	Unstressed
FRA25-1996	0.90	Severe stress
FRA26-1995	1.00	Unstressed
FRA27-1996	0.87	Unstressed
FRA28-1994	0.96	Unstressed
FRA28-1995	1.00	Unstressed
FRA28-1996	0.99	Unstressed

All sites were predicted to Group 4, except FRA25-1996, which was predicted to Group 6.

Replicated samples were taken for quality assurance at selected locations (FRA6, 20). These samples were assessed separately and showed consistent prediction and assessment. Four sites that were sampled for two or three years (FRA13, 14, 16, 28) showed no annual variation in either the predicted assemblage or the assessment, suggesting the database and predictive models are resilient to short-term annual variability.

CONCLUSIONS

Macroinvertebrate assemblages, and habitat and water chemistry variables were measured using 224 reference samples and 98 test, repeat or quality assurance samples from three field seasons. The sites sampled encompass 23 subcatchments, including the Fraser main stem. This database represents the most comprehensive sampling of benthic macroinvertebrate assemblages for the Fraser River catchment.

We have developed family-level predictive models that use six reference groups of benthic invertebrates and up to 27 predictor variables. Our error rates are 34.5 per cent to 42.9 per cent, for predicting reference sites to a group. For a preliminary lower-taxon predictive model, based on 1994–1996 data and using nine groups and 27 predictor variables, the error rate was 32.7 per cent. The most significant predictors of benthic macroinvertebrate assemblages at the family level are simple site descriptors such as presence of grass in the riparian zone, channel and bankfull width, channel depth, substrate measurements such as the size of the largest particles (framework) and the degree of embeddedness, and water chemistry descriptors such as alkalinity and pH. The importance of these variables suggests that at the family level large-scale processes are structuring benthic communities.

We have completed assessments on 46 test sites, which indicate that effects from logging, mining and agricultural activities can be evaluated. The analytical technique, using multivariate assessment methods, allows a judgment on the likely cause for the impairment (*e.g.* physical disturbance versus enrichment).

The reference database, models, and software (being developed) form the basis of an effective tool for assessing impairment of *in-situ* benthic invertebrate assemblages occurring in the Fraser River catchment. The tool can be used as is, or it can be added to over time, thus enhancing its application to the basin. We would encourage future studies to adopt the reference-condition approach and the protocols developed during this study to allow for further development.

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3.7

FISH COMMUNITY STRUCTURE AND INDICATOR SPECIES

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This chapter summarizes the characteristics and distribution of fish assemblages within the Fraser Basin. The detailed information has been assembled in a database which will provide a baseline for the assessment of future changes in both the distribution of individual species and the characteristics of fish assemblages in different portions of the basin. The chapter includes an in-depth assessment of the life history, habitat use and seasonal movements of mountain whitefish (*Prosopium williamsoni*) in the Prince George region of the Fraser Basin. This information is then examined to assess the suitability of the species as an indicator species for contaminant tracking in this reach of the Fraser and, to some degree, in other reaches of the Fraser and its major tributaries (*e.g.* the Thompson River).

FISH DISTRIBUTION AND COMMUNITY STRUCTURE

The core of this project is a database consisting of information on the distribution of fishes throughout the entire Fraser Basin (*i.e.* from the estuary to Yellowhead Lake) (McPhail *et al*. 1998). The bulk of the information in the database comes from the fish collection in the Department of Zoology at the University of British Columbia, but also includes information from the Royal British Columbia Museum; BC Ministry of Environment, Lands and Parks; Department of Fisheries and Oceans; and grey literature (unpublished theses, consultants' reports and government files). Although the database does not include the detailed physical and geomorphological data available in larger government databases, it is designed to interact with them through Geographic Information System programs like ArcView.

A unique property of this database is that it includes distributional information on all 59 species (native and introduced) that occur within the Fraser Basin above the estuary. In contrast, the larger government fisheries databases (both provincial and federal) rarely contain data on species other than salmonids and, when they do contain information on other species, the identifications are unreliable. Thus, the main strength of this database is the breadth and reliability of its coverage of species distributions.

Analysis of the distribution of the 40 native freshwater species in the drainage system and the eight marine species that regularly enter the river (Table 1; see the end of the chapter) suggests three natural species assemblages. An estuarine assemblage (Table 2; see the end of the chapter) occupies the estuary (out to Sturgeon Bank) and the river to the upper limits of tidal influence at Mission. An example of the distribution of a typical estuarine species (starry flounder, *Platichthys stellatus*) is given in Figure 1. A lower river assemblage occupies the river from Mission upstream to the Fraser Canyon (Table 3; see the end of the chapter). This assemblage is characterized by eight native species not found upstream of this reach in the Fraser system. An example of the distribution of a typical lower river species (threespine stickleback, *Gasterosteus aculeatus*) is presented in Figure 2. A distinctive upper river assemblage occupies the river upstream of the Fraser Canyon (Table 4; see the end of the chapter). This assemblage is characterized by six species and one subspecies that are not found below the canyon. Two other species (lake trout [*Salvelinus namaycush*] and lake whitefish [*Coregonus clupeaformis*]) originally absent below the canyon have been transplanted from the upper basin to the lakes in the lower basin. An example distribution of a typical upper river species (lake chub, *Couesius plumbeus*) is also presented in Figure 2.

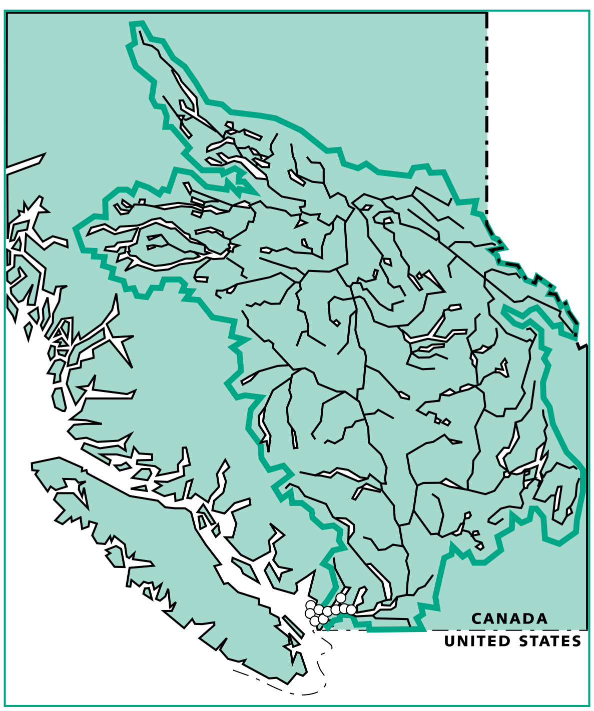

***Figure 1.** The Fraser Basin distribution of the starry flounder,* Platichthys stellatus, *a typical member of the Fraser estuarine assemblage.*

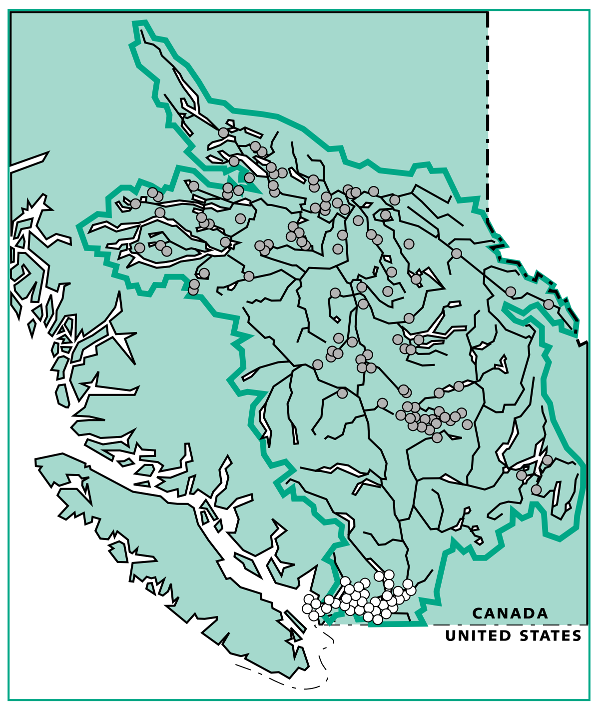

***Figure 2.** The Fraser Basin distributions of the threespine stickleback,* Gasterosteus aculeatus, *and the lake chub,* Couesius plumbeus, *typical members, respectively, of the lower and upper Fraser assemblages.*

The upper river assemblage is further divisible into a species assemblage that occupies the river from the Fraser Canyon upstream to about the confluence of the Bowron and Fraser rivers, and an assemblage that occupies the river above this junction. This uppermost assemblage is characterized by the loss of species—eight species common in the Prince George area (*e.g.* the Nechako River drainage) drop out somewhere between Prince George and the confluence of the Fraser and Bowron rivers (Table 5). The distribution of the white sucker (*Catostomus commersoni*) is typical of species that drop out of the fauna upstream of Prince George (Figure 3). Above this junction, the main stem fish assemblage is dominated by species adapted to life in swift glacial rivers. Interestingly, similar reductions in diversity occur in the upper parts of most Fraser River tributaries, often involving the same species (McPhail and Carveth 1992). Tables 2, 3 and 4 also indicate relative abundance of each species as estimated by the author.

Figure 3. *The Fraser Basin distribution of the white sucker,* Catostomus commersoni*, a species that drops out of the upper river assemblage between Prince George and the confluence of the Fraser and Bowron rivers.*

Another set of species is found throughout the entire Fraser Basin. These species (Table 6) are ecological generalists tolerant of a wide range of environments. One goal of the Fraser River Action Plan (FRAP) was to identify and assess the suitability of fish species such as these as indicators of aquatic ecosystem health, particularly for bioaccumulative toxic chemicals. One desirable characteristic of an indicator species is a basin-wide distribution in the drainage system. This set of widespread species constitutes the pool of potential indicator species. Most of these fish are resident in the river throughout their life cycle and occupy a variety of habitats, both among species and at various life history stages. The two salmon species only reside in the basin during their first year and in their short spawning period. Figures 4 and 5 show the Fraser distribution of two promising potential indicator species: the mountain whitefish and the peamouth chub (*Mylocheilus caurinus*).

An alternative to using one or a few indicator species is to use fish assemblages to monitor, detect and assess anthropogenic impacts on aquatic ecosystems (Scott and Hall 1997). Fish assemblages and individual species distributions are known to change when habitats are modified (Li *et al.* 1987; Rutherford *et al.* 1987) and various authors have attempted to quantify such changes (for a review see Fore *et al.* 1994). In the Fraser system, in the vicinity of Prince George, it is clear that the fish assemblages in some tributary streams have changed in the last three decades. An example is Wright Creek, a tributary of the Salmon River near Prince George. Table 7 presents a list of the species present in Wright Creek in the early 1960s and in the mid-1990s. In this case, the number of species in the creek has been reduced to almost half the number present three decades ago. Interestingly, the species that have disappeared all require cool, clean water.

Similar shifts in species composition have occurred in Lower Fraser Valley tributaries and, perhaps, in the Thompson system. So far, there is no evidence of species loss in the main stem Fraser; however, it is clear that some species (*e.g.* eulachon, *Thaleichthys pacificus*) have declined dramatically in the last two decades (Hay *et al.* 1997). Some of this decline is due to over-fishing but at least part of the decline probably results from exposure to contaminants in the lower river (Rogers *et al.* 1990). Similarly, white sturgeon (*Acipenser*

Table 5. A comparison of the fish faunas of the Nechako and upper Fraser River (above the confluence with the Bowron River).

SCIENTIFIC NAME	COMMON NAME	NECHAKO	UPPER FRASER
Lampetra tridentata	Pacific lamprey	+	-
Acipenser transmontanus	white sturgeon	+	+
Couesius plumbeus	lake chub	+	+
Hybognathus hankinsoni	brassy minnow	+	-
Mylocheilus caurinus	peamouth chub	+	+
Ptychocheilus oregonensis	northern squawfish	+	+
Rhinichthys cataractae	longnose dace	+	+
Rhinichthys falcatus	leopard dace	+	-
Richardsonius balteatus	redside shiner	+	+
Catostomus catostomus	longnose sucker	+	+
Catostomus columbianus	bridgelip sucker	+	-
Catostomus commersoni	white sucker	+	-
Catostomus macrocheilus	largescale sucker	+	-
Oncorhynchus mykiss	rainbow trout	+	+
Oncorhynchus nerka	sockeye salmon	+	+
Oncorhynchus tshawytscha	chinook salmon	+	+
Salvelinus confluentus	bull trout	+	+
Salvelinus malma	Dolly Varden	+	-
Salvelinus namaycush	lake trout	+	+
Coregonus clupeaformis	lake whitefish	+	+
Prosopium coulteri	pygmy whitefish	+	+
Prosopium williamsoni	mountain whitefish	+	+
Lota lota	burbot	+	+
Cottus asper	prickly sculpin	+	-
Cottus cognatus	slimy sculpin	+	+

+ indicates presence
- indicates absence

Table 6. A list of species found throughout the Fraser Basin (potential indicator species).

SCIENTIFIC NAME	COMMON NAME
Acipenser transmontanus	white sturgeon
Mylocheilus caurinus	peamouth chub
Ptychocheilus oregonensis	northern squawfish
Rhinichthys cataractae	longnose dace
Richardsonius balteatus	redside shiner
Oncorhynchus mykiss	rainbow trout
Oncorhynchus nerka	sockeye salmon
Oncorhynchus tshawytscha	chinook salmon
Salvelinus confluentus	bull trout
Prosopium williamsoni	mountain whitefish

Table 7. Changes in the fish assemblage in Wright Creek (1954–1996).

SPECIES PRESENT IN THE 1950s Scientific and Common Name		PRESENT IN 1996
Oncorhynchus mykiss	rainbow trout	
Oncorhynchus tshawytscha	chinook salmon	✓
Prosopium williamsoni	mountain whitefish	
Catostomus catostomus	longnose sucker	
Catostomus columbianus	bridgelip sucker	✓
Ptychocheilus oregonensis	northern squawfish	✓
Rhinichthys cataractae	longnose dace	✓
Richardsonius balteatus	redside shiner	✓
Lota lota	burbot	
Cottus cognatus	slimy sculpin	

Figure 4. *The Fraser Basin distribution of the mountain whitefish,* Prosopium williamsoni, *a potential indicator species in the Fraser Basin.*

Figure 5. *The Fraser Basin distribution of the peamouth chub,* Mylocheilus caurinus, *a potential indicator species in the Fraser Basin.*

transmontanus) in the Fraser system have declined to the point that they are listed by the Committee on Endangered Wildlife in Canada (COSEWIC) as "vulnerable" (Lane 1991). Consequently, there is no longer a commercial fishery for this species and the recreational fishery is catch and release only. The Fraser distribution of another fish, the Salish sucker (a genetically distinct form of longnose sucker, *Catostomus catostomus*), has contracted dramatically in the last few decades and COSEWIC now lists it as an "endangered" species (McPhail 1987).

Fish assemblages have been used to assess and monitor anthropogenic impacts in other northwestern drainage systems (Friesen and Ward 1996; Hughes and Gammon 1987) and there is no reason to suppose they would not be useful in the Fraser system. Monitoring with fish assemblages would be relatively simple—a series of permanent stations could be identified on the lower, middle and upper Fraser. Initially, to establish a normal range of seasonal and annual variation in species number and abundance, these stations would have to be sampled for several years. Once this baseline was established, the stations likely would only need to be sampled once a decade to detect changes in the state of fish assemblages in the river. An advantage of using fish for this purpose is that no elaborate equipment is needed for sampling and fish are easy (relative to invertebrates) to identify. With their intimate knowledge of the river and concern for traditional fisheries, First Nations groups along the river might provide the monitoring crews.

Potentially, both individual fish species and fish assemblages can be used to monitor the health of aquatic ecosystems. Individual species are probably best used for contaminant monitoring, but they are also useful in detecting local changes in distribution. Typically, the local distribution of species changes before differences in the composition of entire fish assemblages can be detected. For example, the fish assemblage in the Salmon River near Fort Langley contains the same species in 1997 (pers. observ.) as it did in 1954 (Hartman 1968); however, over this same time period, the distribution of the Salish sucker within this river contracted steadily and the species is now confined to only a small section of the upper river (Inglis *et al.* 1992). In contrast to individual species, assemblages are best at detecting subtle environmental changes. This is because interactions among species can be sensitive to minor changes to the environment. For example, the competitive interaction between rainbow trout (*Oncorhynchus mykiss*) and bull trout (*Salvelinus confluentus*) appears to change with temperature—in cold water bull trout are dominant (in terms of numbers), but in only slightly warmer water rainbow trout predominate (Parkinson and Haas 1996). Potentially, changes in the relative abundance of these two species could be used to detect subtle changes in the annual temperature regime of a river. Similarly, the observation that exotic species are associated with altered habitats (Karr 1981) suggests that the ratio of introduced to native species in an assemblage could be used to assess anthropogenic impacts on aquatic ecosystems.

BIOLOGY OF MOUNTAIN WHITEFISH

The rest of this chapter focuses on the considerations that must be taken into account when using mountain whitefish as an indicator species. (FRAP also evaluated peamouth chub as an indicator [Culp and Lowell 1999] and both species were used to assess the health of fish populations throughout the basin [Raymond *et al.* 1999]). While the mountain whitefish has been used as an indicator species for tracking contamination in northwestern rivers like the Columbia (Nener *et al.* 1995), the Fraser (Mah *et al.* 1989; Dwernychuk *et al.* 1993) and the Athabasca/Peace system (Muir and Pastershank 1996), remarkably little is known about its biology. Northcote and Ennis (1994) reviewed the biology of the mountain whitefish throughout its geographic distribution. Their review reveals two aspects of its biology that are of potential concern regarding the species' usefulness as an indicator of fish health and contaminant loads: (1) the complex migrations that appear to be characteristic of this species and (2) the species' propensity to form local populations (stocks). If mountain whitefish regularly move between the main stem Fraser and its tributaries, their exposure to contaminants in the main river may be sporadic or occur only at certain life-history stages. Also, if there are distinct stocks in the region that differ in the pattern of their migrations, these stocks may differ in the length of time they are exposed to main stem contaminants.

From the perspective of using mountain whitefish as an indicator species in the basin, there are two important questions to be answered about their biology: (1) are there different stocks in the upper Fraser system and, specifically, is there a stock resident in the main stem Fraser and (2) do samples collected from fall aggregations of mountain whitefish reflect the contaminant loads of main stem residents, or are these aggregations mixtures of stocks with potentially different histories of exposure to contaminants?

To address these questions, this project examines the movements, stock structure, and general biology of mountain whitefish in the Fraser River and its tributaries near Prince George (McPhail and Troffe 1998). This region was chosen because there were other studies planned under FRAP in this same area to examine the effects of pulp mill effluent on benthic communities and on peamouth chub (Culp and Lowell 1999; Raymond *et al.* 1999).

In British Columbia, most mountain whitefish populations spawn from mid-October through November at temperatures ranging from about 3–5°C (Northcote and Ennis 1994); however, some populations spawn as late as February (McPhail and Lindsey 1970; Anonymous 1997). Spawning was not directly observed in the Prince George area but running-ripe mountain whitefish were collected in October 1995, both in the Nechako River about one km above its confluence with the Fraser and in the main stem Fraser. At about the same time, males with fully developed spawning tubercles were observed in the lower Willow and Bowron rivers. Thus, spawning times in the study area appear to be consistent with other B.C. sites. The eggs incubate over winter and newly emerged fry appear in late April. These fry are distributed throughout the main stem Fraser and its tributaries, suggesting that spawning occurs in both the main river and throughout the length of some major tributaries (*e.g.* the Willow River). Thus, the fry at different sites are exposed to different temperature regimes, and temperature clearly influences the growth rate of the fry. In the warmer western tributaries (*e.g.* Nechako and Chilako rivers), fry grow more rapidly and reach a larger size by the end of the growing season than fry in the cooler eastern tributaries (*e.g.* McGregor and Bowron rivers). These differences in first year growth produce different scale circuli counts in different tributaries and provide a scale signature that stays with the fish for life. The scale signatures on adults taken from a late-fall aggregation of mountain whitefish near the mouth of Naver Creek strongly suggest that the fish in this aggregation are of mixed origins.

Newly emerged fry concentrate in shallow, quiet water (a habitat they share with young-of-the-year chinook salmon). Typically, such sites are deposition areas and the substrate is fine sand or mud. This habitat preference confines the young-of-the-year to the edges of streams and rivers. Thus, in the main stem Fraser this life-history stage will more often come in contact with shore-hugging contaminant plumes than other life-history stages. Towards the end of their first growing season the young-of-the-year move into slightly deeper, faster water. It is not known where they over-winter.

Juveniles (1^+ years of age) also occur throughout most of the major tributaries and the Fraser main stem. The most obvious change in habitat use by juveniles relative to fry is a shift to deeper, faster water. In the main stem Fraser, this habitat shift presumably decreases their chances of encountering contaminants directly, but may not isolate them from exposure to food chain contaminants. Like the young-of-the-year, there are among tributaries growth differences in juveniles that suggest a second summer of residence in their natal stream.

In the Prince George area, most mountain whitefish reach sexual maturity at the end of their third summer (2^+ years of age). Consequently, if there are spawning migrations, it is at this age that the newly mature adults should move. Generally, adult foraging behaviour and habitat use are more complex than that of either fry or juveniles. Adults occupy deeper water than juveniles and, in clear tributaries where underwater observations can be made, adults concentrate where shallow riffles or rapids break into deeper scour pools or sweep around large woody debris. At such sites, aggregations of adult mountain whitefish maintain position in the current just off the bottom and forage on the drifting juvenile stages of aquatic insects. They are often associated with rainbow trout but, unlike trout, mountain whitefish rarely make foraging movements towards the surface. Because of the turbidity, underwater observations could not be made in the main stem Fraser but, presumably, the behaviour of main stem adults is similar to that of adults in tributary streams. If so, the use of deeper water by main stem adults should decrease their exposure to shore-hugging contaminant plumes. However, their prey could be a more important route of exposure to contaminants in any case.

One aspect of morphology that appears to influence foraging in adult whitefish is a dimorphism in head shape (Figure 6). In some adults the snout is exceptionally long and pointed. These so-called "pinocchios" forage closer to the bottom and direct a higher proportion of their foraging movements towards the substrate than "normal" mountain whitefish. In addition, pinocchios are slimmer than normals and this produces a significant difference in their apparent condition (length-weight relationship). Since condition often is used as a measure of fish health, the proportion of pinocchios in a sample will bias estimates of fish health; however, these are not sick fish and should be recorded separately from normal mountain whitefish.

Whether there is a resident main stem stock, and whether the practice of sampling tissues from late-fall aggregations biases contaminant exposure estimates, are critical questions. Consequently, special efforts were made to examine seasonal movements and stock structure in the Prince George area. Five methods provided information on stock structure and movement patterns: radio-tagging, traditional attached tags, laser ablation inductively coupled plasma mass spectrometry (LA-ICPMS) of scales, molecular markers (*i.e.* Restriction Fragment Length Polymorphisms [RFLP]) and scale growth data.

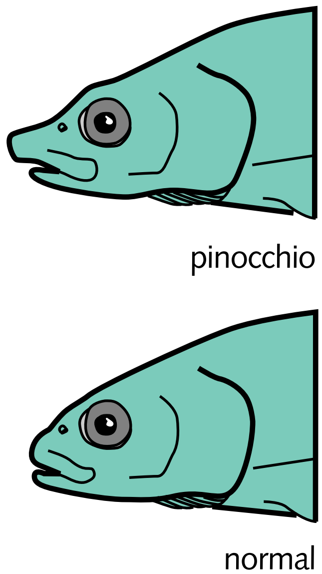

Figure 6. The head morphology of the pinocchio and normal forms of mountain whitefish.

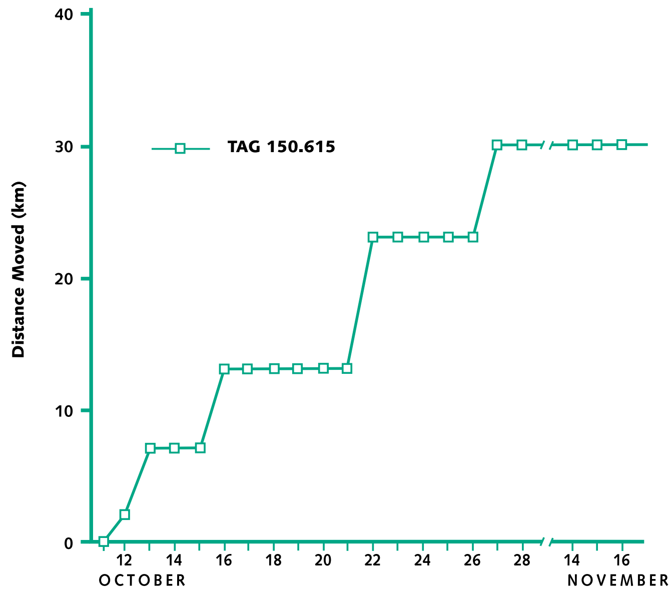

Figure 7. Example of the fall movements of radio-tagged mountain whitefish.

Thirteen radio-tags were applied in the late fall of 1995 and mid-April, 1996. The purpose of the fall tags was to examine spawning and over-wintering migrations. The purpose of the spring tags was to document movement from over-wintering sites to summer feeding sites. The fall tags were applied in the Fraser main stem at Prince George and in the Nechako River about one km above its confluence with the Fraser. According to local anglers, mountain whitefish aggregate at the Nechako site in the fall and remain at this site until spring breakup. The fall pattern of movement in both the Fraser and the Nechako fish was similar (Figure 7). In all cases, movement was downstream and proceeded as a series of stops and starts. Typically, tagged fish held position for a few days, then moved rapidly downstream for several kilometres, held position again for a few days and then started down again. They were tracked from October 11 until November 26, or until the tags could no longer be located. The maximum downstream movement before contact was lost was 40 km. Since all of the fall-tagged fish were adults in, or close to, reproductive condition, these downstream movements may be a spawning migration.

Alternatively, although the tags were the smallest available, they may have been marginal for mountain whitefish and thus the downstream movement could be an artifact. Only five tags were applied in the spring. These were placed on fish angled from the Nechako over-wintering site about one km above the Fraser River. These tags were applied on April 17 and followed until May 11. Like the fall movements, the spring movements proceeded as a series of stops and starts (Figure 8). In this case, however, fish moved upstream into the Chilako River (a Nechako tributary) and downstream into the main stem Fraser. The maximum upstream move-

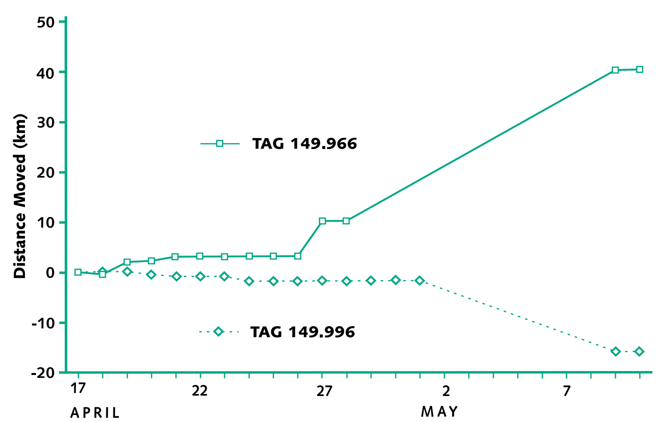

Figure 8. Examples of the spring movements of radio-tagged mountain whitefish (zero indicates release point; positive values indicate upstream movement, and negative values indicate downstream movements).

ment (*i.e.* until contact was lost) was 40 km and the maximum downstream movement was about 15 km.

Although the radio-tag data are limited, they suggest two things. First, that the Nechako site identified as an over-wintering site by local anglers probably is just that—an over-wintering site used by both main stem and tributary fish (Nechako and Chilako). Second, shortly after ice-out, at least some fish move from this site into summer foraging sites in tributary streams and into the main stem Fraser.

Conventional tags were applied in major tributaries (*e.g.* the Bowron, McGregor and Willow rivers) and one minor tributary (Dome Creek), all upstream of Prince George. Dome Creek was chosen because there is a fall aggregation of whitefish near its confluence with the Fraser that was regularly sampled for tissues as a control site in the FRAP fish health assessment program. Thus, if Dome Creek fish were over-wintering at this site, some tags might be recovered during tissue sampling.

Although the number of tags applied in the major tributaries and Dome Creek was relatively small (123), the results of the tag returns are instructive (Table 8). The average time at large for within-season returns was 24 days and all of the within-season returns were within a few metres of the original tagging site. This indicates very little movement during the summer feeding period. Given the low number of tagged fish, the tag returns after one year are startling. Of five fish sampled at the 1995 McGregor River tagging site in 1996, one fish was a recapture from 1995. Similarly, of 14 fish sampled at two tagging sites on Dome Creek in 1997, four were tagged in 1996 at exactly the sites where they were recaptured. These results imply remarkable year-to-year site fidelity in adult mountain whitefish. In Dome Creek, this site fidelity is all the more remarkable since underwater observations and sampling indicated that whitefish had left the tagging areas by the late fall of 1996. It is not known whether they migrated to the main stem Fraser or to the lower

Table 8. *Recaptures of tagged fish (1995–1997).*

WITHIN SEASON RECAPTURES				
LOCALITY	TAGGING DATE	RECAPTURE DATE	ELAPSED TIME (DAYS)	MOVEMENT (m)
McGregor	8/1/95	9/20/95	50	0
Willow	9/14/95	10/24/95	41	0
Dome	6/28/96	8/5/96	38	0
Dome	6/29/96	7/1/96	2	0
Dome	6/29/96	7/19/96	20	0
Dome	7/18/96	9/9/96	72	0
Dome	7/18/96	7/18/96	0.5	0
Dome	7/18/96	8/5/96	18	0
Dome	7/19/96	8/5/96	18	0
Dome	7/19/96	7/24/96	5	0
Dome	7/19/96	7/24/96	5	0
Dome	7/19/96	8/2/96	14	0
Dome	7/24/96	7/25/96	1	0
Dome	7/24/96	9/19/96	57	0
BETWEEN SEASON RECAPTURES				
McGregor	8/1/95	8/1/96	365	0
Dome	6/29/96	7/30/97	396	0
Dome	7/18/96	7/30/97	378	0
Dome	7/19/96	7/30/97	377	0
Dome	8/18/96	7/31/97	357	0

parts of Dome Creek, but it is clear that on their return migration to summer feeding areas, the tagged individuals return to the exact sites they occupied the previous year. If similar summer foraging site fidelity is typical of main stem fish, then, despite possible spawning and overwintering migrations, they could return annually to contaminated foraging sites.

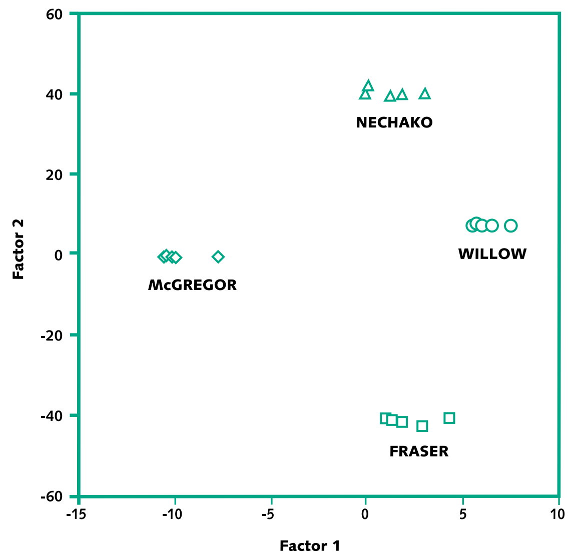

Figure 9. *Principal Components plot based on elemental analyses of mountain whitefish scales from five rivers in the Prince George region.*

Laser ablation inductively coupled plasma mass spectrometry (LA-ICPMS) is a technique that allows the assessment of the relative concentrations of various elements in micro-samples. In this case, scales were used as samples since they are calcified and exhibit annular growth (*i.e.* new layers are laid down each growing season). The technique is still in its infancy and the results should be treated with some caution. Still, some of the results are encouragingly consistent with the tagging data. In one analysis the focus (centre) of a scale from five adult individuals from each of four rivers (the main stem Fraser, Nechako, Willow and McGregor) was analyzed for the ratios of 16 elements (normalized to the concentration of ^{48}Calcium). The fish were sampled during the summer growing season. The results were subjected to Principal Components Analysis and revealed complete separation of the four samples (Figure 9). Since the centre of the scale is laid down in the first growing season, this result implies that the five adult fish in each sample spent their first growing season in the same place—but that the places were different for the four different samples. These results argue that the main stem Fraser and its major tributaries in the Prince George area contain sufficiently distinctive elemental ratios that the portion of the scale laid down in the first growing season can be used to distinguish fish from the main stem and from the different tributaries.

The original intent was to use this initial data to construct a linear discriminant function, and then use this function to classify fall samples taken for contaminant analysis as to stream of origin (*i.e.* to determine if these samples contain a mix of fish of different origins). Unfortunately, without external reference standards of known composition, the technique only provides relative elemental concentrations (normalized to ^{48}Calcium). Thus, the machine must be recalibrated at every run, and its performance changes over time. Consequently, samples run at different times are not directly comparable and, at its present stage of development, the technique cannot be used to reliably classify fish of unknown origin.

Using the LA-ICPMS technique, however, it was possible to assess the temporal pattern of element exposure. Scales from two adult fish were ablated at eight equally spaced sites from the focus (centre) to the edge of the scale. This was done to determine if there were interpretable changes in the elemental composition in the parts of the scale laid down in different years. Since the eight sites on each scale were sampled consecutively on the same run, the relative concentrations of elements in different parts of the scale are directly comparable. Both of the scales chosen for this analysis showed an abrupt change in the spacing of the circuli in the third or fourth year of life and then a return to the previous circuli spacing pattern. This apparent change in growth rate implies a change in habitat. The change in elemental concentration over the scales was not the same for all elements. Some metals (*e.g.* mercury) steadily increased in concentration from the centre of the scale to the edge, other elements (*e.g.* magnesium) steadily decreased along the same axis, whereas the concentration of still other elements was remarkably variable. A few elements (*e.g.* nickel and rubidium), however, showed a pattern of change in concentration consistent with the circuli pattern on the scale. For these elements there is a clear peak in elemental concentration in the third or fourth year of life followed by a return to the original concentration (Figure 10). One possible interpretation of this pattern is that the fish moved to a new environment, perhaps for spawning or over-wintering, and then returned to the original environment. If this interpretation is correct, it suggests long-term fidelity to a specific elemental environment with occasional short-term shifts to different environments.

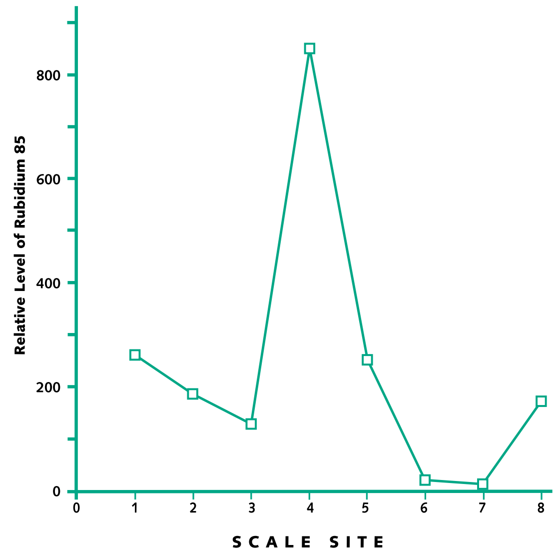

Figure 10. *Relative concentration of ^{85}rubidium at eight equally spaced intervals across a single adult whitefish scale.*

To determine if there are genetically different stocks of mountain whitefish in the Prince George area, tissue samples from the main stem Fraser and different tributaries were examined for mitochondrial restriction fragment length polymorphisms (RFLP). This technique (Lansman *et al.* 1981) identified six haplotypes in the Prince George area. All sexually reproducing organisms receive one set of nuclear genes from their father and another set from their mother. This allows two forms (alleles) of a gene to coexist in a single organism. In contrast to nuclear DNA, mitochondrial DNA is passed only through the mother. Consequently, each individual possesses only a single set of mitochondrial genes and thus a single allele at each locus. These single alleles are called haplotypes.

Two of the haplotypes identified in mountain whitefish (A and B) were common and geographically widespread, while the other four haplotypes (C, D, E and F) were rare and with more restricted distribution. The frequency of the two common haplotypes varied from site to site but there was no evidence of significant differences in the frequency of the common haplotypes among sites. Interestingly, however, the normal and pinocchio forms of mountain whitefish differed dramatically in the frequency of the two common haplotypes. Haplotype A occurred with about equal frequency in both normals and pinocchios, but haplotype B occurred only in pinocchios. This is clear evidence of a genetic difference between the normal and pinocchio forms of mountain whitefish. How this genetic difference is maintained is unknown, but one possibility is assortative mating. Since mitochondrial haplotypes are passed through the maternal lineage, perhaps normal males rarely, if ever, mate with pinocchio females.

Generally, the frequencies of the rarer haplotypes (C, D, E and F) are too low to permit comparisons among sites. There is, however, one instructive comparison. It is clear that haplotype D is more common in the main stem Fraser than it is in any of the tributaries and, if all of the tributary sites are pooled and compared with the main stem samples, there is a significant ($P<0.05$) difference in the frequency of haplotype D between the tributaries and the main river. Thus, the RFLP analysis suggests the presence of a distinct main stem stock.

In summary, the results of this project indicate that the life history, habitat use and movements of mountain whitefish in the Prince George area are similar to what has been found in other areas (Northcote and Ennis 1994). There is evidence of age-related habitat shifts; complex and individually variable migrations for spawning, over-wintering and summer-foraging sites; different foraging forms; and different stocks. In spite of this biological complexity, mountain whitefish have a number of characteristics that suit them for the role of an indicator species in the Fraser Basin. First, they occur throughout the basin and this permits comparison of samples from pristine sites with samples from sites where the fish are exposed to contaminants. Second, they are sufficiently long-lived (up to 20 years) that there is time for individuals to accumulate contaminants and also time for them to reach a body size that provides sufficient tissue for contaminant analysis. Third, their diet (primarily nymphs and pupae of aquatic insects) is unlikely to be seriously confounded by inputs of terrestrial or aerial contaminants. Fourth, at least some individuals appear to show remarkable fidelity to summer foraging sites. Thus, in spite of evidence from this, and other studies (Pettit and Wallace 1975; Thompson and Davies 1976) for complex spawning, over-wintering, and summer foraging migrations, there is evidence that individual fish return to summer foraging sites that were used in previous years. This suggests that although seasonally some adults move over considerable distances, their patterns of movement can be consistent from year to year. Consequently, on an annual basis, fish that forage in sites exposed to contaminants may return to those sites for a number of summers.

The biggest problem with using mountain whitefish as an indicator species in the main stem Fraser is the difficulty of sampling the species. For much of the year the adults in the main river appear to be dispersed in deep water. They will not enter baited traps, and the depth and turbidity of the water precludes electroshocking from a boat. In tributaries, angling with worms on small hooks is a slow but reliable sampling system for adults and the largest juveniles; however, angling is not effective in the main river. In the Prince George area, the only effective sampling gear is a large seine set from a boat, and even this gear only produces useful catches in the fall when mountain whitefish aggregate near the mouths of tributaries. Furthermore, tagging studies in Idaho (Pettit and Wallace 1975) found that over-wintering aggregations of mountain whitefish were made up of mixed stocks (*i.e.* fish that use different summer foraging sites). In the upper Fraser system, the available evidence supports the hypothesis that fish sampled from fall aggregations represent a mixture of stocks that may have had different histories of exposure to contaminants. This makes the interpretation of contaminant loads in fish from such sites difficult, especially if the samples are pooled or averaged over several fish. Given the difficulty of obtaining adequate samples of adult fish at other times, there is no simple alternative to sampling these fall aggregations.

A method of identifying which stocks individuals belong to is critical to the continued use of mountain whitefish as an indicator species. In other salmonids, the analysis of micro-satellite DNA (Tautz 1989) has proven sensitive enough to distinguish adjacent stocks (Angers *et al.* 1995). Micro-satellite loci consist of repeated short nucleotide sequences. In theory the number of possible repeats is infinite, thus, theoretically, the number of possible alleles at such loci is infinite. In practice, the number of alleles that can be distinguished at a micro-satellite locus is not infinite; however, it is often large and it is this profusion of distinguishable alleles that make microsatellites such a powerful means of detecting genetic differences between populations (Avise 1994). Since micro-satellite analyses are quick, relatively inexpensive and require no more tissue than a small fin clip, they might provide a solution to the mixed stock problem in mountain whitefish.

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Table 1. *Native fishes of the Fraser Basin (48 species), including the eight marine species that regularly enter the river.*

SCIENTIFIC NAME	COMMON NAME	LIFE HISTORY
Lampetra ayresi	river lamprey	A
Lampetra richardsoni	western brook lamprey	F
Lampetra tridentata	Pacific lamprey	A
Acipenser medirostris	green sturgeon	A
Acipenser transmontanus	white sturgeon	A, F
Acrocheilus alutaceus	chiselmouth	F
Couesius plumbeus	lake chub	F
Hybognathus hankinsoni	brassy minnow	F
Mylocheilus caurinus	peamouth chub	F
Ptychocheilus oregonensis	northern squawfish	F
Rhinichthys cataractae	longnose dace	F
Rhinichthys falcatus	leopard dace	F
Richardsonius balteatus	redside shiner	F
Catostomus catostomus	longnose sucker	F
Catostomus columbianus	bridgelip sucker	F
Catostomus commersoni	white sucker	F
Catostomus macrocheilus	largescale sucker	F
Catostomus platyrhynchus	mountain sucker	F
Spirinchus thaleichthys	longfin smelt	A, F
Thaleichthys pacificus	eulachon	A
Oncorhynchus clarki clarki	coastal cutthroat trout	A, F
Oncorhynchus clarki lewisi	westslope cutthroat trout	F
Oncorhynchus gorbuscha	pink salmon	A
Oncorhynchus keta	chum salmon	A
Oncorhynchus kisutch	coho salmon	A
Oncorhynchus mykiss	rainbow trout	A, F
Oncorhynchus nerka	sockeye salmon	A, F
Oncorhynchus tshawytscha	chinook salmon	A
Salvelinus confluentus	bull trout	A, F
Salvelinus malma	Dolly Varden	A, F
Salvelinus namaycush	lake trout	F
Coregonus clupeaformis	lake whitefish	F
Prosopium coulteri	pygmy whitefish	F
Prosopium williamsoni	mountain whitefish	F
Lota lota	burbot	F
Gasterosteus aculeatus	threespine stickleback	A, F
Cottus aleuticus	coastrange sculpin	C, F
Cottus asper	prickly sculpin	C, F
Cottus cognatus	slimy sculpin	F
Cottus rhotheus	torrent sculpin	F
Marine species:		
Clupea pallasi	Pacific herring	M
Hypomesus pretiosus	surf smelt	M
Microgadus proximus	Pacific tomcod	M
Cymatogaster aggregata	shiner perch	M
Clevelandia ios	arrow goby	M
Leptocottus armatus	Pacific staghorn sculpin	M
Synchirus gilli	manacled sculpin	M
Platichthys stellatus	starry flounder	M

A = anadromous [migrates between the sea and freshwater but spawns in freshwater]; C = catadromous [migrates between the sea and freshwater but spawns in salt or brackish water]; F = freshwater residents; M = marine

Table 2. *The Fraser estuarine fish assemblage (58 species), including five introduced species.*

SCIENTIFIC NAME	COMMON NAME	LIFE HISTORY	STATUS	ABUNDANCE
Lampetra ayresi	river lamprey	A	N	++
Lampetra richardsoni	western brook lamprey	F	N	++
Lampetra tridentata	Pacific lamprey	A	N	++
Acipenser medirostris	green sturgeon	A	N	+
Acipenser transmontanus	white sturgeon	A, F	N	++
Alosa sapidissima	American shad	A	I	+
Clupea pallasi	Pacific herring	M	N	++
Cyprinus carpio	carp	F	I	++
Hybognathus hankinsoni	brassy minnow	F	N	++
Mylocheilus caurinus	peamouth chub	F	N	+++
Ptychocheilus oregonensis	northern squawfish	F	N	+++
Rhinichthys cataractae	longnose dace	F	N	++
Rhinichthys falcatus	leopard dace	F	N	+
Richardsonius balteatus	redside shiner	F	N	+++
Catostomus catostomus	longnose sucker	F	N	+
Catostomus macrocheilus	largescale sucker	F	N	+++
Ameiurus nebulosus	brown bullhead	F	I	++
Hypomesus pretiosus	surf smelt	M	N	++
Mallotus villosus	capelin	M	N	++
Spirinchus thaleichthys	longfin smelt	A, F	N	+++
Thaleichthys pacificus	eulachon	A	N	+++
Oncorhynchus clarki clarki	coastal cutthroat trout	A, F	N	++
Oncorhynchus gorbuscha	pink salmon	A	N	+++
Oncorhynchus keta	chum salmon	A	N	+++
Oncorhynchus kisutch	coho salmon	A	N	+++
Oncorhynchus mykiss	rainbow trout	A, F	N	+++
Oncorhynchus nerka	sockeye salmon	A, F	N	+++
Oncorhynchus tshawytscha	chinook salmon	A	N	+++
Salmo salar	Atlantic salmon	A	I	+
Salvelinus confluentus	bull trout	A, F	N	++
Salvelinus malma	Dolly Varden	A, F	N	++

Prosopium williamsoni	mountain whitefish	F	N	++
Microgadus proximus	Pacific tomcod	M	N	+
Aulorhynchus flavidus	tubesnout	M	N	+
Gasterosteus aculeatus	threespine stickleback	A, F	N	+++
Syngnathus griseolineatus	bay pipefish	M	N	+
Cymatogaster aggregata	shiner perch	M	N	++
Lumpenus sagitta	Pacific snake prickleback	M	N	++
Apodichthys flavidus	penpoint gunnel	M	N	+
Pholis laeta	crescent gunnel	M	N	+
Pholis ornata	saddleback gunnel	M	N	+
Ammodytes hexapterus	Pacific sandlance	M	N	+
Clevelandia ios	arrow goby	M	N	++
Hexagrammos decagrammus	kelp greenling	M	N	+
Artedius lateralis	smoothhead sculpin	M	N	+
Asemichthys taylori	spinynose sculpin	M	N	+
Cottus aleuticus	coastrange sculpin	C, F	N	++
Cottus asper	prickly sculpin	C, F	N	+++
Enophrys bison	buffalo sculpin	M	N	+
Leptocottus armatus	Pacific staghorn sculpin	M	N	++
Oligocottus maculosus	tidepool sculpin	M	N	+
Synchirus gilli	manacled sculpin	M	N	+
Pomoxis nigromaculatus	black crappie	F	I	++
Citharichthys sordidus	Pacific sanddab	M	N	+
Pleuronectes isolepis	butter sole	M	N	+
Pleuronectes vetulus	English sole	M	N	+
Platichthys stellatus	starry flounder	M	N	+++
Psettichthys melanostictus	sand sole	M	N	+

A = anadromous [migrates between the sea and freshwater but spawns in freshwater]; C = catadromous [migrates between the sea and freshwater but spawns in salt or brackish water]; F = freshwater residents; M = marine
N = native; I = introduced
+ = rare; ++ = modestly common; +++ = common

Table 3. The fish assemblage (44 species) in the lower river between Mission and the canyon.

SCIENTIFIC NAME	COMMON NAME	LIFE HISTORY	STATUS	ABUNDANCE
Lampetra ayresi *	river lamprey	A	N	++
Lampetra richardsoni *	western brook lamprey	F	N	++
Lampetra tridentata	Pacific lamprey	A	N	++
Acipenser medirostris *	green sturgeon	A	N	+
Acipenser transmontanus	white sturgeon	A, F	N	++
Alosa sapidissima †	American shad	A	I	+
Carassius auratus	goldfish	F	I	+
Cyprinus carpio	carp	F	I	+++
Hybognathus hankinsoni	brassy minnow	F	N	+++
Mylocheilus caurinus	peamouth chub	F	N	+++
Pimephales promelas †	fathead minnow	F	I	++
Ptychocheilus oregonensis	northern squawfish	F	N	+++
Rhinichthys cataractae	longnose dace	F	N	+++
Rhinichthys falcatus	leopard dace	F	N	++
Richardsonius balteatus	redside shiner	F	N	+++
Catostomus catostomus	longnose sucker	F	N	+
Catostmous macrocheilus	largescale sucker	F	N	+++
Catostomus columbianus	bridgelip sucker	F	N	++
Catostomus platyrhynchus	mountain sucker	F	N	+
Ameiurus nebulosus †	brown bullhead	F	I	++
Spirinchus thaleichthys *	longfin smelt	A, F	N	+++
Thaleichthys pacificus *	eulachon	A	N	+++
Oncorhynchus clarki clarki *	coastal cutthroat trout	A, F	N	+++
Oncorhynchus gorbuscha	pink salmon	A	N	+++
Oncorhynchus keta *	chum salmon	A	N	+++
Oncorhynchus kisutch	coho salmon	A	N	+++
Oncorhynchus mykiss	rainbow trout	A, F	N	+++
Oncorhynchus nerka	sockeye salmon	A, F	N	+++
Oncorhynchus tshawytscha	chinook salmon	A	N	+++
Salmo solar	Atlantic salmon	A	I	+
Salvelinus confluentus	bull trout	A, F	N	++
Salvelinus fontinalis	brook trout	F	I	++
Salvelinus malma	Dolly Varden	A, F	N	++
Salvelinus namaycush	lake trout	F	T	+
Coregonus clupeaformis	lake whitefish	F	T	+
Prosopium williamsoni	mountain whitefish	F	N	+++
Lota lota	burbot	F	N	+
Gasterosteus aculeatus *	threespine stickleback	A, F	N	+++
Cottus aleuticus	coastrange sculpin	C, F	N	+++
Cottus asper	prickly sculpin	C, F	N	+++
Perca flavescens †	yellow perch	F	I	+
Lepomis gibbosus †	pumpkinseed	F	I	++
Micropterus salmoides †	largemouth bass	F	I	+
Pomoxis nigromaculatus †	black crappie	F	I	++

F = freshwater residents; A = anadromous [migrates between the sea and freshwater but spawns in freshwater]; C = catadromous [migrates between the sea and freshwater but spawns in salt or brackish water]; M = marine
N = native; I = introduced; T = transplanted
+ = rare; ++ = modestly common; +++ = common
** = native species restricted to the lower river; † = introduced species restricted to lower river*

Table 4. The fish assemblage (35 species) in the upper river above the canyon.

SCIENTIFIC NAME	COMMON NAME	LIFE HISTORY	STATUS	ABUNDANCE
Lampetra tridentata	Pacific lamprey	A	N	++
Acipenser transmontanus	white sturgeon	F	N	++
Acrocheilus alutaceus *	chiselmouth	F	N	++
Carassius auratus	goldfish	F	I	+
Couesius plumbeus *	lake chub	F	N	+++
Cyprinus carpio	carp	F	I	++
Hybognathus hankinsoni	brassy minnow	F	N	++
Mylocheilus caurinus	peamouth chub	F	N	+++
Ptychocheilus oregonensis	northern squawfish	F	N	+++
Rhinichthys cataractae	longnose dace	F	N	+++
Rhinichthys falcatus	leopard dace	F	N	+++
Richardsonius balteatus	redside shiner	F	N	+++
Catostomus catostomus	longnose sucker	F	N	+++
Catostomus columbianus	bridgelip sucker	F	N	+++
Catostomus commersoni *	white sucker	F	N	+++
Catostomus macrocheilus	largescale sucker	F	N	+++
Catostomus platyrhynchus	mountain sucker	F	N	+
Oncorhynchus clarki lewisi *	westslope cutthroat trout	F	N	+
Oncorhynchus gorbuscha	pink salmon	A	N	++
Oncorhynchus kisutch	coho salmon	A	N	++
Oncorhynchus mykiss	rainbow trout	A, F	N	+++
Oncorhynchus nerka	sockeye salmon	A, F	N	+++
Oncorhynchus tshawytscha	chinook salmon	A	N	+++
Salvelinus confluentus	bull trout	F	N	++
Salvelinus fontinalis	brook trout	F	I	++
Salvelinus malma	Dolly Varden	A	N	+
Salvelinus namaycush *†	lake trout	F	N	++
Coregonus clupeaformis *†	lake whitefish	F	N	++
Prosopium coulteri *	pygmy whitefish	F	N	++
Prosopium williamsoni	mountain whitefish	F	N	+++
Lota lota	burbot	F	N	++
Cottus aleuticus	coastrange sculpin	F	N	+
Cottus asper	prickly sculpin	F	N	+++
Cottus cognatus *	slimy sculpin	F	N	++
Cottus rhotheus *	torrent sculpin	F	N	+

** = native species (or subspecies) originally restricted to the upper river; † = transplanted to lower river basin*
A = anadromous [migrates between the sea and freshwater but spawns in freshwater]; F = freshwater residents
I = introduced; N = native
+ = rare; ++ = modestly common; +++ = common

3.8

FISH HEALTH ASSESSMENT

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The health of resident fish populations is an important indicator of ecosystem conditions, and a factor that can directly affect human health. Resident fish spend their entire life span in the river and reflect local conditions more than migrants such as salmon. Previous fish studies in the Fraser Basin concentrated on the biology, habitat and population sizes of commercially important migrants such as salmon and trout (Northcote and Burwash 1991). The Fraser River Action Plan (FRAP) provided a unique opportunity to conduct a basin-wide survey of resident fish tissue contaminant levels coincident with an assessment of fish health. FRAP assessed the condition of resident fish in the Fraser Basin, based on the contaminant levels and health of mountain whitefish (*Prosopium williamsoni*) and peamouth chub (*Mylocheilus caurinus*).

A relatively good historical record of fish tissue contaminants exists for the lower Fraser River. Pollution concerns in the early 1970s led to sampling the lower river for persistent organochlorine pesticides and PCBs in fish tissue (Hall *et al.* 1991; Albright *et al.* 1975). Fish tissue contaminant monitoring in the lower river continued as a result of the Fraser River Estuary Study (Singleton 1983; Swain 1986; Swain and Walton 1989) and because of concerns about lumber treatment facility run-off and sewage treatment plant effluent (Rogers and Hall 1987; Carey *et al.* 1988; Rogers *et al.* 1990; Rogers *et al.* 1992; Birch and Shaw 1995).

Information on contaminants in the upper basin has been collected more sporadically. Peterson *et al.* (1971) measured heavy metals in fish from lakes throughout British Columbia and Derksen (1986) measured metals in rainbow trout and mountain whitefish in the Thompson River. In the 1980s, fears about chlorophenolics, dioxins and furans from pulp mills resulted in research on effects of contaminants on juvenile salmonids and other ecosystem components in the upper Fraser and Thompson rivers and a province-wide dioxin survey which included sites on the Fraser and Thompson rivers (Rogers and Mahood 1982; Rogers and Mahood 1983; Rogers *et al.* 1988; Mah *et al.* 1989; Tuominen and Sekela 1992; Servizi *et al.* 1993). Dioxin and furan surveys were repeated by the pulp mills from 1990 to 1992 (Dwernychuk *et al.* 1993).

Mountain whitefish were selected for this study because previous research indicated that they accumulate dioxins and furans to higher levels than other species sampled (Mah *et al.* 1989; Dwernychuk *et al.* 1993; Pastershank and Muir 1995), and research was being conducted on their life history in the upper Fraser River (McPhail 1999). Peamouth chub were selected because they are widely distributed and abundant in the basin. They were a target of pulp mill effects research in the upper Fraser River (Gibbons *et al.* 1995) and were used in the Environmental Effects Monitoring Program (EEM) for pulp and paper mills on rivers in British Columbia.

Because the autopsy-based fish health assessment of Goede and Barton (Goede and Barton 1990) is being widely applied in environmental studies (Adams *et al.* 1993; Hatfield Consultants Ltd. 1996a; 1996b; Healey 1997), it has been incorporated in this study to test whether the health assessment abnormalities could be confirmed by histological analyses.

METHODS

Details of methods are presented in Raymond *et al.* (1999). Adult mountain whitefish and peamouth chub were collected between July and November in 1994, 1995 and 1996, from up to 11 reaches in the Fraser River Basin (Figure 1). The Hansard and North Thompson reaches are located above all major effluent discharges on the Fraser and Thompson rivers, respectively, and were chosen as reference reaches for this study. The Nechako reach is also above all major discharges in the basin, but most of its flow is diverted out of the basin for hydroelectric power generation.

In each reach, approximately 60 fish of each species were examined for external and internal abnormalities, according to Goede and Barton (1990). For the purpose of data analysis, the abnormalities were converted to a numerical health assessment index (HAI) based on Adams *et al.* (1993), with the following modifications: 1) fatty livers were considered normal in peamouth chub and 2) the thymus, pseudobranchs and blood parameters were not assessed. A higher HAI represents a higher incidence and severity of abnormalities. Fish ages were read

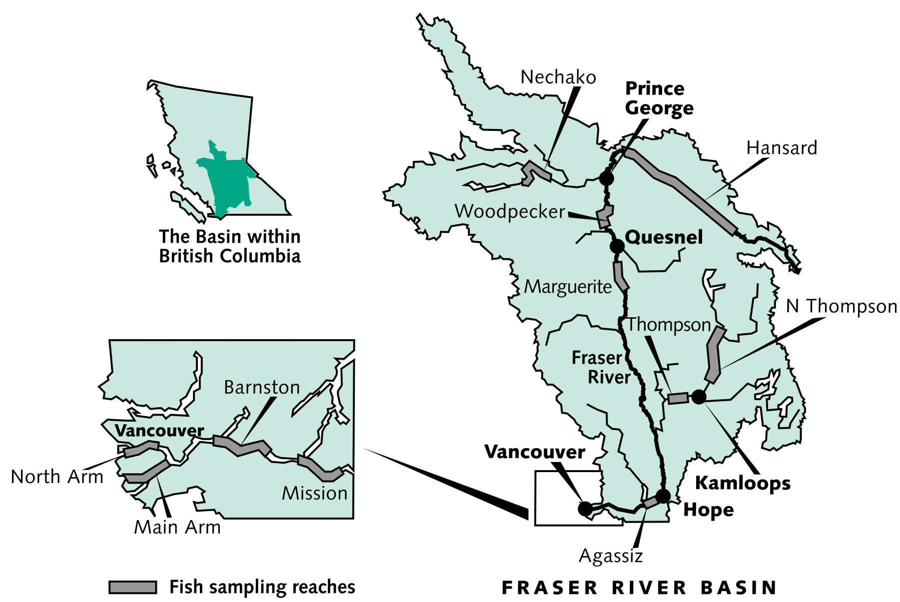

Figure 1. Fish sampling reaches in the Fraser Basin.

from surfaces and burnt cross-sections of the sagittal pair of otoliths. Tissue samples from the gill, liver, spleen, kidney, hindgut and pyloric caecae were analyzed following standard histological procedures (Humasen 1979). The histological evaluation noted only the presence of abnormalities without quantitative analysis of the extent of the abnormalities within each tissue section.

As an indicator of exposure to certain organic contaminants, activity of liver mixed-function oxygenase (MFO) enzymes was measured, as ethoxyresorufin-o-deethylase (EROD) activity, in up to 10 male and 10 female fish per reach. Organochlorine analyses were conducted on composite tissue samples by Axys Analytical (Sidney, B.C.) using standard extraction and cleanup protocols, with subsequent quantitation using gas chromatography (GC) and low-resolution mass spectrometry (MS) (chlorophenolics, PCB congeners, pesticides, resin acids) or high-resolution GC and high resolution MS (coplanar PCBs and dioxins/furans).

A suite of 12 metals was analyzed: arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium and zinc. Tissues from the estuary in 1994 were analyzed for most metals by inductively coupled plasma-atomic emission spectroscopy (ICP-AES), for mercury by cold-vapour AES and for arsenic, lead and selenium by graphite furnace AES (Quanta Trace Laboratories Inc., Burnaby, B.C.). All other samples were submitted to the National Laboratory for Environmental Testing, Burlington, Ontario, where arsenic and selenium were measured by ICP, mercury by cold-vapour atomic absorption spectrophotometry and all other metals by atomic absorption spectrophotometry. Laboratory and field QA/QC protocols were implemented to ensure data quality and validity for all samples.

CONTAMINANTS

Pesticides

Concern about the environmental effects of pesticides, particularly persistent organochlorine insecticides has led to their ban or highly restricted use throughout most of the world. Sources in the Fraser Basin are likely a combination of residual contamination from historical use, long-range atmospheric transport (Allan 1989; Kurtz 1990) and possible clandestine application of remaining stocks. Of the 24 pesticides determined in the analytical scan, all but two, lindane (gamma-hexachlorocyclohexane) and endosulphan, are banned in Canada. Many, including toxaphene and DDT, remain in use elsewhere in the world, particularly India, Central America and Russia (Voldner and Li 1993; Voldner and Li 1995).

The two registered pesticides were not the most frequently detected compounds in Fraser Basin fish in 1994 and 1995. Dieldrin, DDE (a metabolite of DDT), heptachlor epoxide, hexachlorobenzene and trans-nonachlor (a component of chlordane) were detected in more than 80 per cent of all muscle and liver analyses in both years. The highest tissue concentrations were of DDE and toxaphene, both of which are highly stable, bioaccumulative and readily transported atmospherically (Donald *et al.* 1993; Sanchez *et al.* 1993; Voldner and Li 1993).

Toxaphene and DDE concentrations on the main stem Fraser increased from the headwater sites with the highest levels in both peamouth chub and mountain whitefish being found in the Agassiz reach (Figure 2, Figure 3). Inter-species differences in body burden were apparent, probably related principally to differences in tissue lipid content, which is a strong determinant of organochlorine accumulation. The effect of lipid was particularly evident in peamouth chub liver, where lipid levels (12-30%) were typically two to six times those of mountain whitefish livers from the same sample reaches.

Industrial Chemical Contaminants

Organic chemical contaminants released by industry, particularly bleach-kraft pulp and paper mills, have long been a concern in the Fraser River Basin. The bleaching process releases a large volume of discharges

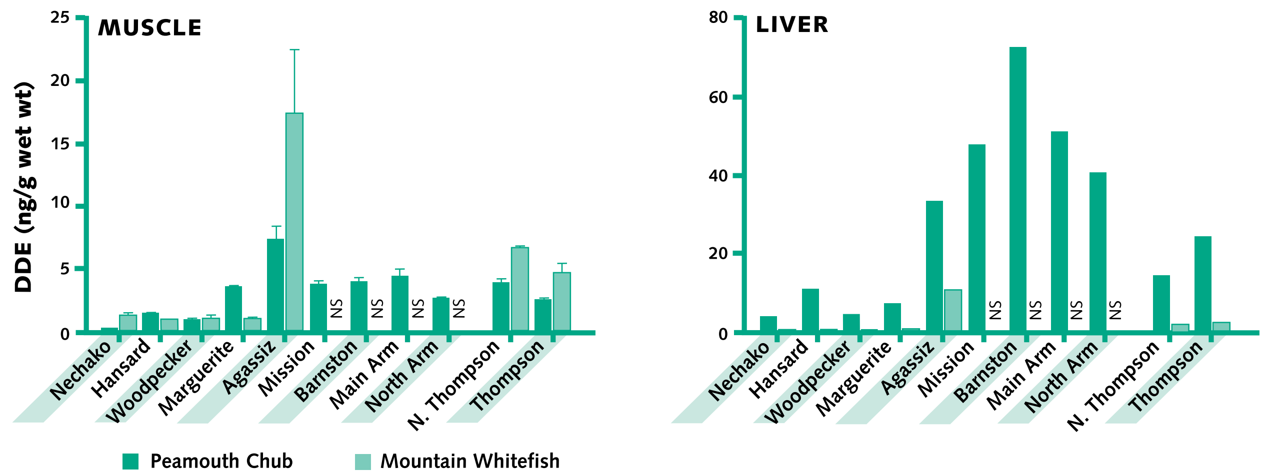

Figure 2. *DDE in fish tissues from the Fraser River Basin, 1994. Values are mean (with standard error); four to five composites for muscle, and one or two composites for liver samples. NS denotes no samples.*

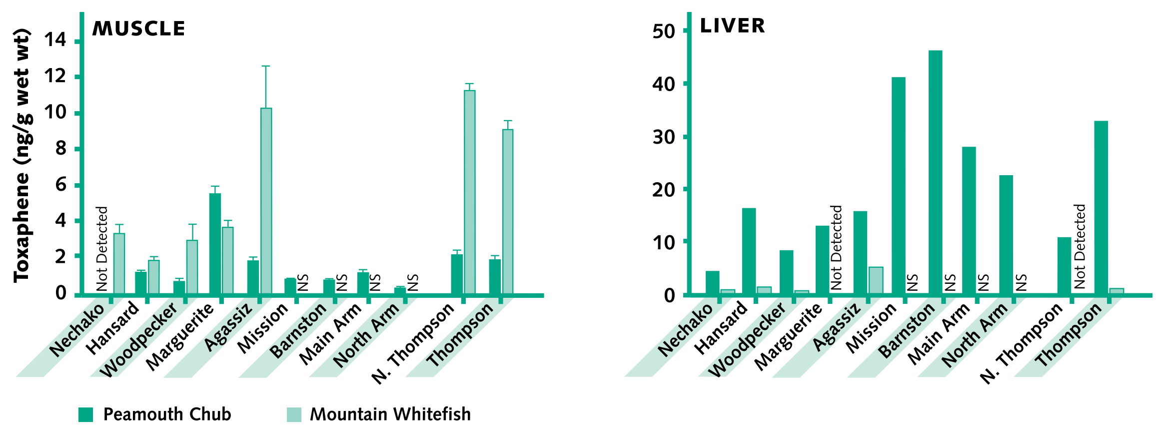

Figure 3. *Total toxaphene in fish tissues from the Fraser River Basin, 1994. Values are mean (with standard error); four to five composites for muscle, and one or two composites for liver samples. NS denotes no samples.*

with a huge number and spectrum of compounds (Dimmel *et al.* 1993; Owens 1991; Paasivirta *et al.* 1992). Particularly worrisome are the chlorinated dibenzodioxins, dibenzofurans and chlorophenolics, which are produced as byproducts of the chlorine bleaching process.

Throughout the Fraser Basin, dioxin and furan detections were dominated by the nearly ubiquitous 2,3,7,8 -tetrachlorodibenzofuran (TCDF) and, to a lesser extent, the more toxic 2,3,7,8-tetrachloro-dibenzo-*p*-dioxin (TCDD), both of which are characteristic components of pulp and paper effluent (Cleverly *et al.* 1997; Trudel 1991). TCDD concentrations were low (<1 pg/g wet weight) in mountain whitefish liver and rarely detected in either mountain whitefish muscle or peamouth chub tissues. The highest levels in 1994 were measured in the Thompson basin, both upstream and downstream of the pulp mill in Kamloops. Dioxin and furan congeners with dioxin-like activity may be expressed as toxic equivalent units (TEQs) relative to 2,3,7,8-TCDD (NATO 1988). Tissue levels of these TEQs throughout the Fraser Basin were well below the current tissue residue guidelines for consumption by humans, but did in some cases exceed a proposed guideline for protection of wildlife (Figure 4).

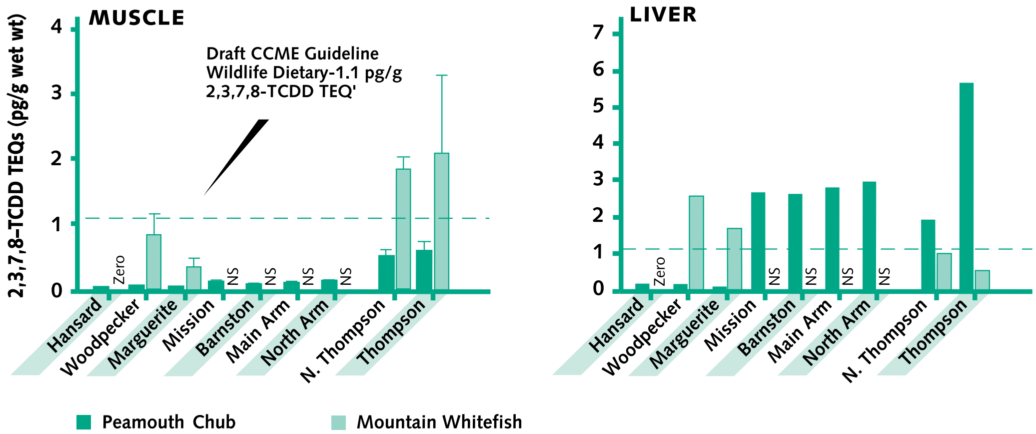

Figure 4. Total dioxin and furan 2,3,7,8-TCDD TEQs in tissues. Calculated using NATO I-TEFs (NATO 1988) with NDs=0. Values are mean (with standard error); four to five composites for muscle, and one or two composites for liver samples. NS denotes no samples.

[1]*CCME 1995 (draft)*

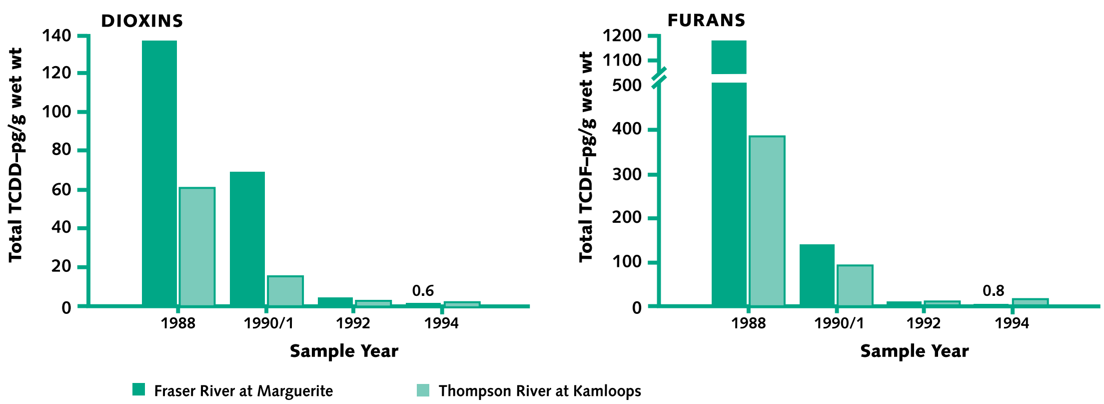

Figure 5. Dioxin and furan declines in mountain whitefish tissues following process changes at Fraser Basin pulp and paper mills.

Recent process changes undertaken by B.C. pulp mills (BC Ministry of Environment, Lands and Parks 1995; Krahn 1995), have resulted in dramatic declines in both effluent and environmental levels of organochlorine contaminants, including the most toxic dioxin congeners. From the extreme levels measured by Mah *et al.* (1989), levels in fish tissue have dropped to near-detection limits (Figure 5).

Chlorophenolics are another important class of contaminants released from pulp mills, wood treatment facilities and municipal wastewater treatment facilities in the Fraser Basin (Norecol 1993). Of the 47 chlorophenols and related compounds (guaiacols, catechols, etc.) targeted in the tissue analyses, nearly all were at low concentrations (<1 ng/g) or below detection in both muscle and liver. This result was not unexpected, since chlorophenolics are rapidly transformed to polar metabolites and excreted in bile (Brumley *et al.* 1996; Oikari 1986; Oikari and Holmbom 1986). They tend to have very short residence periods in tissues. Relatively high chlorophenol concentrations were detected in both peamouth chub and mountain whitefish bile downstream of pulp mill effluent sources (Figure 6). Two features are apparent in the patterns of chlorophenolics in bile. First, high concentrations are indicative of near-field, recent exposures, as was evident near mill effluent sources at Marguerite. The collection sites near Marguerite were as little as 20

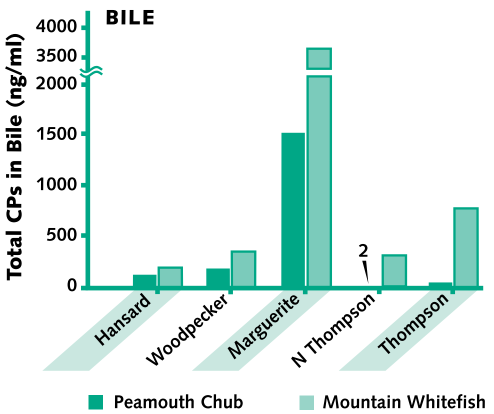

Figure 6. Total chlorophenolics (CPs) in bile in fish from the Fraser River Basin, 1994.

kilometres downstream of the mill effluent discharge at Quesnel, with consequently elevated chlorophenolics in bile of both fish species. Second, the similarity in bile total chlorophenolic concentration between the two reaches in the Thompson Basin indicates that mountain whitefish, in particular, may be moving considerable distances over relatively short periods of time.

Resin acids are also accumulated to high levels in bile (Oikari and Holmbom 1986) and their presence in bile is a sensitive indicator of recent exposure to water-borne resin acids. These are a group of naturally occurring wood extractives which are released from cut timber, as from sawmills, chipping mills, pulp and paper processing or similar industrial operations. Resin acid concentrations in peamouth chub and mountain whitefish bile from reaches near sawmill operations (Hansard and North Thompson reaches, Table 1) were greatly elevated, even compared to samples collected downstream of pulp and paper mills.

As with the organochlorine pesticides, use of polychlorinated biphenyls (PCBs) has been severely restricted since the 1980s. Residues persist in the environment due to the high stability and resistance of the compounds to degradation. Potential sources are many, from atmospheric transport to landfill leachate. Measurements of a total of 84 of the 209 possible PCB congeners in fish tissues in 1994 and 1995 indicated that 1) the contaminant sources (as represented by congener patterns) do not appear to be localized in particular reaches, 2) concentrations seem to be similar throughout the basin and 3) PCB concentrations and congener patterns are quite stable between years.

Table 1. Total resin acids in bile, 1994. Concentrations in µg/ml.

	HANSARD	WOODPECKER	N. THOMPSON	THOMPSON
Peamouth Chub	420.4	1.2	12.9	<0.7

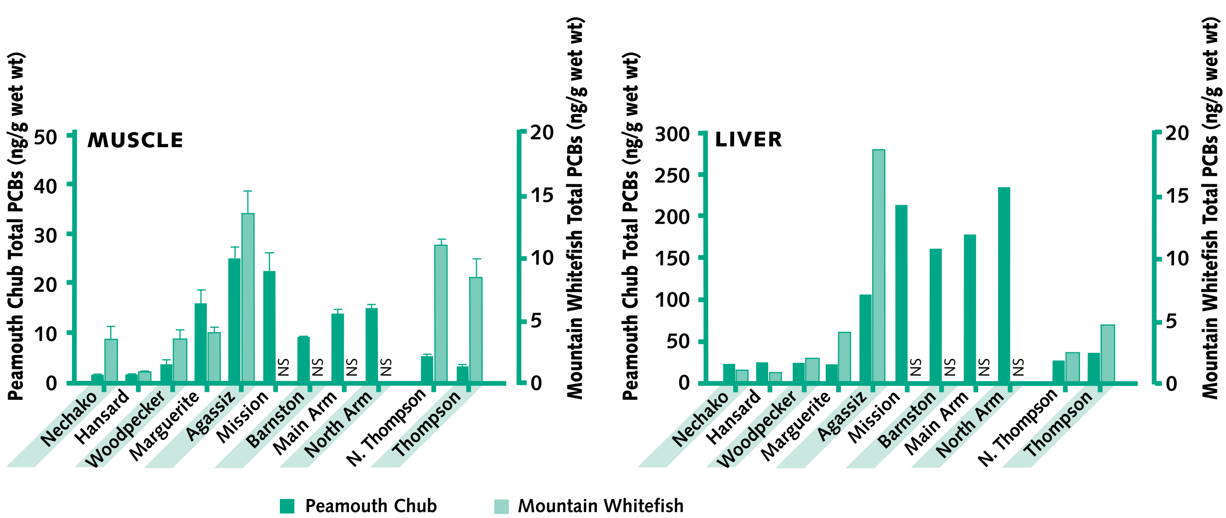

Figure 7. Total PCBs in fish from the Fraser River, 1994. Values are mean (with standard error); four to five composites for muscle, and one or two composites for liver samples. NS denotes no samples.

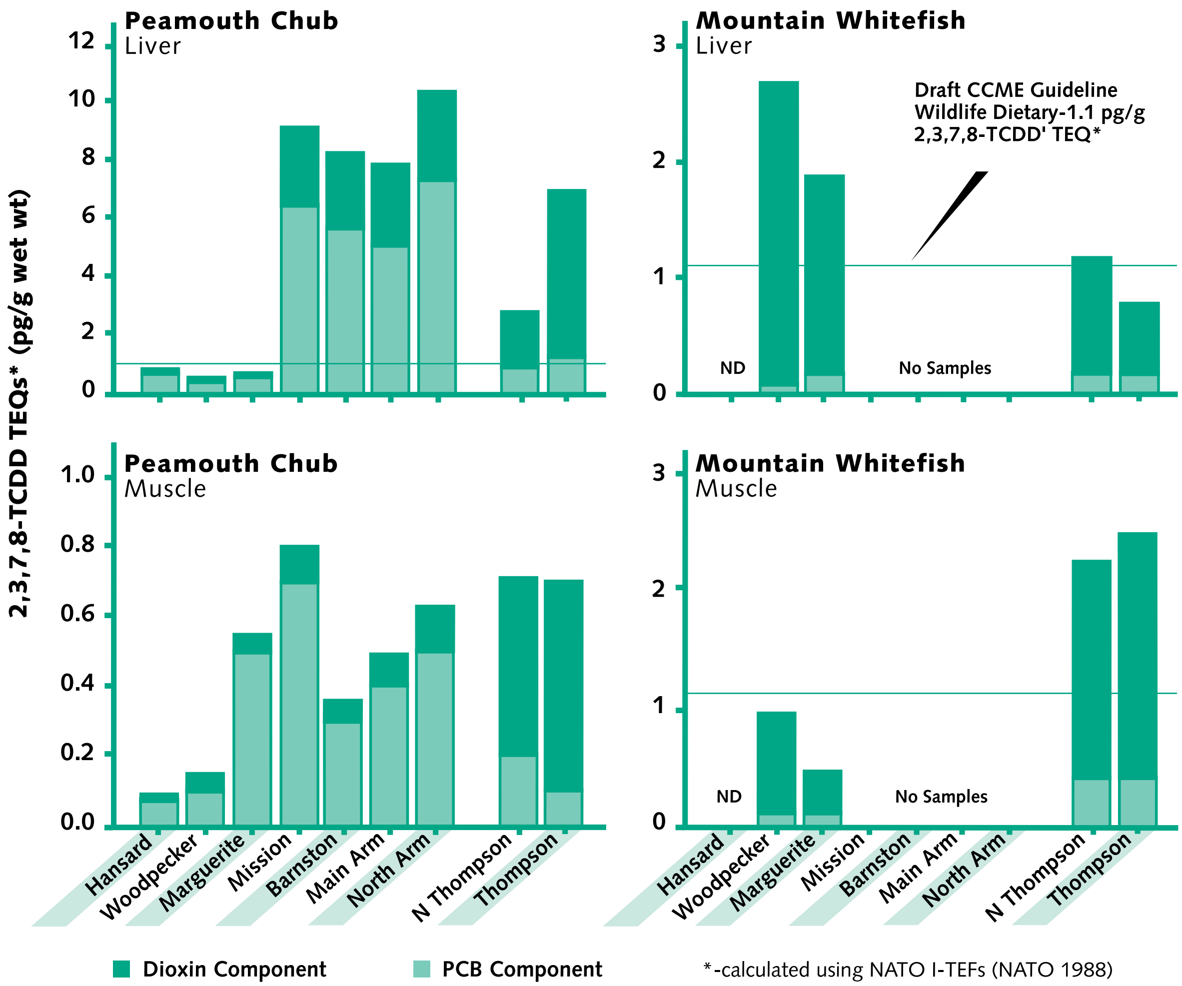

Figure 8. *Total TCDD TEQs as contributed by both dioxin/furan congeners and dioxin-like PCBs.*
[1] *CCME 1995 (draft)*

Total PCB concentrations measured in Fraser fish tissues were far below the current provincial criterion for human consumption (2000 ng/g wet wt.; BC MELP 1998) (Figure 7). However, when the toxicological significance of the level is considered, the potential exists for effects on wildlife consumers. A number of PCBs, particularly the coplanar congeners, have dioxin-like activity which may be represented as “toxic equivalence factors” (TEF) relative to 2,3,7,8-TCDD. Applying these factors in calculating a total toxic burden, it is clear that dioxin-like activity related solely to PCBs can exceed that related to dioxins and furans themselves (Figure 8). This is particularly evident in the lipid-rich livers of peamouth chub in the lower Fraser sampling reaches.

Tissue concentrations of polycyclic aromatic hydrocarbons (PAHs) were measured in peamouth chub tissues from the lower Fraser River reaches in 1994. This class of organic compounds is of particular concern to aquatic health because of the known carcinogenicity and probable teratogenicity of many members (Varanasi 1989). PAHs result from a wide array of sources and processes and are important indicators of urban and industrial contamination. Levels of both low and high molecular weight PAHs in peamouth chub tissue clearly show this association in the lower Fraser (Figure 9). Measured levels were highest in the Fraser River North Arm, a reach which receives a high number of stormwater and industrial discharges.

No metals were in excess of Canadian tissue residue guidelines for consumption by humans, and tissue levels were within expected ranges for fish from uncontaminated sites in British Columbia (Rieberger 1992).

Some levels of total selenium in mountain whitefish liver were slightly higher than the 3 µg/g B.C. tissue residue guideline for the protection of human health, but are unlikely to be of concern to consumers.

Overall, organic contaminant concentrations in fish tissues throughout the basin were low relative to existing environmental and human health guidelines, and have declined substantially from historical levels. Despite the success in removing or reducing release of persistent organochlorines in the environment, residues continue to be detected in fish tissues. The cumulative effect of these contaminants and their metabolic transformation products should remain a subject of some concern.

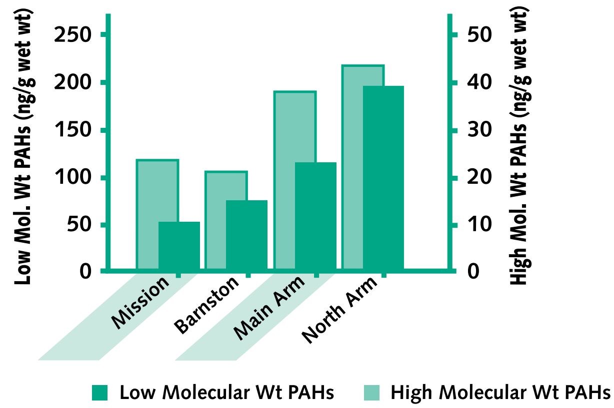

Figure 9. PAHs in peamouth chub liver from the lower Fraser to estuary.

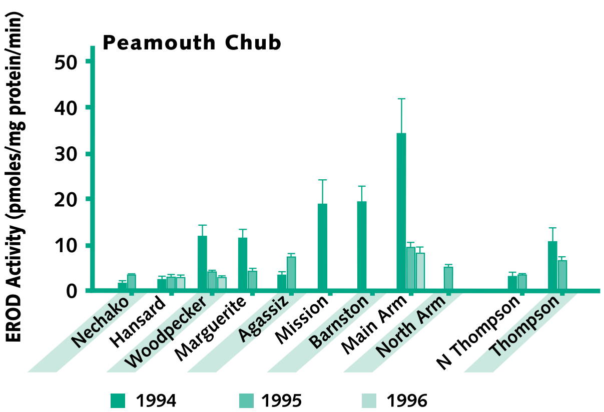

Figure 10. EROD activity levels in peamouth chub in the Fraser River Basin from 1994 to 1996.

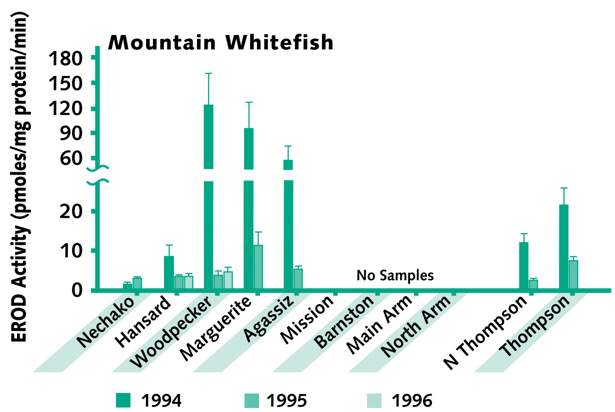

Figure 11. EROD activity levels in mountain whitefish in the Fraser River Basin from 1994 to 1996.

MFO ACTIVITY

MFO enzymes are consistent indicators of exposure to certain organic contaminants found in pulp mill effluent and urban runoff. EROD activity was induced downstream of urban centres and pulp mills in the Fraser Basin. (Fig. 10, Fig. 11). EROD induction in peamouth chub was highest in the Main Arm. Compared to reference reaches, up to 18- and 78-fold increases were seen in peamouth chub (Main Arm) and mountain whitefish (Woodpecker), respectively. These levels of induction were above levels seen in mountain whitefish collected downstream of pulp mills from the Wapiti River (30-fold), where there were no observed health effects (Swanson *et al.* 1993), and from the Columbia River (13-fold) where health effects were observed (Nener *et al.* 1995). There were no significant relationships between EROD induction and health effects in this study.

Higher EROD induction was measured in 1994 than in 1995 and 1996. The year to year variability could be attributed to either the low flow condition of the Fraser River in 1994 or a change in methodology between 1994 and 1995.

Data on contaminants in suspended and bed sediments were collected by FRAP studies concurrently from the same reaches as EROD activity (Sekela *et al.* 1995; Brewer *et al.* 1998; Sylvestre *et al.* 1998). The strongest relationship between EROD activity and body burdens of specific contaminants or classes of contaminants, or sediment contaminant levels, was with TEQs (dioxin, furan and PCB) in mountain whitefish liver in 1994 (r=0.89, p=0.05, n=5) (Figure 13). The correlation between EROD and TEQs in peamouth chub liver was weaker (r=0.70, p=0.05, n=8) (Figure 12); additionally, there were weak significant relationships between peamouth chub EROD and liver pesticide residues as well as bed sediment retene. The weak correlations between specific contaminants and EROD activity may be due to low contaminant body burdens, an unmeasured inducer, and/or synergistic/additive effects of complex mixtures.

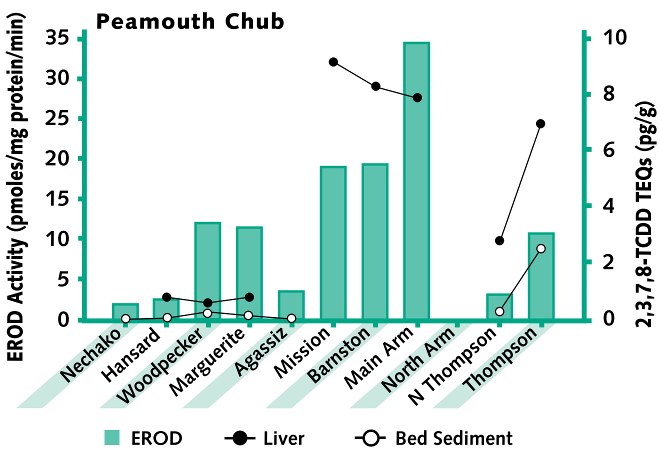

Figure 12. *EROD activity levels and dioxin, furan and PCB TEQs for peamouth chub liver and bed sediment in the Fraser River Basin in 1994.*

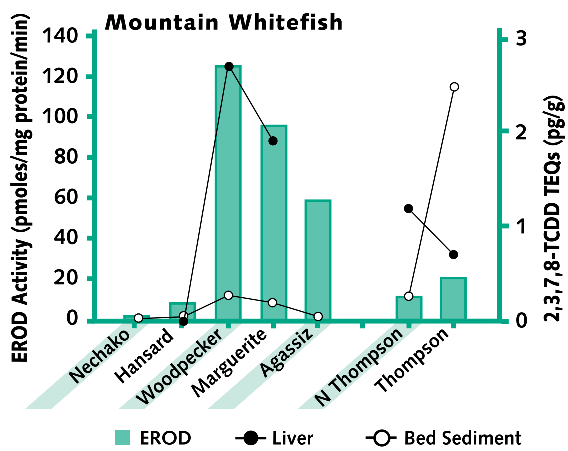

Figure 13. *EROD activity levels and dioxin, furan and PCB TEQs for mountain whitefish liver and bed sediment in the Fraser River Basin in 1994.*

FISH CONDITION, ENERGY RESERVES, GROWTH RATES, AND REPRODUCTIVE INDICATORS

Fish condition indices, based on the length-weight relationship, energy reserves, growth rates, and relative gonad size (gonadosomatic index) are generally used as indicators of the well-being of fish (Adams and Ryon 1994; Munkittrick 1992; Goede and Barton 1990). Low condition, energy reserves, and gonadosomatic index may be caused by stress such as contaminant exposure, but they also fluctuate seasonally with feeding activity, migrations and sexual maturation. In this study, energy reserves were assessed using lipid levels in major lipid storage depots—muscle for mountain whitefish and liver for peamouth chub—and a qualitative mesenteric fat index for both. Fish lengths, adjusted for the basin-wide mean ages (size-at-age), were used as indicators of growth rates. Condition indices and fat reserves are presented in Figures 14 and 15, size-at-age in Figures 16 and 17, and gonadosomatic indices in Figures 18 and 19. Mesenteric fat indices are not presented but they exhibited patterns similar to the lipid levels. Higher condition indices, lipid reserves, growth rates, and gonadosomatic indices infer better condition.

Peamouth chub condition indices, fat reserves and growth rates were all highest in the Nechako, Thompson and lower Fraser rivers. This was probably due to higher temperatures and lower sediment levels in the Thompson and Nechako rivers, and higher temperatures in the lower Fraser resulting in higher productivity. Peamouth chub condition and growth rates were lowest at Marguerite in the central basin downstream of Prince George and Quesnel. The low condition and growth rates may be due to contaminant inputs but were likely due to high suspended sediment levels causing stress on fish and reducing productivity in the river. Scrivener *et al.* (1994) suggested that sediment levels in the Fraser River downstream of Quesnel in

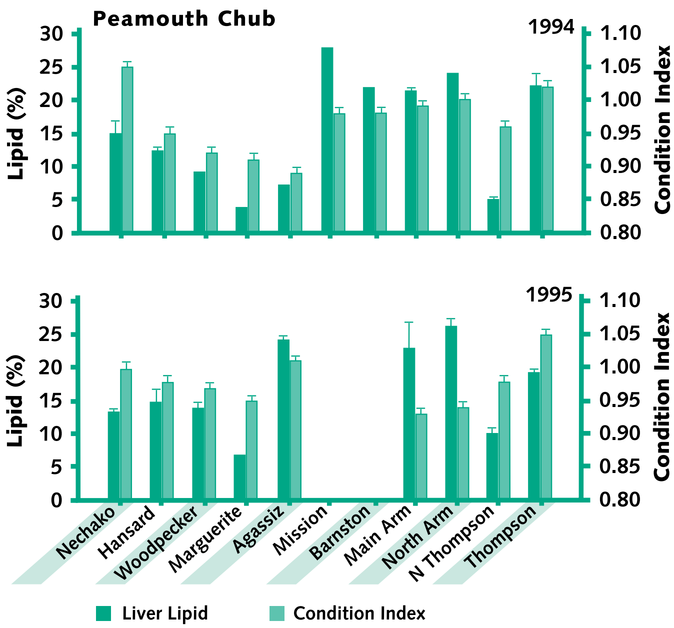

Figure 14. *Peamouth chub liver lipid levels and condition indices in the Fraser River Basin from 1994 and 1995.*

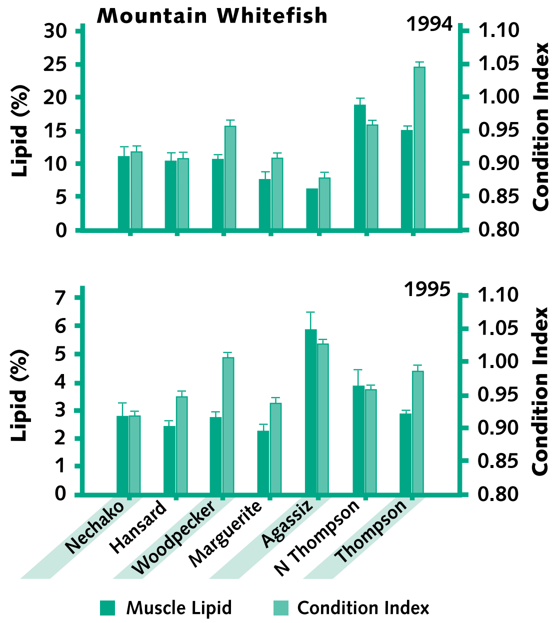

Figure 15. *Mountain whitefish muscle lipid levels and condition indices in the Fraser River Basin in 1994 and 1995.*

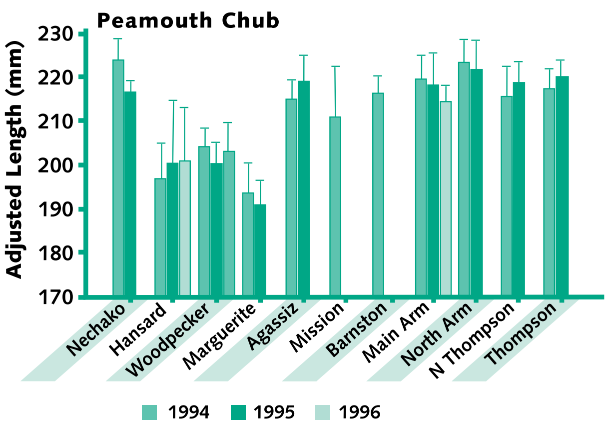

Figure 16. *Peamouth chub size-at-age—mean lengths adjusted for mean age 6.03 years.*

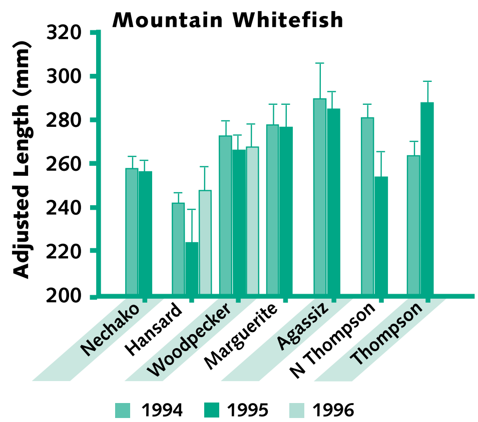

Figure 17. *Mountain whitefish size-at-age—mean lengths adjusted for mean age 4.10 years.*

spring and summer were high enough to harm juvenile salmonids by disrupting feeding, growth and social behaviour and increasing susceptibility to disease.

In contrast, mountain whitefish condition indices and lipid reserves were generally similar among reaches in the Nechako and Fraser rivers, and highest in the Thompson basin reaches. Condition was high at Woodpecker and Agassiz (in 1995) compared to other Fraser reaches. Growth rates increased from north to south (upstream to downstream), indicating that latitude and altitude were the primary controlling factors.

Mountain whitefish trends may be different from peamouth chub because mountain whitefish can migrate long distances (Swanson *et al*. 1993) and may not be good indicators of environmental factors at their capture sites. McPhail (1999) studied the movement of mountain whitefish in the Fraser River near Prince George. McPhail's research indicates that individual mountain whitefish return to the same summer forag-

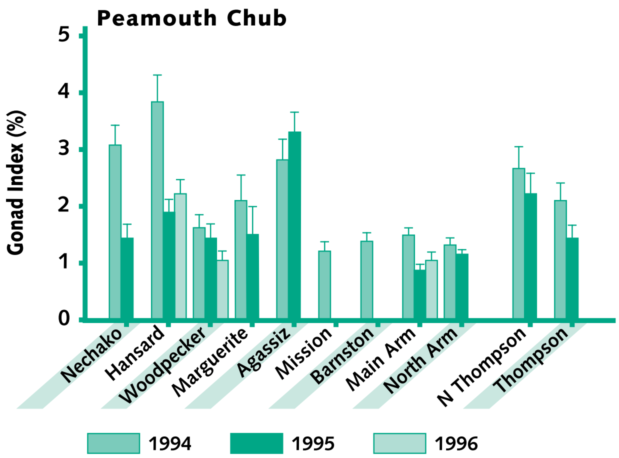

Figure 18. Peamouth chub gonadosomatic index – gonad size as a percentage of body weight.

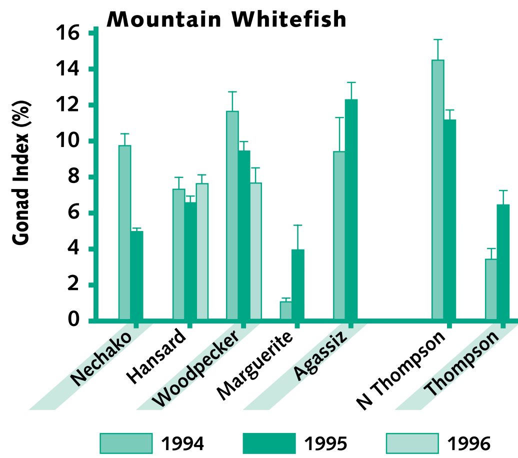

Figure 19. Mountain whitefish gonadosomatic index – gonad size as a percentage of body weight.

ing sites every year and that there is very little movement during the summer feeding period. However, fall aggregations in the main stem and large tributaries, which were sampled in this study, may contain mountain whitefish from various summer feeding locations, including smaller tributaries.

A standard weight equation has been calculated for mountain whitefish. This equation was developed by Rogers *et al.* (1996) to allow fisheries managers to compare condition of mountain whitefish among populations over a wide geographical range. Relative weights are calculated from the equation to produce values similar to the condition index, with the optimal relative weight being 100 per cent. Mean relative weights for the Fraser Basin ranged from 78.6 per cent to 91.7 per cent, except at Agassiz in 1995 (101.1%). These data indicate that mountain whitefish from the Fraser Basin were thinner than optimal for the species and that problems may exist in food and feeding relationships (Anderson and Gutreuter 1983). However, genetic dimorphism in Fraser Basin mountain whitefish populations, with some adults having a long, slender snout and thinner bodies ("pinnochios"), may result in low condition indices which may bias estimates of fish health (McPhail 1999).

Although sampling dates were partially standardized to eliminate seasonal effects, reaches were sampled a month earlier in 1995 than in 1994. It may be noted from the figures that condition and lipid reserves at Agassiz, the latest-sampled reach, were lowest in both species in 1994 but highest in 1995. These year-to-year differences in condition may reflect seasonal effects due to the change in sampling dates. In addition, fish were sampled in a north (Hansard) to south (Agassiz) progression, from mid-September to early November in 1994. The upstream to downstream decrease in peamouth chub condition indices and fat reserves in 1994 may also be partially explained by seasonal effects. However, the same pattern was still evident in 1995 when the sampling was conducted earlier.

Gonadosomatic indices (GSI) were measured as an indicator of reproductive maturity and capacity. Generally, larger GSI indicates greater maturity and capacity. Age- or size-at-maturity are also important indicators because the age and size at which fish achieve reproductive maturity affects reproductive capacity of the population (Weatherley and Gill 1987). For both fish species, GSI were low and age/size-at-maturity were high in the central basin at Marguerite and in the Thompson River compared to upstream reaches, indicating impaired reproductive capacity. This was particularly the case for mountain whitefish females from Marguerite, which had small, and sometimes swollen and discoloured gonads, despite large body size. Perhaps mature mountain whitefish move out of these reaches to spawn, *i.e.*, into tributaries from the central main stem Fraser or up the North Thompson from the Thompson River; therefore we could be sampling only from populations that were not spawning. However, the fact that both reaches are immediately downstream of pulp mill inputs and urban centres suggests that more study is warranted to assess whether or not contaminants are causing the impairment.

FISH HEALTH

An elevated HAI has been linked to contaminant exposure and associated decreased growth and condition in other studies (Adams *et al.* 1993; Adams *et al.* 1996). In the Fraser Basin, HAIs did not increase downstream of urban centres and pulp mills and, therefore, did not appear to be associated with contaminant exposure or EROD activity (Figure 20, Figure 21). In fact, HAIs at reference reaches, where contaminant levels were low, were generally higher than or comparable to other reaches in the basin. The highest HAIs occurred in the Nechako River, due primarily to heavy parasite infestations. In addition, the pattern of variability in the HAI was different from those of condition, lipid levels and growth. Therefore, in this study, observed "abnormalities" used to compute the HAI cannot be linked to decreased growth and condition.

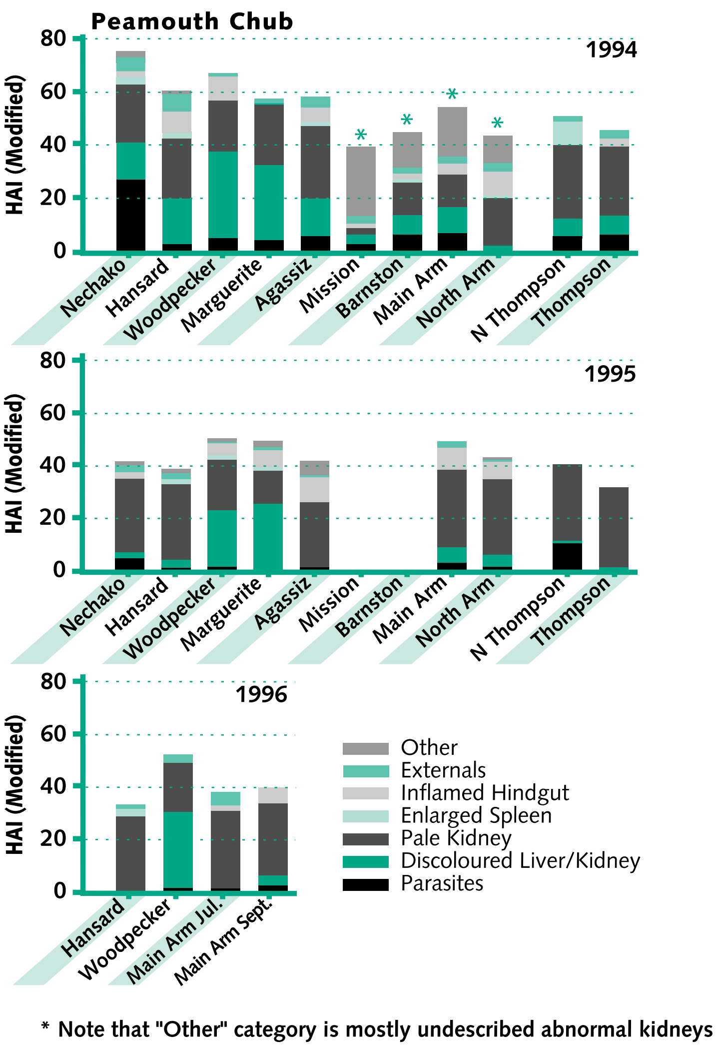

Figure 20. Peamouth chub HAIs at reaches in the Fraser River Basin from 1994 to 1996.

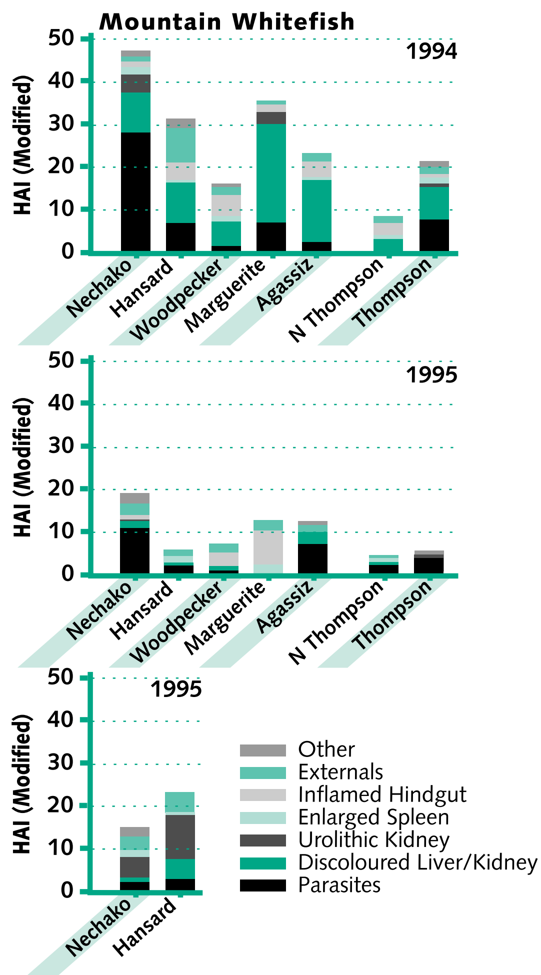

Figure 21. Mountain whitefish HAIs at reaches in the Fraser River Basin from 1994 to 1996.

One possible response to contaminant exposure was a high incidence of discoloured (highly pigmented) organs of both species downstream of Prince George and Quesnel at Woodpecker and Marguerite, respectively. Discoloured or dark kidneys and livers were also seen in mountain whitefish sampled downstream of a pulp mill and metals smelter at Trail on the Columbia River (Nener *et al.* 1995) and in peamouth chub sampled for the pulp mill EEM program at Prince George, Woodpecker and Quesnel (Hatfield Consultants Ltd. 1996a).

HAIs and their component abnormalities varied between species, among reaches and even between years within the same reaches, confounding the interpretation of the data. Examples of this variability are:

1. Peamouth chub HAIs were higher than mountain whitefish primarily because all peamouth chub kidneys were swollen and most were pale. Pale, swollen kidneys may be normal for this species or be caused by a myxosporidean parasite that infested all peamouth chub collected in this study.

2. HAIs and parasite ratings were highest in 1994, possibly a result of the extreme low flows and high water temperatures in the Fraser Basin in 1994.

3. Mountain whitefish from Marguerite had the second highest HAIs (after Nechako) in both 1994 and 1995, the result of discoloured livers in 1994 and inflamed hindguts in 1995.

In spite of observing the above health assessment abnormalities, microscopic (histological) examination of tissues indicated that tissues were generally in good condition. As with the HAI, histological abnormalities did not appear to be related to contaminant exposure, since incidences were as high at reference reaches as at downstream reaches. Most histological abnormalities were attributed to parasite infestations, which were most common in the Nechako River.

CONCLUSIONS AND RECOMMENDATIONS

- Peamouth chub and mountain whitefish were successfully used as indicators of contaminant exposure in fish in the Fraser Basin – they were captured throughout the basin and showed differences in contaminant levels and MFO induction between sites upstream and downstream of contaminant sources.

- Levels of organochlorine contaminants and metals in tissue were virtually unchanged between 1994 and 1995, were low relative to existing environmental and human health guidelines, and have declined significantly from historical levels. These compounds could be measured at long-term intervals to monitor trends unless some local change dictates a closer interval.

- Despite the success in removing or reducing release of persistent organochlorines in the environment, residues continue to be detected in fish tissues. The cumulative effect of these contaminants and their metabolic transformation products should remain a subject of some concern.

- The value of peamouth chub and mountain whitefish as indicator species for the Fraser Basin will be enhanced with a better understanding of their life histories to support interpretation of health and contaminant data. Data need to be acquired on patterns of movement and geographic and temporal variability in parasitism, condition, biochemistry, physiology, reproduction and growth. Results from this study support this recommendation. Two years was too short a time period to establish baselines of fish condition—variability between the years in the health assessment, somatic indices and growth was significant, particularly at Agassiz and in the Thompson basin.

- The increase in PAHs in fish tissue in the estuary is of concern because of continuing population growth in the Lower Fraser Valley. Levels and effects of these compounds should continue to be monitored to assess the effectiveness of urban pollution abatement activities. Suitable methods for monitoring PAH biomarkers or metabolites need to be developed and applied.

- Environmental levels of non-bioaccumulating compounds, such as current-use pesticides and surfactants, and their effects on fish, need to be assessed.
- One goal of this study was to assess effects of contaminants on fish health in the Fraser Basin. However, elevated contaminant levels in environmental samples could not be linked to elevated HAI or decreased growth and condition because the specific factors affecting these variables could not be isolated. Many natural factors, such as flow, sediment loads and temperature, which affect fish health, growth and condition, were too variable over the large geographic area sampled to allow for a separation of effects due to contaminants alone. Local, sub-basin comparisons over a gradient of contaminant levels or long term experimental exposures may be more successful in assessing the effects of contaminants on fish health.
- Contaminant levels and fish health were assessed in the main stem Fraser and its largest tributaries because of concerns about high-volume point sources of contaminants in these reaches. As contaminants from non-point sources in small tributaries flowing through urban and agricultural areas are not as diluted as those entering the main stem or larger tributaries, it is likely that fish condition and HAI responses would be more definite in such streams. It is recommended that selected streams exposed to non-point sources should be assessed with techniques utilized in this study.

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3.9

CONTAMINANTS IN WILDLIFE INDICATOR SPECIES FROM THE FRASER RIVER BASIN

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The Fraser River Basin originally contained large areas of pristine riparian and wetland habitat important for many resident and migratory wildlife species. The intensive use and alteration of these habitats by agricultural, urban and industrial development have lead to concerns about the viability and health of local wildlife populations. Scientists have focused their research on a small number of wildlife species which are used to gauge the health of the entire wildlife community. These species are known as 'indicator species.'

Discrete breeding and/or resident populations of four bird species and two mammal species have been extensively monitored along the Fraser River since the early 1980s. These species embody many of the elements that characterize a good indicator species (Moul *et al.* 1996). In an assessment of 57 indigenous amphibian, bird and mammal species, four birds (great blue heron, osprey, bald eagle and tree swallow) and two mammals (river otter and mink) monitored in our studies achieved among the highest scores for potential use as indicator species in the Fraser River Basin (Moul *et al.* 1996). Ranking criteria were based on published knowledge and the practicality of sample collection. Criteria included: (1) fundamental attributes of residency, distribution, abundance and diet; (2) current understanding and documentation of natural history, such as home range, habitat, migration and historic abundance; (3) ease of collecting samples; (4) suitability of each species for laboratory study; and (5) availability of knowledge on contaminant responses. Five of the six species selected (great blue heron, osprey, bald eagle, mink, river otter) had comparatively high scores also because they have been monitored in the Fraser River Basin before. Consequently, data are available on their contaminant levels and biology. Each of these species has its utility as an indicator affected by its distinct habitat requirements and regional distribution as described below.

The great blue heron was selected as an indicator species in the Lower Fraser Valley because it is a year-round resident with a dominantly piscivorous diet. Herons eat mostly one-year-old fish or younger (Harfenist *et al.* 1993), which implies that contaminants acquired through the food chain originate locally. Double-crested cormorant, while not evaluated in the Moul *et al.* (1996) report because of its marine habitat preference, was also added to the Lower Fraser Valley indicator group because it possesses many characteristics of a good indicator species, and there is a breeding colony in the Fraser delta. Heron and cormorant eggs were collected from breeding colonies in the estuary (Fig. 1). Cormorants, however, were found to forage beyond the Fraser estuary to the Strait of Georgia; thus they better represent the condition of the southern Georgia Basin, rather than specifically the Fraser estuary. Herons have been regularly monitored since the late 1970s; cormorants have been sampled periodically since 1985.

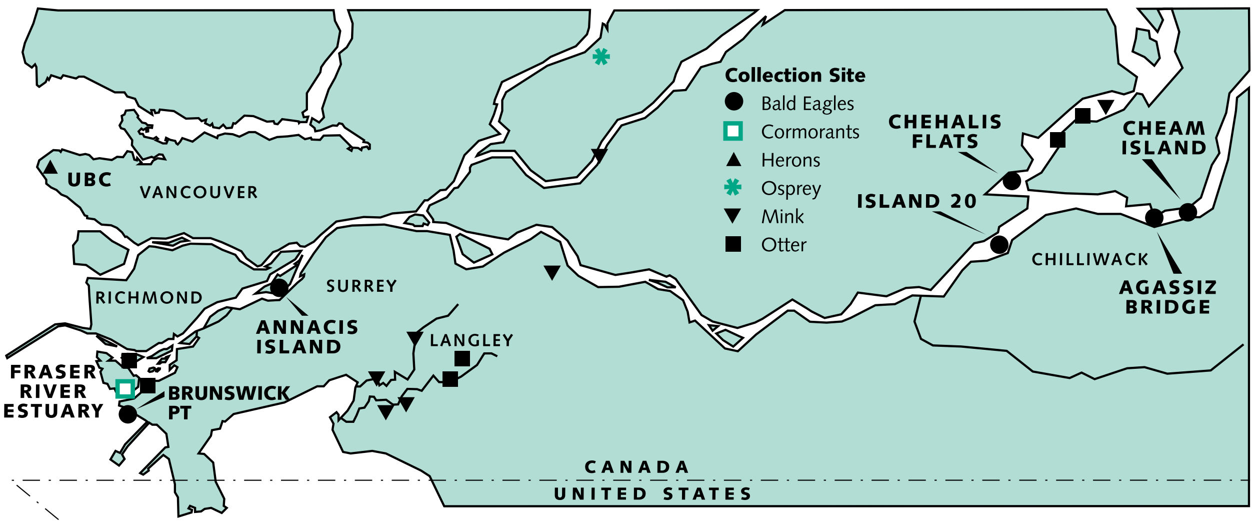

Figure 1. *Collection sites of selected indicator species in the Lower Fraser Valley, 1990–97.*

The two raptor species chosen for contaminant surveys occupy all major habitats in B.C. The bald eagle is a year-round resident along the coast, and also uses the upper reaches of the Fraser River for breeding. It has been useful as an indicator species in the lower basin, largely because of its scavenging habits, which include the selection of impaired (often contaminated) prey (Noble *et al.* 1993). The osprey breeds along the full length of the river, but migrates to southern foraging grounds, primarily in Latin America, for the winter. Osprey predominantly eat large, slow-moving fish (*e.g.* largescale sucker). As these fish feed along the stream bottom, and Fraser River sediments have substantial levels of persistent chlorinated compounds (Mah *et al.* 1989; Hatfield Consultants 1995; Brewer *et al.* 1998), osprey represent the top of an aquatic food chain that is potentially biomagnifying historical and ongoing inputs of contaminants. While osprey, cormorants and herons have direct, essentially exclusive links to aquatic food webs, the more opportunistic bald eagles prey on aquatic (fish), semi-aquatic (ducks), and terrestrial (birds, small mammals) species. This will increase the diversity of contaminants and contaminant sources to which the eagles are exposed. Bald eagle eggs and chicks collected from nests along the river in the Fraser Valley in 1990–91 (Fig. 1) were analyzed for contaminants. Dead or dying bald eagles were also collected from 1989 to 1994 by a variety of agencies, autopsied for cause of death, and assayed for contaminants in liver tissue (Elliott *et al.* 1996a). Osprey eggs were collected from nests near Kamloops, Quesnel and Prince George from 1991 through 1997 for contaminant analysis, while eggs collected in 1995 and '96 were hatched in captivity to study the impact of pulp mill contaminants on embryo development (Fig. 2).

The tree swallow was added to the monitoring regime only recently (1994) as the fifth bird indicator species. These swallows nest along the full length of the Fraser River, and feed heavily on insects over the

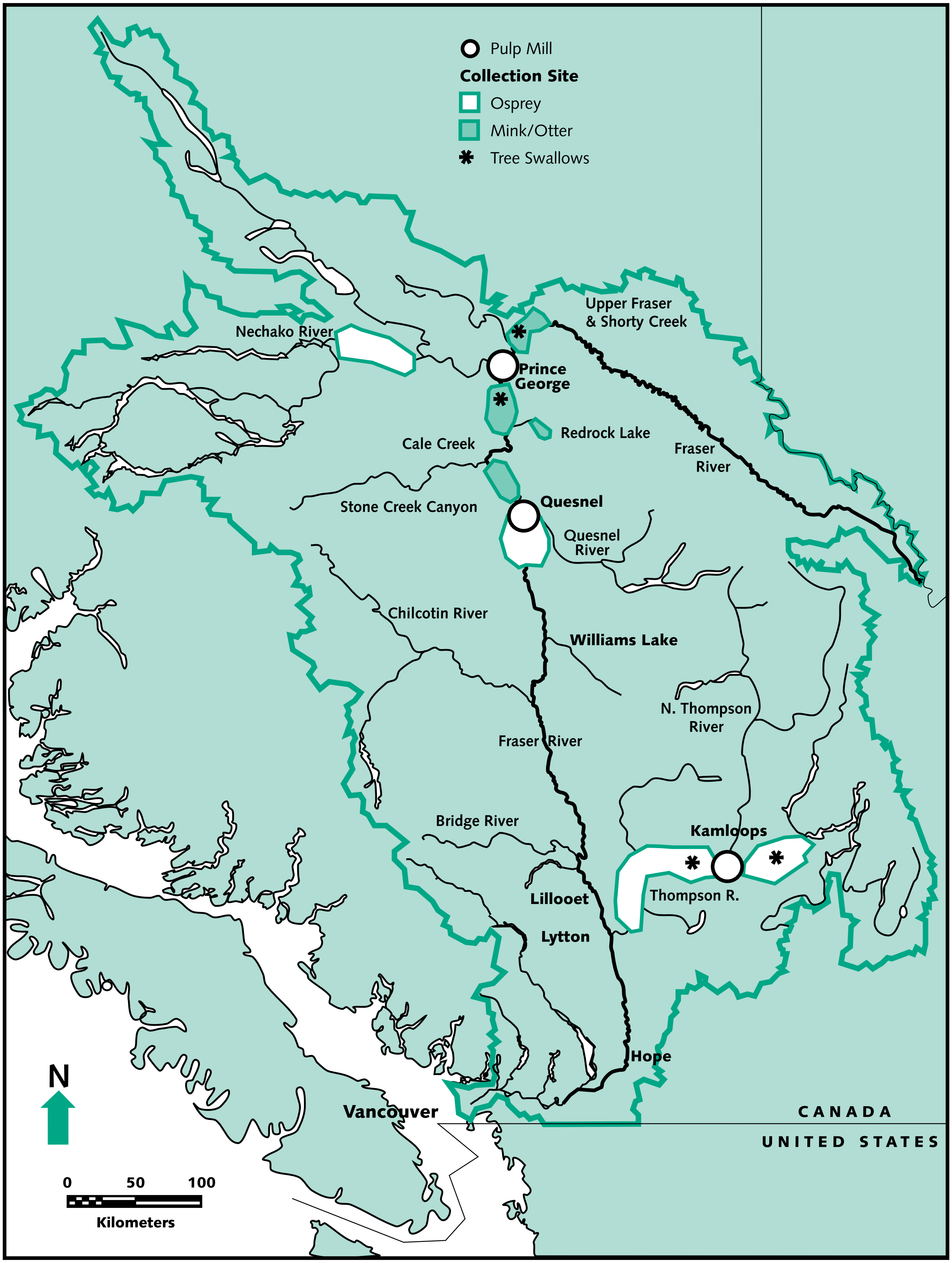

Figure 2. *Collection sites of selected indicator species in the upper reaches of the Fraser River Basin, 1990–97.*

river, especially during emergence events (of chironomids, for example). Because of their insectivorous nature, we hypothesized that they would be good indicators of the effects of persistent as well as less persistent contaminants on the river. Tree swallow nestlings were collected near pulp mills at Kamloops and Prince George (Fig. 2) from nest boxes placed specifically to attract birds for the study.

Finally, mink and river otter were chosen as mammalian indicator species. They have relatively small home ranges, without the migratory behaviour characteristic of some of the bird species. Also, they potentially prey on different species than birds. Mink are opportunistic predators (Wise *et al.* 1981). River otter likely have tighter ties to aquatic food webs, since the majority of their diet consists of fish with only minor seasonal variation (Melquist and Dronkert 1987). Mink and otter were initially only collected from the upper reaches of the river (Fig. 2), but surveys were expanded in 1995 to include collections in the Lower Fraser Valley (Fig. 1).

LEVELS AND EFFECTS OF CONTAMINANTS BY CHEMICAL GROUP

The contaminant families most frequently measured in wildlife from the basin are discussed in the following subsections. While most contaminant groups could be detected in each species, each indicator species is exposed to different levels in their food and have differing biochemical responses to this exposure. In addition, species and class differences in sensitivity varied the risk associated with some compounds. For instance, mercury residues were predominant in more piscivorous species, because fish are a major environmental source (via bioaccumulation pathways) of mercury (Fimreite *et al.* 1971). Lead and anticholinesterase pesticide residues, on the other hand, were predominant in bald eagles, because routes of uptake are typically through waterfowl foraging on agricultural and foreshore areas (see Wilson *et al.* 1999). However, the persistent chlorinated compounds, including polychlorinated-*para*-dibenzo-dioxins (PCDDs), polychlorinated dibenzofurans (PCDFs), polychlorinated biphenyls (PCBs) and organochlorine (OC) pesticides were detected in all species.

Organochlorine Pesticides

Contamination of B.C. wildlife with OC pesticides is a consequence of local persistence of historically applied compounds, long-range atmospheric transport of compounds from jurisdictions still allowing their use or having deposits from past use that are still volatile, and exposure on wintering grounds in Latin America. Only a handful of OCs are still registered for use in Canada (*e.g.* endosulfan, dicofol) and they are not very persistent nor readily bioaccumulated (Fry 1995). The most prevalent OC pesticide detected in wildlife from the Fraser River Basin was dichlorodiphenyldichloroethane (DDE), a more persistent breakdown product of dichlorodiphenyltrichloroethane (DDT).

The bald eagle and osprey showed higher egg concentrations of DDE than the other species (Table 1; see end of chapter); established toxicity levels were attained or surpassed in some nests. Wiemeyer *et al.* (1993) reported that 3.6 to 6.3 mg/kg wet weight DDE in bald eagle eggs would inhibit the production of young to approximately half normal values. Two of six eggs collected in 1990–91 (from Annacis Island and Chehalis Flats, downstream of Harrison and Agassiz) contained DDE within that range, and mean eggshell thickness was less than the pre-1947 average for the Pacific Northwest (Elliott *et al.* 1996b). Both findings suggest that DDE might still limit productivity of some eagle nests in the Lower Fraser Valley.

Survey data suggest that bald eagles currently have stable populations in the basin. Productivity calculations indicate that reproductive success is well above levels needed to sustain populations (Elliott and Norstrom 1998). However, in a five-year survey of nest success across coastal B.C., DDE plasma concentrations in chicks were weakly related to productivity declines (Elliott and Norstrom 1998), implying that DDE might influence provincial bald eagle populations in a limited fashion. In addition, back calculations based on current DDE levels in eagle and heron eggs and historical concentrations in heron eggs, suggest that breeding populations of bald eagles in the 1970s would have been affected by DDE levels around 25 mg/kg in eggs (Elliott *et al.* 1996b). Although DDE concentrations still fall into the range for effects in bald eagles, we suggest that any minor impacts at the population-level have been offset by other factors, including regulation of lead and specific pesticides causing mortalities in the Lower Fraser Valley (Elliott *et al.* 1992; Elliott *et al.* 1997), reduced persecution and more attention to protection of habitat, particularly nest trees (Elliott *et al.* 1996b).

Concentrations of DDE in osprey eggs showed high variability among individual eggs within study areas, although there were no significant differences in mean concentrations among study areas (Fig. 3). Twenty percent (10 of 51) of eggs contained DDE concentrations >4.2 mg/kg, the value associated with 15 per cent eggshell thinning in osprey (Wiemeyer *et al.* 1988), while eight per cent (4 of 51 eggs) contained >10 mg/kg which is associated with improper embryo development (Poole 1989). There was also consistent evidence of egg shell thinning in the eggs with elevated DDE levels (>4.2 mg/kg)(Machmer *et al;* in prep). Findings to date suggest that DDE might limit productivity of some osprey nesting at sites along the Fraser River. Reproductive success of osprey nesting along the Fraser River has been analyzed from the perspective of examining possible effects of exposure to pulp mill contaminants in populations breeding upstream and downstream of bleached kraft pulp mill sites (see section 2.4.3: Machmer *et al.,* in prep). These data do not differentiate eggs with varying DDE levels; consequently, potential effects of DDE exposure on osprey productivity are not currently available.

Most of the DDE acquired by ospreys during the breeding season likely entered the environment via long-range atmospheric transport as there was minimal historic use of its parent substance, DDT, in most of the Fraser Basin (Elliott *et al.*, in press). However, while Donald *et al.* (1993) and Macdonald *et al.* (1999) found that long-range atmospheric transport and deposition of pesticides could result in relatively high concentrations in trout and burbot in remote alpine headwater lakes, the latter study found even higher levels in lakes near agriculture (*e.g.* Nicola Lake). In agricultural areas where DDT was historically used in large quantities, the primary source of DDE for osprey is likely the ongoing release of DDT compounds from soil. This theory is supported by elevated DDE levels measured in osprey eggs collected near Pitt River, which is adjacent to an area of intensive horticultural activity in the Lower Fraser Valley. The most probable source of high DDE levels in osprey from the upper Fraser is exposure on their wintering grounds. Osprey nesting in the Fraser Basin likely migrate south to winter along the west coast of

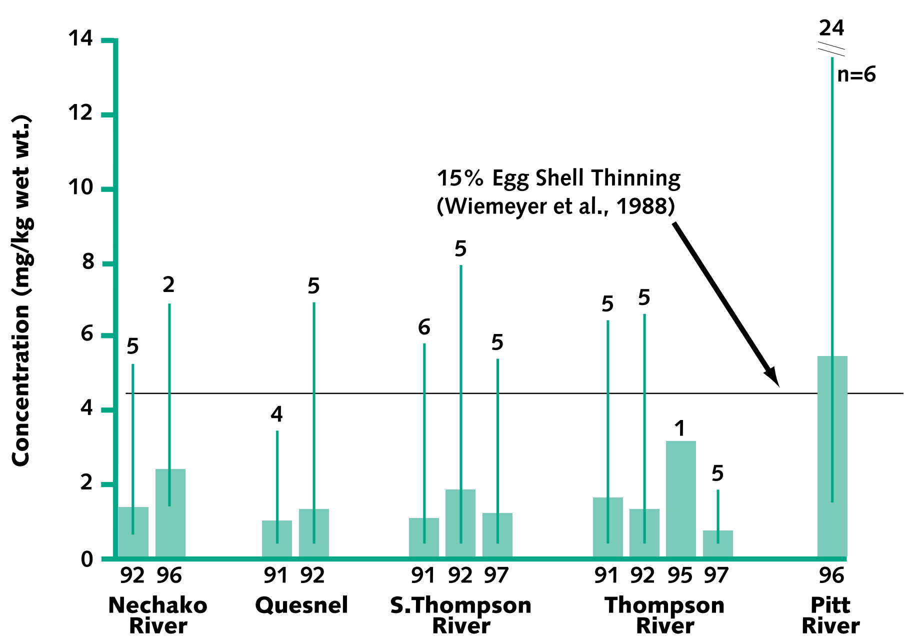

Figure 3. *DDE in osprey eggs collected in the Fraser River Basin, 1991–97. Values expressed as geometric means with 95% confidence interval error bars (mg/kg wet wt.). Number above bar graph represents the number of eggs analyzed per site per year.*

Mexico and other Central American countries where areas of intensive agriculture are located next to common osprey roosting sites. Although the DDE:DDT ratios are extremely variable, there is no indication of recent DDT input. Even ospreys with DDE concentrations >10 mg/kg had relatively high DDE:DDT ratios, suggesting that DDE exposure on wintering grounds is also from historic rather than recent DDT usage. The high degree of individual variability within sites also may be caused by differences in diet or related food chain factors (Elliott *et al.*, in press).

Other indicator species showed lower levels of DDE, and likely were not affected by measured concentrations (Table 1; see end of chapter). Great blue heron surveys indicate that residual values have been declining from peaks seen in the late 1970s (Fig. 4). Although past agricultural use probably has some influence on body burdens in wildlife from the Lower Fraser Valley and in the Nicola Lake area (Macdonald *et al.* 1999), it appears that atmospheric transport from regions where DDT is still used (*e.g.* Asia) might be responsible for the widespread occurrence of DDE even in areas without agriculture. Elliott *et al.* (1989a) found elevated DDE concentrations in eggs of seabirds along the B.C. coast. DDE plasma concentrations from bald eagle chicks collected on the west coasts of Vancouver and Langara islands were also elevated (Elliott and Norstrom 1998). These trends suggest a substantial DDE input from atmospheric sources as has been noted in the assessment of organochlorine accumulation in selected headwater lakes in the basin (Macdonald *et al.* 1999).

A suite of other OC pesticides was consistently detected in wildlife at low levels (Table 1). Chlordanes (trans- and cis-nonachlors, transchlordane, oxychlordane) were commonly found in eggs and adult livers, along with dieldrin, heptachlor epoxide, chlorobenzenes, hexachlorocyclohexane and mirex. Organochlorine residues detected in mink and river otter were not notably different in the agricultural setting of the Lower Fraser Valley from other collection sites along the river. This observation, along with the presence of compounds like mirex, suggests that contaminant sources are primarily atmospheric. None of the compounds in this suite of pesticides poses a threat to wildlife health at concentrations detected.

Polychlorinated Biphenyls (PCBs)

PCBs is the generic term for a family of 209 congeners that contain a variable number of chlorine atoms on a biphenyl molecule. Their major use was in electrical transformers. While all uses in Canada were significantly curtailed in 1977, up to 40% of the PCBs imported are not accounted for and have been dispersed in the environment (Moore and Walker 1991). Unfortunately, chemicals in this family are relatively persistent and semi-volatile. A major route of re-distribution of PCBs existing in soils, sediments and waste

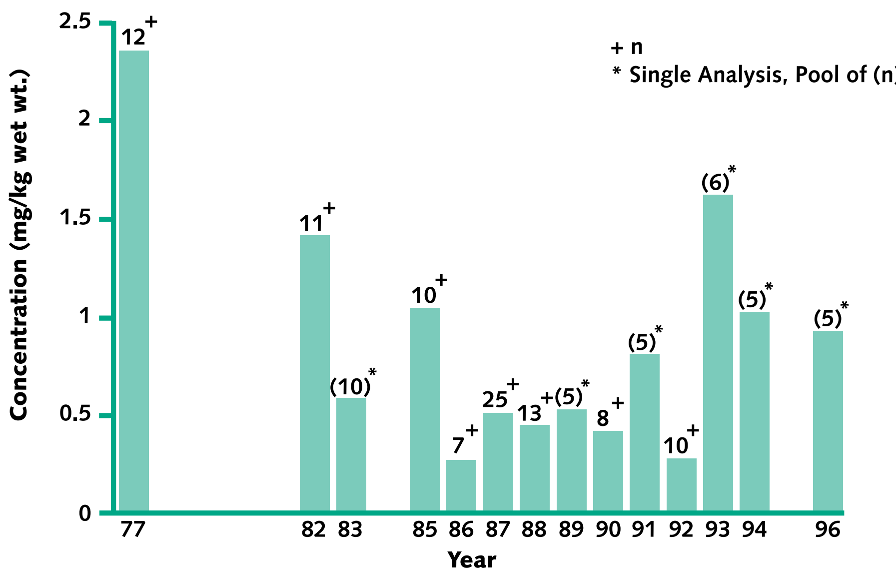

Figure 4. DDE (geometric means, mg/kg wet wt.) in great blue heron eggs collected from the colony at the University of British Columbia Endowment Lands in the Lower Fraser Valley, 1977–96. Number above bar graph represents the number of eggs analyzed per year.

dumps occurs via the atmosphere and so deposition can contain PCBs from sources nearby or from sites on other continents in the Northern Hemisphere.

The PCB family of contaminants poses more of a risk to mustelids than birds. Mink reproductive success is extremely affected by exposure to small amounts of PCBs (Aulerich and Ringer 1970; Heaton *et al.* 1995; Leonards *et al.* 1994; 1995), and the threatened or extinct status of otter in most European countries has been linked to PCB contamination (Mason 1989; Smit *et al.* 1994). Mink and river otter from the Fraser River Basin showed low levels of total PCBs (Elliott *et al.* 1998a) compared to those seen in the Columbia River Basin (Table 2; see end of chapter). It is difficult to categorically state whether these levels in the Fraser Basin have an effect on these species. A negative correlation found between juvenile male mink baculum (penis bone) length and liver Aroclor 1260 (a subset of PCBs) levels found using the Columbia and Fraser data set, suggests there could be impacts on reproductive success in some locations (Harding *et al.* 1999; Henney *et al.,* in prep.). As the Fraser sample data formed the "normal" range for baculum length and the low range for PCBs, we cannot assess its potential impact in mink at these lower levels. It is interesting to note, however, that the non-ortho (77, 126, 169) PCB congeners contributed consistently and substantially to toxic body burdens (as measured with toxic equivalent units [TEQs]) in these mammals (see Fig. 13 in PCDD/F section, *Mink and river otter*).

Long-term monitoring of heron colonies in the Fraser estuary suggests that egg concentrations of PCBs declined precipitously a few years after strict regulations on use were imposed (1977), and fluctuated at lower levels from the early 1980s until 1994. Eggs collected in 1996 had the highest levels measured since 1983, however, it is too early to determine if this is an increasing trend (Fig. 5). PCBs in great blue heron eggs in 1977 and 1982 might have been sufficient to affect embryonic development, based on a lowest-observed-effect level (LOEL) for black-crowned night herons of 4.5 mg/kg (Hoffman *et al.* 1986), but more recently measured values were below toxicity thresholds. Similarly, recent total PCB residues in cormorants, raptors and tree swallows from the Fraser River Basin were all low and toxicologically inconsequential (Table 2). The early high concentrations in herons, however, imply that raptorial species in the basin probably also had toxicologically relevant residues during that period (1977–1982). Toxicological studies on osprey embryos from the upper Fraser River, undertaken in the 1995 and 1996 breeding seasons, showed that biochemical responses (hepatic EROD induction, elevated plasma vitamin A) were positively correlated with total PCB and PCB 126. No similar relationship was observed with hatching success. The implications for individual and population health are not known.

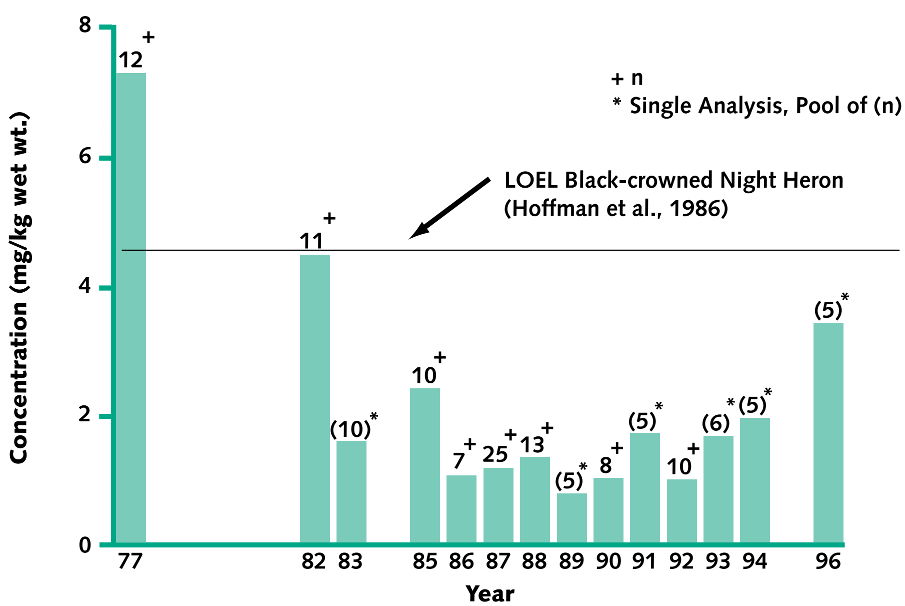

Figure 5. Total PCBs (geometric means, mg/kg wet wt.) in great blue heron eggs collected from the colony at the University of British Columbia, 1977–96. Number above bar graph represents the number of eggs analyzed per site per year.

LOEL = lowest-observed-effect level.

Even though total PCB residues in wildlife have dropped since monitoring began, the most toxic non-ortho congeners still contribute substantially to TEQs (see PCDD/F section). In fact, the relative contribution of specific PCB congeners to TEQs has increased over time for some species, reflecting the decreased presence of the more highly toxic PCDD/F congeners in the Fraser River. Thus, the toxic stress contributed by PCBs to wildlife will extend further into the future than the dioxins and furans released, in the past, from pulp mills.

Mercury

Mercury residues can persist from past industrial and agricultural releases. However, recent concerns focus on mobilization of mercury from man-made reservoirs (Hughes *et al.* 1997) and deposition of atmospherically transported mercury from coal-fired utilities and municipal waste incineration (Swain *et al.* 1992).

Higher mercury levels observed in bald eagles compared to great blue herons from the Fraser estuary may reflect dietary differences. A greater proportion of the eagles' diet is larger, older fish which may have accumulated more mercury than the year-old fish preferred by herons. Osprey from inland waterways have less mercury contamination than coastal eagles (Fig. 6) even though mercury concentrations are usually higher in freshwater versus marine environments (Elliott *et al.* 1989b) because eagles, in addition to consuming fish, also feed on other fish-eating birds thereby bioaccumulating a larger amount from the local food chain. Mercury concentrations in osprey eggs from the upper Fraser River were quite uniform among sites and were comparable to those reported in the literature for ospreys nesting on naturally formed lakes and rivers (Elliott *et al.,* in press). Based on the levels in osprey eggs from the Thompson and eagle eggs from coastal B.C. (Elliott *et al.,* in press), past mercury discharges from pulp mills have not had a lasting impact on local food chains.

Liver tissue in mink and river otter had substantially higher mercury levels than eggs from the fish-eating birds (Fig. 6). Mink and river otter collected from the Lower Fraser Valley showed greater average mercury values than those collected near Prince George, but the variability amongst individuals was very high. Two juvenile male mink from the upper basin were actually the most contaminated, with 4.2 and 4.3 mg/kg wet weight mercury in liver tissue (Harding *et al.* 1998). Although such concentrations would be insufficient to produce mercury poisoning (>24 mg/kg; Wren 1991), they might be within the range of chronic toxicity. Wren *et al.* (1987) discovered that a combination of 0.5 mg/kg methylmercury

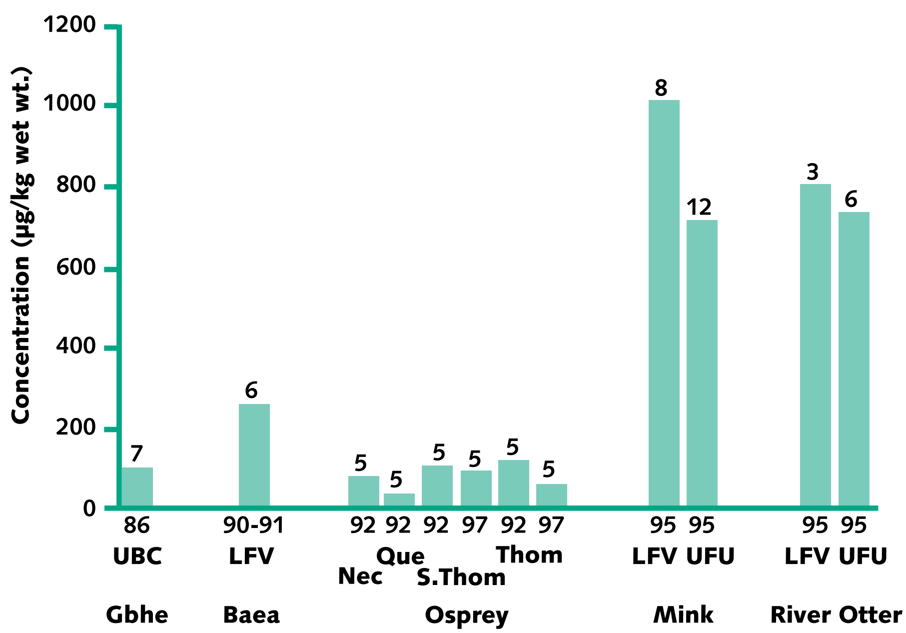

Figure 6. *Mercury (geometric means, µg/kg wet wt.) in eggs of great blue heron (gbhe), bald eagle (baea) and osprey, and in liver tissue of mink and otter from the Fraser River Basin. Number above bar graph represents the number of individuals analyzed.*

UBC=University of British Columbia in the Lower Fraser Valley; LFV=Lower Fraser Valley; nec=Nechako River; Que=Quesnel; S.Thom=S. Thompson River; Thom = Thompson River; UFV=upper Fraser Valley and 0.5 mg/kg Aroclor 1254 reduced mink kit survival even when neither compound, tested independently, elicited a measurable response. Since mink and river otter from this region were also typically contaminated with low levels of PCBs, the combined presence of mercury and PCBs could pose some risk to mustelids along the Fraser River. An examination of reproductive success and the presence of these contaminants over a wider area would be necessary for further confirmation.

Polychlorinated Dibenzo-p-dioxins (PCDDs), Dibenzofurans (PCDFs) and Toxic Equivalent Units (TEQs)

The major incentive for initiating residue studies of polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs) contamination in B.C. wildlife was to evaluate impacts of the persistent pollutants from the pulp and paper industry on wildlife at the top of the aquatic food chain. As monitoring progressed over the years, other PCDD/F sources, such as sites of past chlorophenolic-based wood preserving chemical production, were also identified as probable contributors to contaminants in wildlife. Much of the monitoring results are already published (Elliott *et al.* 1989a, b; 1996a, b, c; 1998a, b; in press; Elliott and Norstrom 1998; Sanderson *et al.* 1994a, b; Wilson *et al.* 1996).

Concentrations of major PCDD and PCDF congeners detected in key indicator species are listed in Tables 3 and 4 (see end of chapter), respectively. The major findings (residual patterns, associated toxicity) are summarized by species in the following subsections. The toxic equivalent (TEQ) methodology (using WHO Toxic Equivalent Factors [TEFs]; Van den Berg *et al.*, submitted) was used to evaluate the toxicological relevance of body burdens of PCDDs, PCDFs and PCBs.

Great blue herons and double-crested cormorants

A dramatic decline in egg concentrations of PCDDs and PCDFs between 1982 and 1989 was observed in the great blue heron colony near the University of British Columbia (UBC). The most toxic congener, 2,3,7,8-TCDD, however, peaked in 1989 and then dropped to near or below the detection limit by 1992. The timing of these declines (post-1990) suggests that implementation of chlorine dioxide substitution at upstream pulp mills was effective in reducing the discharge and subsequent concentrations of the tetrachlorinated congeners in prey and that body burdens in eagles responded quickly to these declines. The shorter data set for double-crested cormorants followed the same pattern from 1985, but as no eggs were collected in 1989, we cannot confirm the peak in 2,3,7,8-TCDD found in great blue heron that year.

Concentrations of pentachloro-dibenzo-*p*-dioxins (PnCDDs) and hexachloro-dibenzo-*p*-dioxins (HxCDDs) also declined in both species over time. Heron egg 1,2,3,6,7,8-HxCDD values dropped from hundreds of ng/kg in the early 1980s to an average of 23 ng/kg for 1991–94. The production of HxCDDs has been associated with the synthesis of pentachlorophenol (PCP) (Van den Berg *et al.* 1987) and use of chlorophenolic-treated wood in pulp mills. Reductions in HxCDD burdens in wildlife attest to the effectiveness of PCP use restrictions implemented in 1989.

Investigations of biological effects associated with contaminant burdens in B.C. heron eggs (Sanderson *et al.* 1994a) estimated that herons nesting at the UBC colony in 1988 were negatively impacted by PCDD/F contamination, but that reductions in body burdens over the next several years improved chick health. TEQs for great blue heron eggs were very high in the early 1980s (Fig. 7), TEQ contributions from PCDDs far exceeding those from PCBs. TEQs declined dramatically in 1983 but peaked again in 1989 when 2,3,7,8-TCDD releases in pulp mill effluents were at their maximum. Post-1990, TEQs were substantially reduced and PCBs dominated the total toxic burden in both herons and cormorants.

Although attempts to conduct toxicological studies on double crested cormorants, similar to those conducted for great blue heron, were unsuccessful in the cormorant colony from the Fraser River Estuary (Sanderson *et al.* 1994b), total TEQs in these cormorants (approximately 150 ng/kg in the ’80s and 50 ng/kg in the ’90s; Fig. 8) were always lower than the TEQs correlated with effects in a Lake Ontario cormorant colony. This implies that contamination from the Strait of Georgia, where the cormorants prefer to hunt, was not sufficient to affect their breeding success. Limited surveys of nest activity suggest that both herons (Gebauer 1995) and cormorants (Vermeer *et al.* 1989) have increased their use of the estuary for breeding.

Bald eagles

Six individual bald eagle eggs collected from the Lower Fraser Valley in 1990–91 contained higher total residues than heron and cormorant eggs collected in the same years. Although the proportions of 2,3,7,8-TCDD to 2,3,7,8-TCDF were comparable to those in residues found in fish tissues collected during that time period (Raymond *et al.* 1999), the levels of TCDF were much greater than those detected in other estuarine bird species. Together, the two congeners, most reflective of pulp mill contaminant sources, contributed 40–80 per cent of the total PCDD/F burden. In yolk sacs of chicks collected in 1992, the tetra-congeners contributed 11–25 per cent of the total PCDD/Fs. Investigations in other bird species suggest that 2,3,7,8-TCDF is eliminated quickly from the body (Norstrom *et al.* 1976); therefore, the elevated concentrations detected in bald eagle eggs and chicks were probably the result of recent exposure to pulp mill contaminated prey and direct deposition of contaminated lipids into the yolk (Elliott *et al.* 1996d).

A reasonable database on exposure and biological effects of PCDDs and PCDFs in bald eagles has been produced

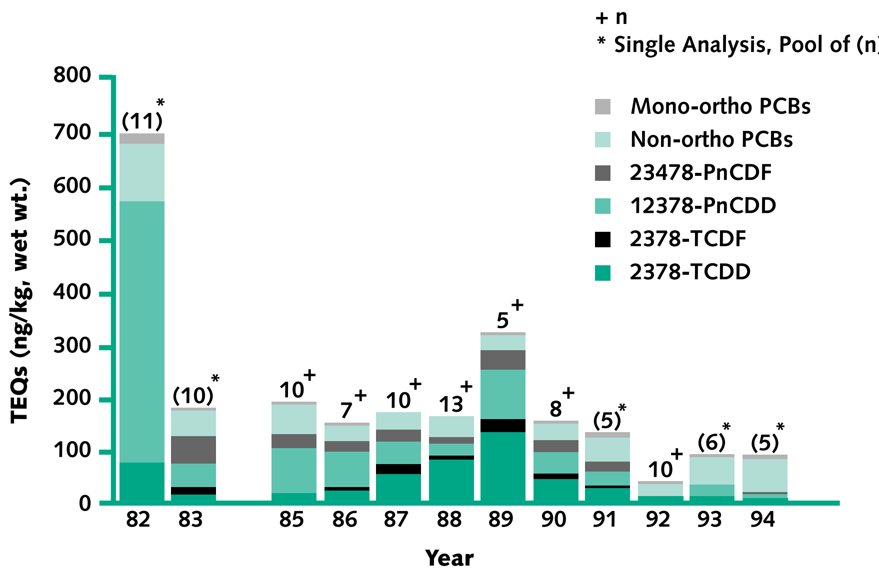

Figure 7. Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in great blue heron eggs collected from the University of British Columbia colony in the Lower Fraser Valley, 1982–1994. Calculations used WHO-TEFs. Number above bar graph represents the number of eggs analyzed per year.

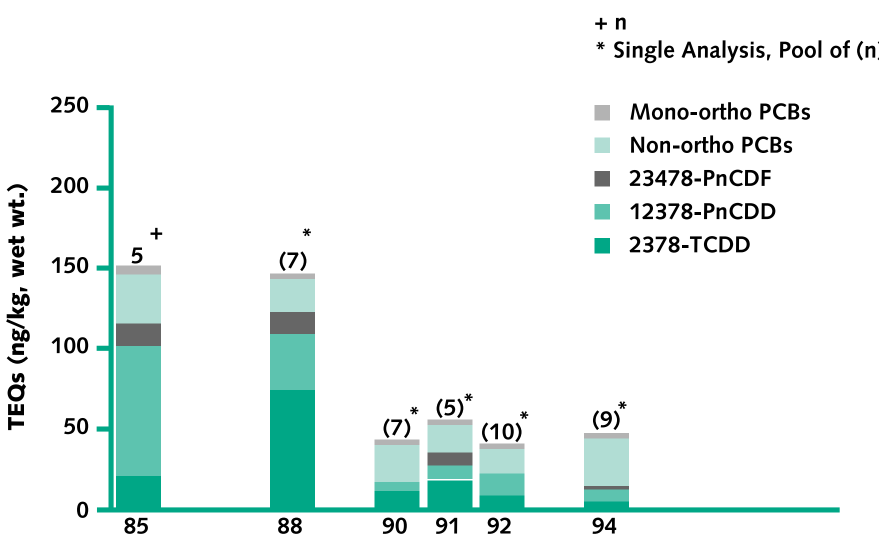

Figure 8. Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in double-crested cormorants eggs from the Fraser River Estuary, 1985–1994. Calculations used WHO-TEFs. Number above bar graph represents the number of eggs analyzed per year.

for B.C. (Elliott *et al.* 1996a, d; Elliott and Norstrom 1998). The no-observed-effect level (NOEL; 100 ng/kg TEQs) was exceeded in five of six eggs collected from the Lower Fraser Valley, while a lowest-observed-effect level (LOEL; 210 ng/kg TEQs) was exceeded in two eggs (Fig. 9). As suggested above, 2,3,7,8-TCDD was a major contributor to total TEQs, indicating that pulp mill effluents were still a dominant source of toxicity for this species in the early 1990s—probably via contaminated prey (*e.g.* fish). In nestling blood samples collected in 1993–94, the pattern of contamination was more skewed toward PCBs instead of TCDD (Elliott *et al.* 1996d; Elliott and Norstrom 1998). The PCDD/F burdens did not appear to affect reproductive success in Fraser Valley populations between 1990 and 1994. Most recent surveys (1992–97) estimated that average productivity of bald eagles in the lower basin was higher (1.1–1.2 young/nest) than the minimum 0.7 young/nest needed to maintain a population (Elliott and Norstrom 1998).

Osprey

The osprey, along with three other species (tree swallows, mink, river otter) collected from the upper reaches of the basin, were predominantly contaminated with octachloro-dibenzo-*p*-dioxin (OCDD) (Table 4). In particular, OCDD concentrations were high (374–472 ng/kg) in South Thompson River osprey, and still high, but markedly lower (91–94 ng/kg) in the Thompson River group, except for the single osprey egg sampled in 1995. OCDDs also contributed 60–85 per cent of the PCDD/F burden in Quesnel, Nechako and Pitt River eggs, although the absolute values were lower.

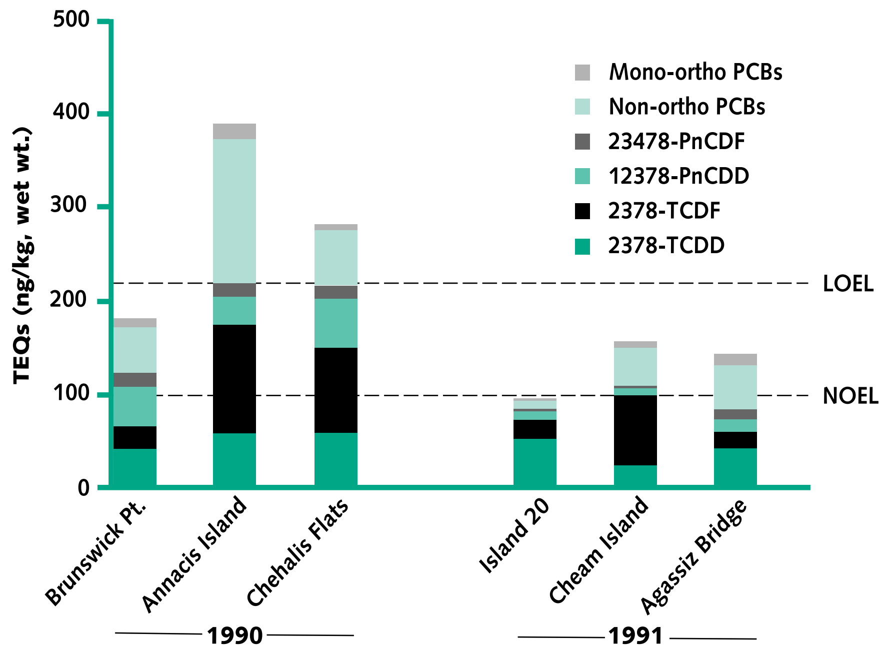

Figure 9. *Toxic equivalent units(TEQs, ng/kg wet wt) for major PCDDs, PCDFs and PCBs in individual bald eagle eggs from the Lower Fraser Valley, 1990–1991. Calculations used WHO-TEFs.*

NOEL: no-observed-effect level; LOEL: lowest-observed-effect level

Variability in values was extremely high, even between neighbouring nests, perhaps suggesting that minor differences in diet or feeding areas (*e.g.* mainstream versus tributaries or neighbouring lakes) have major repercussions for contaminant loads (Elliott *et al.* 1998b). A significant correlation between pentachlorophenol and both OCDD and 1,2,3,4,6,7,8-heptachloro-dibenzo-*p*-dioxin (HpCDD) concentrations in Thompson River eggs (OCDD: $r=0.870$, $p<0.01$; HpCDD: $r=0.891$, $p<0.01$; Fig. 10) strongly implicates a pentachlorophenol source for this contamination. The hypothesis is further supported by the pattern of trace HpCDFs in osprey eggs, which is characteristic of a pentachlorophenol source (Elliott *et al.* 1998b). Numerous primary timber processing facilities—major users of pentachlorophenols—are also located along the South Thompson and Thompson Rivers around the Kamloops area where the highest amount of elevated PCDDs were measured in osprey eggs.

There was also a strong upstream/downstream trend in the lower chlorinated PCDD/Fs in the Thompson River osprey eggs. Concentrations of 2,3,7,8-TCDD and 2,3,7,8-TCDF were greater in eggs collected downstream compared to upstream nests near the Kamloops pulp mill in 1991. As a result of changes in bleaching technology (chlorine dioxide substitution) at the mill, by 1997, concentrations of 2,3,7,8-TCDD, and -TCDF were significantly lower than in previous years in nests sampled downstream. Osprey along this stretch of the Thompson eat primarily mountain whitefish and largescale sucker, two species shown to carry large burdens of TCDD and TCDF during a 1988 fish survey (Mah *et al.* 1989). Between 1988 and 1992, concentrations of 2,3,7,8-TCDD in mountain whitefish muscle samples collected downstream of the Kamloops mill decreased from 61 to 1.7 ng/kg, while 2,3,7,8-TCDF levels dropped from 390 to 8.2 ng/kg (Hatfield Consultants Ltd. 1995). More recent samples in 1994 show levels have stopped declining with 2, 3, 7, 8-TCDD and -TCDF concentrations of 1.1 and 10.6 ng/kg, respectively (Raymond *et al.* 1999). The large-scale sucker is a bottom-feeder whose close association with sediments likely results in the bioaccumulation of PCDD/Fs deposited in sediments from mill effluent. In addition, mountain whitefish eat benthic invertebrates which in 1993 exhibited elevated tissue levels of OCDD and 2,3,7,8-TCDF downstream at Agassiz (Richardson and Levings 1996).

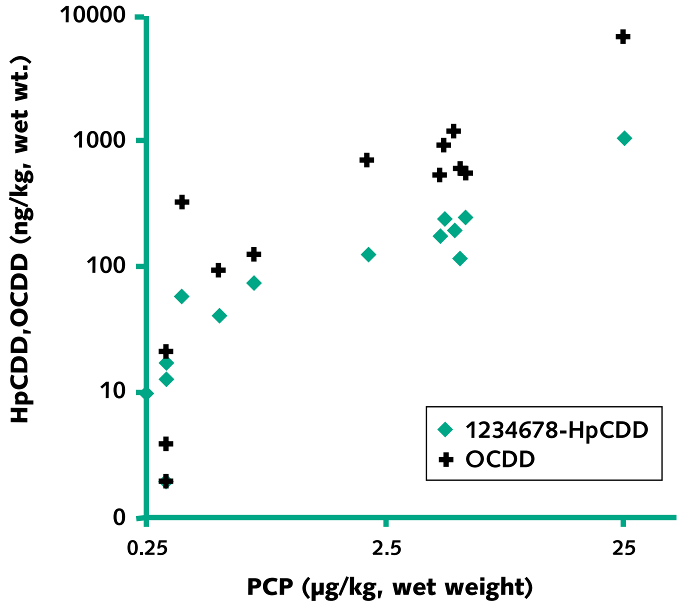

Figure 10. Relationship between concentration of PCP (pentachlorophenol) and 1,2,3,4,6,7,8-HpCDD and OCDD in osprey eggs from the Fraser River Basin of British Columbia, 1991 (OCDD, r=0.870, p<0.01; HpCDD, r=0.891, p<0.01).

Embryonic toxicological studies were undertaken in 1995 and 1996 breeding seasons which involved osprey chicks from upstream and downstream of the pulp mill at Kamloops along the South Thompson and Thompson rivers and in the Nechako and Pitt rivers (Elliott *et al.*, in prep). No differences were observed in laboratory hatching success between these areas. Hepatic EROD activity was greater (2.8-fold) in chicks collected downstream of Kamloops compared to the Nechako site, and correlated positively with total PCBs and PCB 126, but not 2,3,7,8-TCDD. A hepatic protein (cytochrome P4501A-like) was detected in all samples and correlated with EROD activity. Preliminary results suggest osprey chicks in the upper Fraser River exhibited toxicological response to chlorinated contaminant exposure. While no differences were observed in laboratory hatching success between the upstream and downstream study areas, eggs which did not hatch had, on average, a two-fold higher level of DDE compared to eggs which did hatch, regardless of collection area. The implication for individual and population health are not known.

The osprey eggs accumulated lower TEQs (Fig. 11) than the herons and bald eagles, principally because the TEF for OCDD is relatively small. Highest TEQs were calculated for osprey eggs collected downstream of the pulp mill at Kamloops, and, not surprisingly, those were dominated by contributions from 2,3,7,8-TCDD. Preliminary analyses showed that osprey clutch size, estimated in 1991, '92, '94 and '97 surveys, was always greater in upstream versus downstream nests on the Thompson River. Similarly, fledgling success, estimated in 1992, '94 and '97, was always greater in upstream nests; but only fledgling success in 1992 was significantly greater in upstream nests (Machmer *et al.,* in prep). The low productivity of osprey downstream of pulp mills at Kamloops on the Thompson River was not caused by poor embryonic survival, as shown in the embryonic toxicological study where there was no difference in hatching success between areas. However, the study did show a positive correlation between biological effects (hepatic EROD induction, elevated plasma vitamin A) in the chicks and elevated PCB concentrations in the yolk sac suggesting that contaminants may influence reproductive success by adversely affecting chick survival.

Tree swallows

Tree swallow nestlings were collected in 1994 from nest boxes placed upstream and downstream of the pulp mills at Kamloops on the South Thompson and Thompson rivers and Prince George on the upper Fraser River. Analyses of pooled whole body homogenates detected elevated PCDDs and PCDFs in swallows downstream of Prince George mills (Table 4). The presence of highly chlorinated congeners, as well as the detection of pentachlorophenol (Table 5), is consistent with patterns shown in all other monitored wildlife species from the upper basin, and supports the suggestion that benthic prey species are accumulating highly chlorinated PCDDs from sediments exposed to runoff from areas of high chlorophenol use (Elliott *et al.*, 1998b). Kamloops samples and those upstream of Prince George showed negligible contamination of PCDD/Fs, yet all homogenates had detectable levels of 2,3,7,8-TCDF, suggesting that nestlings from both upstream and downstream of the mills were being fed contaminated insects.

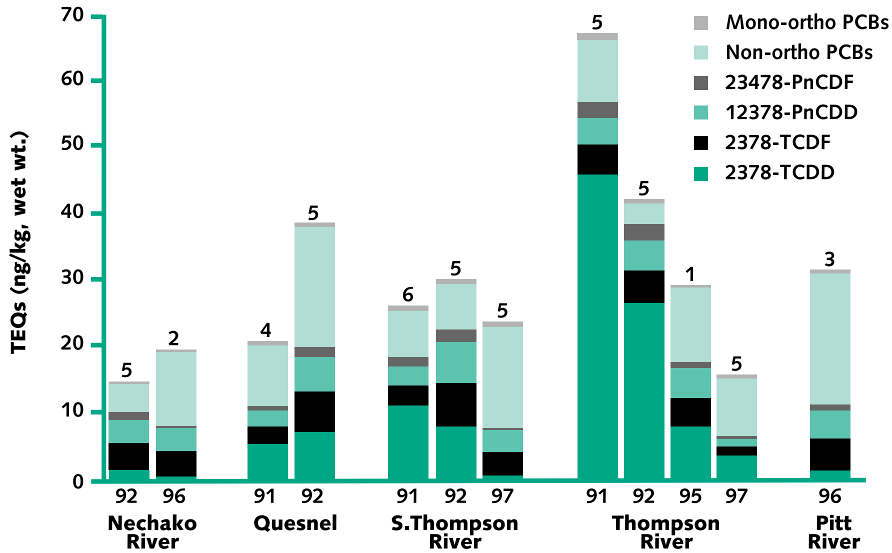

Figure 11. Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in osprey eggs from the Fraser River Basin, 1991–97. Calculations used WHO-TEFs. Numbers above bars represent the number of eggs analyzed.

Tree swallow whole body homogenates had higher ratios of TCDF:TCDD compared to levels detected in eggs of most other birds studied (Table 4). TCDF is cleared rapidly by birds (Norstrom *et al.* 1976; Braune and Norstrom 1989), thus, the detection of this compound in birds collected near bleached kraft pulp mills results from dietary exposure to high concentrations in prey. TCDF may be directly deposited to the yolk from dietary lipids during the period of rapid yolk deposition in ovarian follicles (Elliott *et al.* 1998b). The comparatively high levels of TCDF detected in the tree swallows may also reflect the swallows' position in the food chain. Tree swallows consume emergent insects which acquire high burdens of TCDF from sediments. Fairchild *et al.* (1992) reported that emergent insects exported up to 2.1 per cent of TCDF burden in sediment one year after chemical addition (Elliott *et al.* 1998b). Raptorial species, such as ospreys, forage higher in the food chain, consuming prey which may have already degraded some of their TCDF burden. Alternatively, the different concentrations of TCDF detected in the various indicator species

Table 5. Relationship between concentrations of highly chlorinated PCDDs (OCDD and 1234678-HpCDD) and Pentachlorophenol in tree swallow nestlings (whole body homogenates) collected from upstream and downstream of pulp mills in British Columbia, 1994.

SITE[a]	N[b]	OCDD & 1234678-HpCDD (mg/kg wet wt)	PENTACHLOROPHENOL (µg/kg wet wt)
S. Thompson River (u/s Kamloops)	3p	4.89	NA[c]
Thompson River (d/s Kamloops)	11p	4.3	<1
upper Fraser River (u/s Pr. George)	18p	3.78	<1
upper Fraser River (d/s Pr. George)	14p	70.45	4

[a] *u/s - upstream; d/s - downstream*
[b] *number of nestlings pooled in single analysis*
[c] *NA - not analyzed*

may reflect the relative metabolic capability (inducible levels of a hepatic cytochrome P450 1A-like protein and associated ethoxyresorufin-o-deethylase activity) of the species to break down these contaminants. Osprey chicks had an enhanced ability to metabolize chlorinated hydrocarbons (Elliott and Trudeau, unpublished data) compared to bald eagle chicks (Elliott *et al.* 1996d) even though eagles were exposed to higher concentrations of contaminants (Elliott *et al.* 1998b).

The dominant PCDD/Fs in the tree swallows were the higher chlorinated congeners which, because of their low TEF value, did not contribute substantially to the overall TEQ values (Fig. 12). Non-ortho PCBs were a major contributor to tree swallow TEQ values, suggesting PCBs may be toxicologically more relevant than PCDD/Fs. Productivity data have not yet been analyzed, so the biological relevance of chlorinated contaminant exposure in these swallow populations cannot be assessed.

Mink and river otter

The difference in magnitude of contamination was substantial amongst otter groups collected from the Lower Fraser Valley in 1995 (Table 3) and the upper basin in 1991 and 1995 (Table 4). Total concentrations of PCDD/Fs, particularly OCCD, were at least 20-fold greater in two river otter composites from the Lower Fraser Valley compared to the otter composite from the upper Fraser River. Liver tissue from mink collected near Prince George and Quesnel had low levels of PCDD/Fs; only one individual (Cale Crk #1, Table 4), with elevated concentrations of 1,2,3,4,6,7,8-HpCDD (123 ng/kg) and OCDD (186 ng/kg), was roughly as contaminated with the highly chlorinated congeners as the Fraser Valley river otters. The dominant PCDD/F congener, as in osprey eggs and tree swallow nestlings, was OCDD, which suggests that the contaminant source for these mammals is similar and likely related to pentachlorophenol use. Otter from the Lower Fraser Valley had higher concentrations of higher chlorinated contaminants than otter from upper reaches of the Fraser River, which may reflect the greater prevalence of industrial activities along tributaries in the Lower Fraser Valley.

Mink and river otter from the Fraser River had low TEQs (Fig. 13). Mink TEQs were all low and dominated by PCBs, except for the one individual discussed above. All values were considerably lower than the 160 ng/kg TEQs estimated as a critical body residue for mink reproduction by Leonards *et al.* (1995). River otter TEQs were low in samples collected from the upper Fraser River; one composite collected in the Lower Fraser Valley was comparatively more contaminated, especially with 2,3,4,7,8-PnCDF and 1,2,3,6,7,8-HxCDD.

Comparison of PCDD/F burdens and patterns with other regions

Elliott *et al.* (1996c) compared PCDD and PCDF residues in North American wildlife, and found that B.C.

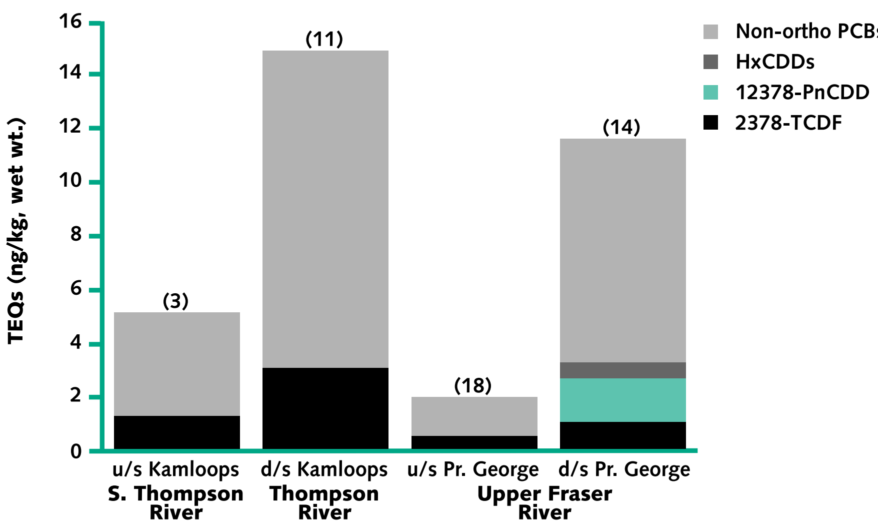

Figure 12. Toxic equivalent units(TEQs) for major PCDDs, PCDFs and PCBs in tree swallows collected upstream and downstream of pulp mills at Kamloops on the Thompson River system and Prince George on the Fraser River, 1994. Calculations used WHO-TEFs. Numbers above bars represent the number of whole body homogenates analyzed as a single pool.

populations were often among the most contaminated of monitored groups. However, heron, cormorant and bald eagle eggs collected in the Fraser River Basin were less contaminated than those in other areas of B.C., for example, colonies near coastal pulp mills such as Crofton on Vancouver Island, which was the most contaminated of all monitored colonies.

Osprey, mink and river otter collected from the Fraser River Basin were generally less contaminated than Columbia River populations (Elliott *et al.* 1998 a, b; Harding *et al.* 1998). However, OCDD concentrations seen in wildlife from the upper Fraser River Basin were usually higher than those from Columbia River samples (except for two river otter and mink pools from the lower Columbia). Although mink and otter were collected proximate to pulp mills on both rivers, there was no pattern suggestive of pulp mill influences in the Fraser River animals, compared to the strong mill effluent patterns (higher levels downstream compared to upstream of mill site) in animals near Castlegar on the Columbia River as well as mink from Quebec (Champoux 1996). Fraser River mustelids had lower concentrations of PCDD and PCDFs compared to Quebec mink.

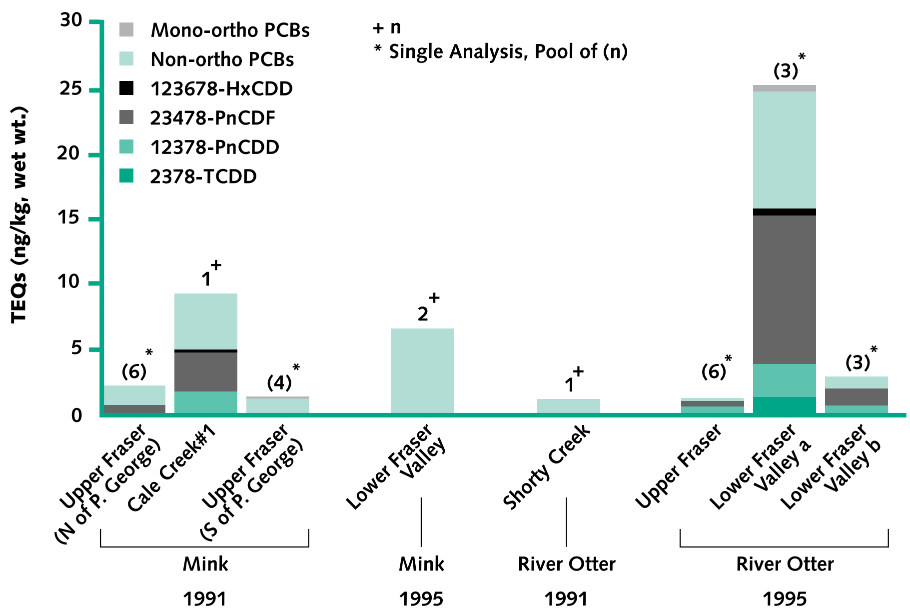

Figure 13. *Toxic equivalent units (TEQs) for major PCDDs, PCDFs and PCBs in liver tissue of mink and river otter from the Fraser River Basin, 1991 and 1995. Calculations used WHO-TEFs.*

SUMMARY OF CONTAMINANT TRENDS IN SELECTED INDICATOR SPECIES

Great Blue Heron

Chlorinated contaminants (OC, PCBs, PCDD/Fs) in eggs from the great blue heron colony at the University of British Columbia in the Lower Fraser Valley have been monitored since the late 1970s.

DDE levels have declined from peaks in the late 1970s. The major source of DDE is from historical use of DDT for agricultural activities, although atmospheric sources also contribute to body burdens. Present levels are below toxic thresholds.

Concentrations of total PCBs in eggs declined a few years after regulatory controls (1977) and have fluctuated at lower levels from the early 1980s to 1994. While eggs collected in 1996 had the highest levels measured since 1983, it is too early to determine if this is an increasing trend. PCB concentrations measured in heron eggs in the late 1970s/early 1980s might have affected embryonic development, but more recent values are below toxicity thresholds.

A substantial decline in PCDD/F levels in heron eggs has been observed since 1989. In particular, concentrations of 2,3,7,8-TCDD/F dropped after implementation of chlorine dioxide substitution at upstream pulp mills, and levels of PnCDDs and HxCDDs diminished after bans on, and reduced usage of, chlorophenolic antisapstains and wood preservatives.

Investigations of biological effects associated with contaminant burdens in heron eggs estimated that, in 1989, herons were affected by PCDD/F contamination, but reductions in body burdens over the next several years greatly improved population health, including reproductive success.

Double-crested Cormorants

Eggs from the double-crested cormorant colony located at the Fraser estuary have been monitored for chlorinated contaminants since the mid 1980s.

Similar to trends in great blue herons, levels of DDE, PCBs and PCDD/Fs have declined in double-crested cormorant eggs since restrictions on their use were implemented.

Extensive biological studies have not been carried out for double-crested cormorants in the Fraser estuary colony. However, TEQs in the Fraser estuary colony were lower than TEQs correlated with biological effects in Lake Ontario, suggesting present levels of chlorinated contaminants are toxicologically inconsequential.

Bald Eagles

Eggs from bald eagle nests around the Lower Fraser Valley were collected in 1990 and 1991 and analyzed for chlorinated contaminants. Plasma from eagle nestlings was collected during the same time period.

Bald eagles had higher DDE concentrations than other species in the Lower Fraser Valley. Several eggs contained DDE levels within the range associated with reproductive effects, and mean eggshell thickness was less than values reported in the pre-DDT era (earlier than 1947). Although DDE concentrations may still be limiting productivity of some eagles nesting in the Lower Fraser Valley, impacts at the population level have been offset by other positive factors (such as abundant food supply) leading to increased survival of juveniles and adults.

Total PCB residues in eagle eggs collected in the early 1990s were below toxic thresholds.

Bald eagle eggs collected from the Lower Fraser Valley in 1990–91 contained higher total PCDD/F residues compared to those detected in ’90–91 heron and cormorant eggs. The proportion of 2,3,7,8-TCDD/F in eagle eggs was much greater than that detected in other estuarine birds, but levels were comparable to residues in fish tissues collected during that time period. As 2,3,7,8-TCDF is eliminated quickly from the body, the elevated levels detected in eagle eggs and chicks likely reflect recent exposure to pulp mill contaminated prey and direct deposition of contaminated lipids into the yolk.

The major contributor to total TEQs was 2,3,7,8-TCDD indicating that pulp mill effluent was still a dominant source of toxicity to eagles in the early 1990s. The PCDD/F burdens do not appear to affect reproductive success in Fraser Valley populations based on average productivity measured between 1990–94.

While bald eagle eggs had higher mercury levels than eggs from herons in the Lower Fraser Valley and osprey from the upper reaches of the Fraser River, levels were still below toxic thresholds.

Osprey

Osprey egg collections have been conducted primarily upstream and downstream of pulp mill sites along the Fraser and Thompson rivers since 1991.

Similar to eagle eggs, osprey eggs contained comparatively high amounts of DDE relative to other species. Twenty percent of eggs had concentrations exceeding toxic thresholds. Eggs with elevated DDE levels also showed evidence of egg shell thinning. Laboratory hatching success in an embryotoxicological study found that unhatched eggs contained, on average, two-fold higher DDE levels than eggs which hatched. These data suggest that DDE might be limiting productivity of some osprey nesting along the Fraser and Thompson rivers. Spatial trends suggest the primary local source of DDE for osprey is likely ongoing soil release of DDT compounds present from past usage in agricultural areas where DDT was historically used in large quantities. The high degree of individual variability within sites is likely caused by differences in diet, or related food chain factors, or by differences in exposure outside of breeding grounds.

Total PCBs, specifically PCB 126, were positively correlated with biochemical responses (hepatic EROD induction, elevated plasma vitamin A) in osprey embryotoxicity studies conducted in 1995 and 1996. The implications for individual and population health are not known.

Ospreys, along with other species collected from the upper reaches of the basin, were predominately contaminated with OCDDs. The higher chlorinated PCDDs are believed to have originated from a pentachlorophenol (PCP) source as there is a significant correlation between PCP and OCDD/HpCDD levels in eggs from the Thompson River. Tetra-chlorinated congeners, most reflective of pulp mill sources, were higher in eggs collected downstream of the pulp mill at Kamloops, although levels have declined since 1991 as a result of implementation of chlorine dioxide substitution at the mill.

TEQs for osprey were “low” compared to other species, principally because the TEF for OCDD is relatively small. Highest TEQs were observed in eggs collected downstream of pulp mills. Productivity (clutch size, fledgling success) was lower in downstream nests compared to upstream nests, although this difference has declined. However, embryonic studies undertaken in 1995 and 1996 showed no difference in hatching success betwen the two areas. The downstream chicks exhibited physiological responses to chlorinated contaminants (hepatic EROD induction, elevated plasma vitamin A); these responses were correlated to total PCBs and PCB 126 and not 2,3,7,8-TCDD. It is possible that PCBs negatively affect subsequent development characteristics like fledgling success.

Tree Swallows

Tree swallow nestlings were collected from nest boxes constructed upstream and downstream of pulp mills in the upper Fraser River in 1994.

DDE and total PCB levels were below toxic thresholds.

The dominance of highly chlorinated PCDD congeners relative to the other congeners and their correlation with PCP suggests that insect larvae are biomagnifying these compounds from contaminated runoff where chlorophenol-treated wood is used or manufactured.

Tree swallows had lower TEQs compared to osprey also collected from the same area, due to the lower concentration of tetra-congeners, especially 2,3,7,8-TCDD, associated with pulp mill contaminant sources. The toxicological significance of TEQs in whole-body homogenates is unknown.

Mink and River Otter

Mink and river otter were collected from the upper and lower reaches of the Fraser River in 1991 and 1995; liver tissues were analyzed for chlorinated contaminant residues.

DDE values were low compared to other species sampled in the same area and were toxicologically irrelevant.

Mink and otter from the Fraser Basin had low levels of PCBs compared to the Columbia River Basin.

Compared to river otter from the upper reaches of the Fraser River, otter from the Lower Fraser Valley had significantly higher concentrations of total PCDD/F, which were dominated by higher-chlorinated congeners, possibly reflecting the greater prevalence of industrial activities.

Similarly, river otter from the lower Fraser River had higher TEQs compared to those collected from upper reaches. All TEQ values in mink were below the critical body burden which is known to affect reproduction.

The highest mercury levels of all species tested were detected in river otter. Whether these levels pose a health risk to otters is unknown. However, the combined presence of mercury and PCBs could pose some risk to mink along the Fraser River.

General Conclusions on the Extent of Contamination in Fraser Basin Aquatic-Based Wildlife

Monitoring trends showed that during the 1980s, some indicator species in the Fraser River Basin had levels of chlorinated contaminants above concentrations associated with toxicological impairment. Contaminant levels declined very rapidly in the environment after regulations were implemented to eliminate chlorinated contaminants in mill effluent. However, contaminants still pose concern for a number of indicator species, notably DDE in osprey, TCDD in bald eagles and possibly PCBs in mink and otter. In addition, the world is a dynamic place and atmospheric transportation of specific contaminants may result in an increased concentration in local wildlife even though the chemical is banned from use in our country. Alternatively, it is possible that specific hot spots remain to be discovered (*e.g.* Cale Creek).

RESEARCH AND MONITORING RECOMMENDATIONS

Monitoring of contamination in indicator species should be continued to assess the persistence and bioavailability of contaminants in the Fraser River Basin.

Productivity and reproductive success of eagles, cormorants and herons at the presently monitored sites should continue on a three- to five-year rotation to take advantage of the baseline and trend data established over the last 20 years. This data would also provide information on broader ecological trends related to habitat disturbance and food chain productivity that need environmental management action beyond that required to control toxic contamination.

The utility of otter and mink as indicator species should be further examined. While they both "sample" the environment at different scales and points in the food chain, their use in combination might be very advantageous in the long term. The possibility of using live trapping for obtaining blood samples and observing condition factors is also attractive.

Although we have made advances in assessing the impacts of less-chlorinated dioxins and furans, the movement and toxicity of higher-chlorinated dioxins in the local food chains are not well understood. There is a need to evaluate these congeners which continue to be released to the environment from combustion sources, wood preserved with pentachlorophenol solutions and pulp mills.

Ospreys breeding on hydroelectric reservoirs can accumulate mercury levels two- to three-fold higher than birds on naturally formed water bodies (Hughes *et al.* 1997). A study of mercury accumulation in ospreys breeding on hydroelectric reservoirs in the Fraser Basin is warranted.

Ospreys eggs at DDE 'hot spots' (*e.g.* Nicola Lake, Moose Lake) should be assessed for DDE levels, fledgling success and general productivity.

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Table 1. Concentrations of organochlorine pesticides (µg/kg wet weight) in wildlife from the Fraser River Basin. Values are geometric means (of 'N' samples) or one analysis of a sample composite.

SPECIES	SITE	YEAR	N[a]	DDE	DDT	OXY[b]	t-NONA-CHLOR	DIELDRIN	HE[c]	HCB[d]	β-HCH[e]	MIREX
Great blue heron (eggs)	UBC	1977	12	2354	66	10	NA[f]	73	64	20	NA	22
		1982	11	1424	77	50	70	72	38	NA	10	12
		1983	10p	582	NA	23	22	20	35	NA	NA	NA
		1985	10	1047	19	37	112	49	32	17	3	16
		1986	7	273	9	11	27	10	7	11	NA	5
		1987	25	510	24	25	60	21	16	15	NA	6
		1988	13	447	9	17	36	17	13	13	4	9
		1989	5p	531	16	21	47	19	17	14	5.2	4
		1990	8	415	12	11	27	13	6	7.0	4.0	4
		1991	5p	808	22	28	43	13	8	15	11	13
		1992	10	280	1	11	30	10	5	10	6.0	4
		1993	6p	1622	19	31	59	54	14	14	11	11
		1994	5p	1020	31	27	34	18	52	9.4	5.6	7
		1996	5p	926	10	22	44	58	28	13	<0.1[g]	11
Double-crested cormorant (eggs)	Estuary	1985	5	464	5	12	NA	4	3	14	6	8
		1988	7	531	1	3	2	2	1	15	5	5
		1990	7p	176	1	6.2	<0.3	6	2	7.7	7.0	3
		1991	5p	236	<0.1	6.2	<0.3	5	2	8.6	5.6	4
		1992	10p	222	2	5.6	<0.3	2	2	9.4	4.6	3
		1993	10p	382	2	5.2	0.4	1	2	9.4	4.1	3
		1994	9p	411	1	6.0	0.3	1	2	7.6	4.2	2
Bald eagle (eggs)	Brunswick Point	1990	1	2536	NA	33	135	66	69	16	11	9.7
	Annacis Island	1990	1	4142	NA	82	165	91	38	42	32	38
	Chehalis Flats	1990	1	4912	NA	31	234	44	6.5	18	3.1	14
	Island 20	1991	1	898	9.1	42	82	24	11	21	7.2	<1.1
	Cheam Island	1991	1	1171	<0.4	35	125	20	4.0	30	<0.5	17
	Agassiz Bridge	1991	1	1916	11	22	151	21	10	37	6.7	15

Table 1. Continued from previous page.

SPECIES	SITE	YEAR	N[a]	DDE	DDT	OXY[b]	t-NONA-CHLOR	DIELDRIN	HE[c]	HCB[d]	β-HCH[e]	MIREX
Osprey (eggs)	Nechako R.	1992	5	1384	20	1.9	0.8	3.5	2.8	1.1	3.2	0.8
		1996	2	2357	10	3.5	<1.1	0.5	3.4	<0.8	0.5	<1.0
	Quesnel	1991	4	1014	NA	<8.5	NA	0.1	1.8	0.2	NA	NA
		1992	5	1309	3.1	2.7	<0.5	5.2	3.0	0.1	0.4	2.0
	S. Thompson R.	1991	6	1174	NA	<6.7	NA	0.1	1.6	1.2	NA	NA
		1992	5	1800	28	3.7	1.2	1.5	2.4	3.0	1.3	3.3
		1997	5	1194	10	0.8	0.7	0.1	0.5	0.5	<1	1.4
	Thompson R.	1991	5	1646	NA	0.1	NA	0.7	2.7	0.6	NA	NA
		1992	5	1294	15	4.2	2.3	2.2	4.1	2.6	4.2	2.8
		1995	1	3173	18	3.8	<1.1	<0.2	2.5	<0.8	5.8	<1
		1997	5	706	1.1	1.8	0.3	0.2	0.4	0.3	<1	0.5
	Pitt R.	1996	3	5429	<0.2	3.7	<1.1	0.3	1.4	<0.8	0.2	<1
Tree swallow (nestlings)	S. Thompson R.	1994	3p	1783	ND	1.4	<0.1	0.8	0.7	<0.05	<0.05	<0.15
	Thompson R.	1994	11p	87	ND	1.5	<0.1	1.2	ND	<0.05	<0.05	<0.15
	upper Fraser R. - u/s	1994	18p	90	ND	1.3	<0.1	0.4	0.4	2.2	<0.05	<0.15
	upper Fraser R. - d/s	1994	14p	165	ND	1.5	<0.1	0.3	1.1	<0.05	<0.05	<0.15
River otter (liver)	Shorty Creek	1991	1	<1.0	<1.0	1.7	<1.0	1.3	<1.0	2.7	<1.0	<1.0
	upper Fraser R.	1995	6	<2	<2	NA	NA	<2	5.8	NA	NA	NA
	Lower Fraser Valley	1995	3	9.4	<2	NA	NA	<2	5.0	NA	NA	NA
Mink (liver)	upper Fraser R.	1991	6p	3.7	<1.0	1.7	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
	Cale Creek 1	1991	1	6.9	<1.0	>0.1[h]	<1.0	<1.0	<1.0	<1.0	<1.0	>0.1
	Cale Creek 2	1991	1	9.0	<1.0	1.0	<1.0	>0.1	<1.0	1.0	<1.0	<1.0
	Cale Creek 3	1991	1	1.4	<1.0	3.6	<1.0	1.2	<1.0	<1.0	<1.0	<1.0
	Redrock Lake	1991	1	1.3	<1.0	<1.0	<1.0	2.3	<1.0	<1.0	<1.0	<1.0
	Stone Ck Canyon	1991	1	4	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0	<1.0
	upper Fraser R.	1995	12	6	1.4	NA	NA	1.4	<2	NA	NA	NA
	Lower Fraser Valley	1995	8	31	<2	NA	NA	4.3	<2	NA	NA	NA

[a] *'p' = one pool of listed number of individual eggs or whole body homogenates*
[b] *Oxy = oxychlordane*
[c] *HE = heptachlor epoxide*
[d] *HCB = hexachlorobenzene*
[e] *b-HCH = b-hexachlorocyclohexane*
[f] *NA = not available*
[g] *Detection limit is contaminant specific therefore listed individually as '< particular value'*
[h] *Non-quantifiable peak observed below the calculated minimum detection limit, but still indicating presence of the compound*

Table 2. Concentrations of total PCBs in wildlife from the Fraser River Basin. Values are geometric means (of 'N' samples) or one analysis of a sample composite.

SPECIES	SITE	YEAR	N[a]	TOTAL PCBS (mg/kg wet wt.)	NON-ORTHO PCBS (ng/kg wet wt.)			MONO-ORTHO PCBS (µg/kg wet wt.)	
					77	126	169	105	118
Great blue heron (eggs)	UBC	1977	12	7.31	NA[b]	NA	NA	NA	NA
		1982	11	4.44	NA	1055	132	107	536
		1983	10p	1.57	NA	392	54	37	190
		1985	10	2.39	NA	582	77	42	297
		1986	7	1.05	NA	273	41	30	113
		1987	25	1.23	NA	314	45	3.4	18
		1988	13	1.37	NA	345	49	9.3	35
		1989	5p	0.79	NA	212	33	0.2	97
		1990	8	1.00	178	319	47	17	118
		1991	5p	1.76	NA	436	60	33	265
		1992	10	1.01	NA	263	39	22	102
		1993	6p	1.70	NA	423	58	24	230
		1994	5p	1.92	988	653	29	41	275
		1996	5p	3.41	NA	NA	NA	NA	NA
Double-crested cormorant (eggs)	Estuary	1985	5	1.23	NA	336	48	27	141
		1988	7	0.88	NA	213	34	13	151
		1990	7p	0.74	57	252	32	16	92
		1991	5p	0.76	NA	172	30	16	111
		1992	10p	0.75	NA	167	29	17	96
		1994	9p	0.71	11	317	27	12	78
Bald eagle (eggs)	Brunswick Point	1990	1	2.87	282	446	84	44	202
	Annacis Island	1990	1	6.21	605	1070	221	91	429
	Chehalis Flats	1990	1	2.96	291	462	87	46	194
	Island 20	1991	1	1.09	110	113	11	18	100
	Cheam Island	1991	1	2.33	230	346	62	36	191
	Agassiz Bridge	1991	1	2.73	269	420	78	45	240

Table 2. Continued from previous page.

SPECIES	SITE	YEAR	N[a]	TOTAL PCBS (mg/kg wet wt.)	NON-ORTHO PCBS (ng/kg wet wt.)			MONO-ORTHO PCBS (µg/kg wet wt.)	
					77	126	169	105	118
Osprey (eggs)	Nechako R.	1992	5	0.10	21	37	18	0.8	6.6
		1996	2	0.35	49	87	23	0.9	1.6
	Quesnel	1991	4	0.27	35	74	15	6.2	22
		1992	5	0.29	39	166	26	1.1	32
	S. Thompson R.	1991	6	0.37	50	55	25	7.8	28
		1992	5	0.38	32	138	37	4.0	23
		1997	5	0.25	54	94	26	4.9	15
	Thompson R.	1991	5	0.30	16	59	11	5.7	22
		1992	5	0.29	37	77	15	4.5	18
		1995	1	0.23	8.6	27	94	<1.4[c]	15
		1997	5	0.20	33	71	11	3.8	11
	Pitt R.	1996	3	0.44	44	179	23	0.3	26
Tree swallow (nestlings)	S. Thompson R.	1994	3p	<0.1	59	9	0.8	<0.15	<0.2
	Thompson R.	1994	11p	0.03	163	35	2.0	<0.15	7.2
	upper Fraser R. - u/s	1994	18p	<0.1	18	4.6	1.0	<0.15	<0.2
	upper Fraser R. - d/s	1994	14p	0.02	80	42	2.8	<0.15	<0.2
River otter (liver)	Shorty Creek	1991	1	38	3.5	5.8	18	<0.3	<0.3
	upper Fraser R.	1995	6p	0.02	2	1	6	<0.02	<0.03
	Lower Fraser Valley a	1995	3p	0.2	7	68	36	3.3	1.5
	Lower Fraser Valley b	1995	3p	0.2	2	6	9	<0.02	0.4
Mink (liver)	upper Fraser R.	1991	6p	4.1	4.2	14.4	<1.8	<0.1	<0.1
	Cale Creek 1	1991	1	18	3.3	39.3	4.6	<0.1	<0.1
	Cale Creek 2	1991	1	24	3.4	13.5	3.2	<0.1	<0.1
	Cale Creek 3	1991	1	5.5	2.6	6.7	4.0	<0.1	<0.1
	Redrock Lake	1991	1	<1	4.4	6.8	3.1	<0.1	<0.1
	Stone Ck Canyon	1991	1	5.0	3.3	2.9	1.3	17	29
	Lower Fraser Valley	1995	2	0.02	18	62	<5	0.4	1.4

[a] *'p' = one pool of listed number of individual eggs or whole body homogenates*
[b] *NA = not available*
[c] *Detection limit is contaminant specific therefore listed individually as '< particular value'*

Table 3. Major PCDDs and PCDFs (ng/kg wet wt.) in heron, cormorant and eagle eggs and mustelid liver tissue collected from sites within the lower Fraser River Basin. Values are geometric means (of 'N' samples), one analysis of a composite/pool of samples or one analysis of an individual.

SPECIES	SITE	YEAR	N[a]	2378 - TCDD	12378 - PnCDD	123678 - HxCDD	1234678- HpCDD	OCDD	2378- TCDF	23478- PnCDF	TEQS[b]
Great blue heron (eggs)	UBC	1982	11p	76	491	740	31	76	NA[e]	NA	700
		1983	10p	15	59	104	ND[c]	27	11	48	179
		1985	10	22	85	125	2.0	0.3	0.2	18	191
		1986	7	25	74	104	3.5	5.9	3.0	16	150
		1987	10	55	49	83	1.3	0.2	17	16	169
		1988	13	86	27	37	2.8	3.5	5.7	10	165
		1989	5	133	103	112	3.0	1.9	19	38	319
		1990	8	46	47	52	0.6	0.8	7.5	18	153
		1991	5p	29	34	33	<3[d]	<5	4.0	15	132
		1992	10	9.2	5	8.3	0.1	0.1	0.3	0.003	40
		1993	6p	9.7	35	33	14	11	<2.8	<2.7	92
		1994	5p	4.4	12	18	<0.8	13	1.2	4.5	95
Double-crested cormorant (eggs)	Estuary	1985	5	20	81	188	0.04	<7.2	ND	13	152
		1988	7p	74	34	52	8	6	3	14	151
		1990	7p	11	5	7	<1	<10	<2	<2	44
		1991	5p	18	10	23	<6	<10	<1	8	56
		1992	10p	8	14	11	<0.1	<5.2	<1	<3.5	42
		1994	9p	3.3	8.2	11	1.0	1.4	<1.3	2.3	48
Bald eagle (eggs)	Brunswick Point	1990	1	42	37	42	<2	<3	23	13	182
	Annacis Island	1990	1	58	55	112	9	<4	112	12	390
	Chehalis Flats	1990	1	58	52	55	<6	<10	89	14	282
	Island 20	1991	1	51	7	17	<6	<9	16	<3	96
	Cheam Island	1991	1	23	6	15	<6	<10	73	2	157
	Agassiz Bridge	1991	1	41	15	18	<3	<3	13	10	143
River otter (liver)	Lower Fraser Valley (a)	1995	3p	1	2	57	120	160	0.5	15	26
	Lower Fraser Valley (b)	1995	3p	<0.2	0.6	4.1	39	180	<0.2	1.5	3.0
Mink (liver)	Lower Fraser Valley	1995	2	<1.9	<3.7	<4.0	<7	<13	<1.9	<1.7	

[a] 'p' = one pool of listed number of individual eggs or whole body homogenates
[b] TEQs include all PCDD/F congeners, non-ortho PCBs 77, 126, 169 and mono-ortho PCBs 105, 118 except for Great Blue Heron and Double-crested Cormorants which don't include CB 77. Calculations were completed using WHO-TEFs
[c] ND = not detected; detection limit not specified
[d] Detection limit is contaminant specific therefore listed individually as '< particular value'
[e] NA - not analyzed

Table 4. Major PCDDs and PCDFs (ng/kg wet wt.) in osprey eggs, nestling tree swallow (whole body homogenates) and mustelid liver tissue collected from sites within the upper Fraser River Basin. Values are geometric means (of 'N' samples) or one analysis of a sample composite.

SPECIES	SITE	YEAR	N[a]	2378-TCDD	12378-PnCDD	123678-HxCDD	1234678-HpCDD	OCDD	2378-TCDF	23478-PnCDF	TEQS[b]
Osprey (eggs)	Nechako River	1992	5	1.4	3.7	13	54	148	0.5	0.9	13
		1996	5	0.5	3.5	7.0	32	69	0.2	0.3	17
	Quesnel	1991	4	5.5	2.3	5.3	16	33	<1.0[c]	0.7	20
		1992	5	7.3	5.3	12	36	143	0.4	1.5	35
	S. Thompson River	1991	6	11	2.6	14	92	354	1.4	0.9	27
		1992	5	8.0	6.4	16	133	475	0.7	1.6	35
		1997	5	0.6	3.3	8.1	48	198	0.5	0.3	18
	Thompson River	1991	5	46	4.2	21	79	112	2.5	2.4	64
		1992	5	27	4.9	11	38	88	3.7	2.4	49
		1995	1	7.8	4.2	16	101	513	2.8	0.7	22
		1997	5	3.6	1.2	22	26	97	0.9	0.4	16
	Pitt River	1996	3	1.6	4.2	5.0	1.6	3.2	<0.4	0.9	28
Tree Swallow (nestlings)	S. Thompson River	1994	3p	<0.8	<0.8	<1.2	2.2	2.7	1.3	<0.9	5.2
	Thompson River	1994	11p	<0.9	<0.7	1.2	1.9	2.4	3.2	<1	14.9
	upper Fraser River - u/s	1994	18p	<1.1	<0.6	<0.5	1.7	2.1	0.6	<0.2	1.9
	upper Fraser River - d/s	1994	14p	<1.6	1.8	5.7	36	34	1.1	<0.8	11.8
Mink (liver)	upper Fraser River	1991	6p	<0.6	<0.3	2.9	8.0	26	<0.9	0.5	2.2
	Cale Creek 1	1991	1	<1.6	1.8	22	123	187	<0.8	3.1	9.3
	Cale Creek 2	1991	1	<1.9	<1.8	1.5	18	16	<1.8	<0.9	1.6
	Cale Creek 3	1991	1	<0.7	<0.8	<0.4	1.1	3.5	<0.8	<0.2	0.3
	Redrock Lake	1991	1	<0.4	<1	<0.8	<1.3	1.6	<0.4	<0.6	0.9
	Stone Creek Canyon	1991	1	<0.7	<0.9	<0.6	<1.2	1.4	<0.9	<0.4	2.4
River Otter (liver)	Shorty Creek	1991	1	<1	<1	11	16	20	<1.1	<0.4	0.9
	upper Fraser River	1995	6p	<0.2	0.5	0.6	2.3	5.3	<0.2	0.3	1.0

[a] *'p' = one pool of listed number of individual eggs, whole body homogenates or livers*
[b] *TEQs for all species include all PCDD/F congeners, non-ortho PCBs 77, 126, 169 and mono-ortho PCBs 105, 118 except for Great Blue Heron and Double-crested Cormorants which don't include CB 77. Calculations were completed using WHO-TEFs*
[c] *Detection limit is contaminant specific therefore listed individually as '< particular value'*

3.10

THE DELTA FORESHORE ECOSYSTEM: PAST AND PRESENT STATUS OF GEOCHEMISTRY, BENTHIC COMMUNITY PRODUCTION AND SHOREBIRD UTILIZATION AFTER SEWAGE DIVERSION

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THE FRASER RIVER ESTUARY AND ITS INTERTIDAL REGION

Of the various ecosystems that together describe the Fraser River and its estuary, the most understudied but ecologically important is its mudflat/intertidal region. The intertidal region of the estuary provides vital habitat for a diverse macro and meiofauna, which in turn supports, during winter months, the highest densities of waterfowl, shorebirds and raptors in Canada. Furthermore, this intertidal area provides habitat for numerous fish species, as well as serving as the point of entry and exit for five salmon species. Not surprisingly, the Fraser River Estuary is one of the most biologically productive systems in Canada (Kennett and McPhee 1988). It is also adjacent to Canada's third largest urban region which is the source of much of the pollution stress in the estuary. Despite its ecological importance, little is known about the fundamental biochemical and ecological response of the estuary's intertidal zone to this pollution. Given that anthropogenic pressures are increasing, it is critical that comprehensive knowledge on the impacts of present and future contaminant exposure be obtained to effectively protect and conserve this unique habitat.

As present loading is predominantly delivered via the river, an overview of the distribution of flow is described below. Approximately 25 km upstream from the mouth, the river bifurcates into the North Arm and the Main Arm (Fig. 1). The North Arm, which carries approximately 16 per cent of the total river discharge, bifurcates again at Sea Island where about 30 per cent of the flow (~5% of the total Fraser River flow) exits via the Middle Arm onto Sturgeon Bank while the remaining 70 per cent (9% of the total flow) exits just north of Sturgeon Bank through the North Arm (Feeney 1995). The Main Arm carries the majority of the flow (>80%) and exits onto Roberts Bank between Westham and Lulu Island. A small portion of the flow goes through Canoe Pass, south of Westham Island (Kennett and McPhee 1988). The maximum discharge occurs in June (~10,000 m^3/s) and the minimum in February/March (~750 m^3/s).

The tidal cycle is a major factor influencing biological and biogeochemical processes in the intertidal zone through its control of the period and range of inundation and exposure. The maximum tidal range (~5 m) occurs in June and December, during spring tides, and the lowest range (~3 m) occurs in March and September.

The Fraser River delta, an area of ~337 km^2, lies between the North Arm of the Fraser River and Boundary Bay (Kistritz 1978). This area can be divided into six relatively distinct ecosystems: (1) Iona Island sand flats between the Iona Island Jetty and the North Arm; (2) Sturgeon Bank; (3) Roberts Bank; (4) the inter-causeway, between the coal port terminal and the ferry terminal; (5) south of the ferry terminal to Point Roberts peninsula; and (6) Boundary Bay. None of the foreshore habitats are undisturbed as diking has cut off much of the original high marsh from the sand flats. The sediment texture is variable and sandy areas dominate over finer-grained substrates.

The estuary has abundant fauna and flora. The shoreline of Sturgeon Bank is bordered by a marsh composed primarily of sedges (*Carex* sp.) and bullrushes (*Scirpus*

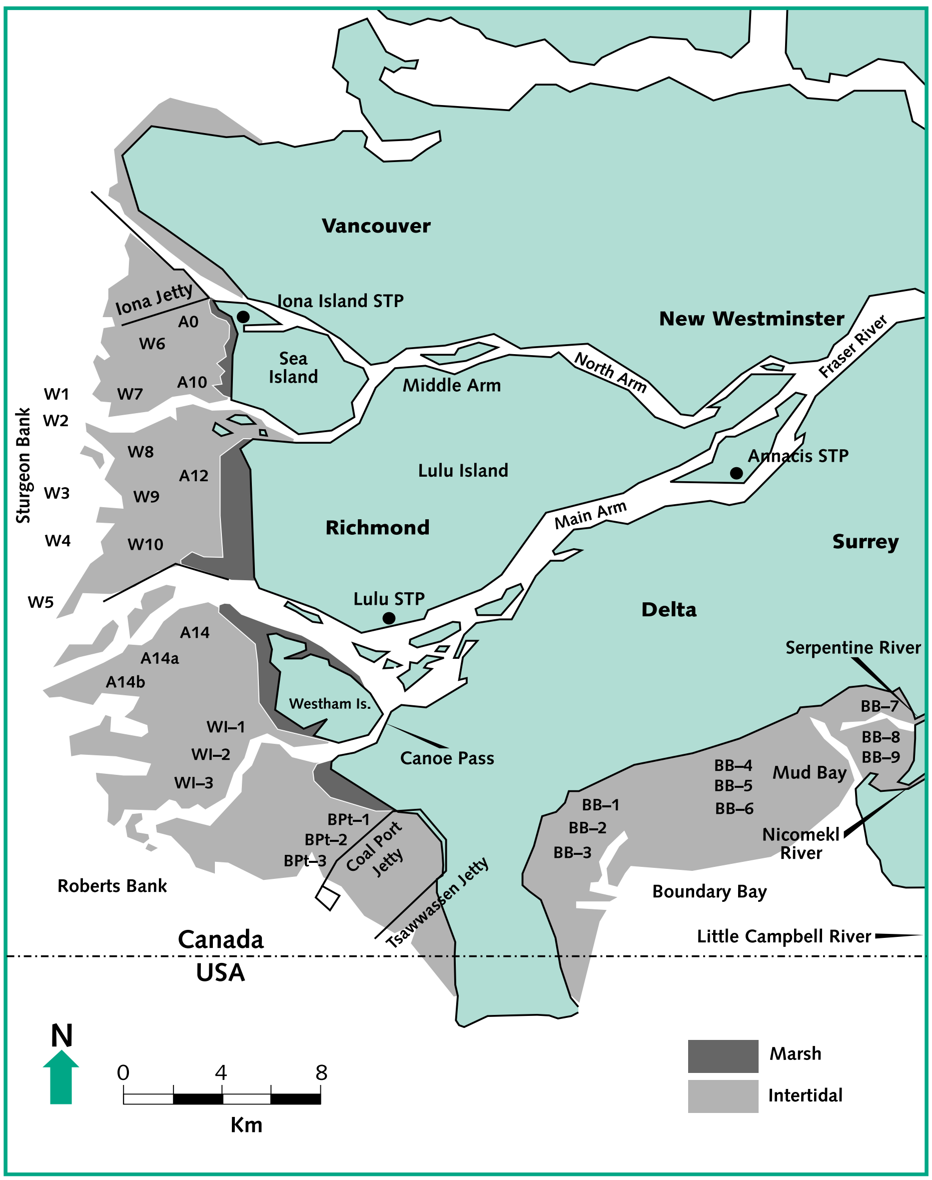

Figure 1. Map of the study area showing sampling sites on Sturgeon Bank, Roberts Bank and Boundary Bay.

sp.) (BC Research 1975). The main primary producers are benthic microalgae (*e.g.* diatoms) on top of the sediment, but benthic macroalgae such as *Ulva* and *Enteromorpha* are occasionally present. The intertidal area has a diverse invertebrate infauna dominated by polychaete worms, bivalves, amphipods, isopods, decopods, harpacticoid copepods, nematodes and oligochaetes (Otte and Levings 1975; BC Research 1975). In addition, 27 species of fish, including commercial species of salmon, have been reported on Sturgeon Bank (Birtwell *et al.* 1983). Young salmon migrate down the Fraser River and feed extensively in the nearshore areas of the estuary and undergo osmoregulation changes before moving out to sea.

The estuary supports internationally important populations of birds, including some threatened species. In particular, the vast intertidal flats from Iona Island to Boundary Bay provide critical habitat for millions of migratory waterfowl (*e.g.* snow geese) and shorebirds. The shorebirds, in particular, depend on benthic invertebrates during their migration or while they overwinter. Despite international recognition of the importance of these intertidal flats to birds, the flats and connecting foreshore continue to be altered by industrial port development and urbanization, and are exposed to contaminated water and sediments.

The Fraser River transports contaminants introduced to the river from industrial and municipal discharges, non-point source runoff from agricultural and urban land and, in the case of metals, from natural geological erosion and leaching. The contaminants are carried in solution, colloidal phases, and adsorbed to suspended organic and inorganic sediments. Some portion of all these phases can end up being deposited in the estuarine sediments of the delta's extensive intertidal sand and mudflats (approximately 150 km^2 to the 10 m deep shelf break and about 50 km^2 to lowest tide depth).

While the rest of the delta front has been receiving seasonally variable inputs of contaminants, Sturgeon Bank, particularly the northern part off Iona and Sea Islands, received direct inputs of sewage through the outfall from the Iona Island wastewater treatment plant (WWTP) from 1962–1988. During this time, the plant served a population of approximately half a million people. In 1988, sewage was redirected to discharge into the Strait of Georgia at a depth of 90 m.

OBJECTIVES

The objectives of our research program were the following: (1) to increase the knowledge of the nutrient dynamics, and of primary and secondary production of the benthic community in this unique estuarine ecosystem; (2) to identify food web links between benthic biota and shorebirds; (3) to assess metal contamination in sediments and biota and identify their sources, sinks and potential impacts; (4) to quantify the extent of recovery of the area near the original site of sewage discharge; and (5) to establish a baseline of biological and chemical characteristics to measure the response of the delta ecosystem to the WWTP effluent treatment upgrade, which will be completed by 1999 at two major plants. Prior to their upgrade, these plants discharged primary treated effluent from approximately 800,000 people and associated industrial/institutional discharges into the Main Arm of the river.

POLLUTION STRESS

The benthic community on the intertidal zone is exposed to high levels of natural environmental stress caused by tidal fluctuations which control the light climate, wave energy and inputs of fresh water and associated suspended sediments from the three arms of the Fraser entering the delta front. In addition, there have been alterations to the way the river interacts with the delta through the construction of dikes and training walls and the dredging of navigation channels. Most importantly, there has been a long history of pollution with metals (Bawden *et al.* 1973; McGreer 1979), PAHs, heavy wood preservatives, chlorinated phenols (Carey and Hart 1988), pesticides (Hall *et al.* 1991), and sewage effluents containing organic matter and nutrients (Otte and Levings 1975; Birtwell *et al.* 1983). Our studies focused on the impact of past discharges of nutrients and metals from the Iona Island WWTP and on a general assessment of metal contamination in the whole area including Boundary Bay.

SAMPLE SITES

Sites along the intertidal zone of the lower Fraser River Estuary were chosen to obtain data on the influence of the Fraser River, past pollution from the Iona Island WWTP discharge, and the composition of sediment (grain size and per cent organic matter) on the distribution of metals (all sites, Fig. 1) and of nutrients and productivity (sites labeled A, Fig. 1).

Three areas of the foreshore region were sampled: (1) the most northerly area, Sturgeon Bank, is estimated to receive 15 per cent of the industrial and municipal wastes discharged into the Fraser River (Fraser River Estuary Study 1979; Feeney 1995) and, before 1988, it received primary treated sewage effluent directly onto its foreshore. This effluent contained trace metals, such as copper, iron, lead, mercury, nickel and zinc (Tevendale and Eng 1984); (2) the Roberts Bank area receives approximately 80 per cent of the total flow of the Fraser River and is estimated to receive 60 per cent of the municipal and industrial effluent discharged into the Fraser River including the increasing discharges from the Annacis and Lulu Island WWTPs (Fraser River Estuary Study 1979); and (3) Boundary Bay, which is influenced by the Serpentine, Nicomekl and Little Campbell rivers which drain a combination of agricultural and residential areas. In addition, drainage from the agricultural areas surrounding the bay is discharged at five pump stations along the west and north shores (Swain and Walton 1990). Both Roberts Bank and Boundary Bay had nine sampling locations, whereas Sturgeon Bank had eight for a total of 26 sampling locations.

ASSESSMENT OF METAL CONTAMINATION

Spatial variability in sediment texture and associated contaminants within the intertidal area is high. To obtain a representation of metal levels in sediments from the study areas, several locations from each site were sampled, as indicated above. Boundary Bay, in most respects, can be considered a control area for comparison.

Mathematical normalization to eliminate the effect of sediment grain size was not performed for bulk trace metal data in our study because the fine sediment fraction (grain size of less than 0.05 mm) represented less than 60 per cent of the sediment samples. Horowitz and Elrick (1988) suggest that grain size normalization should only take place when the fine fraction of the sediment represents at least 50 to 60 per cent of the sample.

Sturgeon and Roberts banks had higher concentrations of metals (*e.g.* mercury [Hg], copper [Cu], zinc [Zn] and nickel [Ni]), except cadmium (Cd), which was higher in Boundary Bay, and possibly linked to use of a fungicide on golf course greens in the area (Swain and Walton 1990). In general, the highest concentrations of metals were observed in Roberts Bank, followed by Sturgeon Bank; Boundary Bay had very low metals levels, except for Cd. While the southern part of Sturgeon Bank is cleaner than Roberts Bank, the northern portion near Iona Island WWTP (Station AO) is still a metals "hot spot" and it had the maximum observation for Cu, Cd, lead (Pb), Zn and Hg (Table 1). Silver (Ag) is very high as well, as reported in the following section on sewage tracers.

The bioavailability of a metal is related to its association with the various geochemical components present in the sediment (Bendell-Young and Harvey 1992). Metals associated with the lattice framework of the sediment, particularly in the sand and possibly the larger sizes of the silt fraction, are not considered

Table 1. Summary of maximum concentrations of metal in the total (aqua regia extract) and labile fractions of bed sediment from the Fraser delta in the spring (spr) and summer (sum), as well as the site at which the maximum occurred. Corresponding salinities for each location are presented in parentheses. Refer to Figure 1 for site locations.

METAL	SEASON	MAXIMUM 'TOTAL' (µg/g dry wt)	REGION	MAXIMUM LABILE (µg/g dry wt)	REGION	LOEL (µg/g dry wt)	
						MARINE	FRESH
Copper	spr	54.1	SB-A0 (10‰)	25.9	SB-A0 (10‰)	70	16
	sum	45.4	SB-A0 (10‰)	32.1	RB-BPt-2 (10‰)	70	16
Cadmium	spr	0.43	SB-A0 (10‰)	0.43	SB-A0 (10‰)	5.0	0.6
	sum	0.25	BB-BB-9 (32‰)	0.21	BB-BB-9 (32‰)	5.0	0.6
Nickel	spr	44.7	RB-WI-1 (3‰)	20.0	RB* (3-10‰)	30	16
	sum	49.0	RB-WI-1 (3‰)	31.3	RB-BPt-2 (10‰)	30	16
Lead	spr	19.0	SB-A0 (10‰)	12.7	SB-A0 (10‰)	35	31
	sum	18.7	SB-A0 (10‰)	13.9	SB-A0 (10‰)	35	31
Zinc	spr	95.8	SB-A0 (10‰)	65.2	SB-A0 (10‰)	120	120
	sum	100.8	RB-BPt-2 (10‰)	64.3	RB-BPt-2 (10‰)	120	120
Hg	sum	0.22	SB-A0 (10‰)	-	-	0.15	0.2

SB Sturgeon Banks
RB Roberts Bank
BB Boundary Bay
LOEL Lowest Observable Effects Level (Nagpal, 1994)
** found at several locations*

available for uptake by an organism as they are tightly bound within the sediment. However, metals associated with iron (Fe) and manganese (Mn) oxides and organic matter can either be taken up by organisms or not, depending on environmental factors such as redox conditions, other metal complexes and salinity (Luoma and Bryan 1981).

The Fraser River was found to be an important source of Mn oxides and trace metals to estuarine sediments. This has important implications, given that metals associated with Mn oxides have an enhanced bioavailability. Hence, the Fraser River could be contributing significantly to the bioavailable fraction of metals in the intertidal region. In contrast, iron, which is primarily supplied from some of the pore waters, reduces metal bioavailability. It acts as a modifier of metal uptake, either through a protective or competitive effect, with the potential of reducing metal uptake in an organism. The results of this study suggest that while greater than 50 per cent of each metal supplied by the Fraser River is in a bioavailable form (*i.e.* in the labile fraction of the sediment associated with Mn oxides), once they settle in the intertidal region, in areas where Fe is generated by diagenesis in the pore waters, this Fe modifies metal uptake and reduces or even prevents bioaccumulation.

The metal availability to organisms was tested by measuring the amount of various metals in the tissue and shell of the most common clam, *Macoma balthica*. In most cases, there was a strong correlation between the amount of a metal in the tissues of *Macoma* and the amount of the biologically available fraction of that metal in the sediment as measured by the easily reducible (ER) metal analysis.

Cadmium in bivalve tissues was negatively correlated with reducible (RED) Fe and positively correlated with easily reducible (ER) Cd—Cd associated with Mn oxides (r=0.59). Hence, higher tissue concentrations of Cd in *M. balthica* are predicted to occur at sites where concentrations of RED Fe are low and ER Cd is high. The negative correlation between Cd tissue concentrations and RED Fe suggests that this component is modifying what Cd is available for uptake, possibly through a 'protective' or 'competitive' effect. An explanation for this inverse dependence could be: (1) RED Fe (presumably as Fe oxides) that enters the gut competes with uptake sites on the intestinal tract for solubilized metals; (2) RED Fe becomes solubilized in the gut and, as a result, the Fe itself competes with trace metals for uptake sites; and (3) RED Fe adsorbs dissolved trace metals in the external phase such as on the gill or mantle tissue.

The strongest correlation was found between Pb in tissue and Pb in the ER phase (r=0.78). Luoma and Bryan (1978) found that the biological availability of Pb to *Scrobicularia plana* (a deposit-feeding estuarine bivalve) was controlled mainly by the concentration of Pb in the sediment extracted with a weak acid digestion similar to that used for the RED Fe fraction. It is important to note, however, that the extraction scheme in the Luoma and Bryan study did not include an ER phase.

Nickel concentrations in the tissues did not correlate significantly with any of the sediment parameters; however, concentrations in the shell correlated positively with per cent loss on ignition and negatively with total Ni concentrations (r=0.52). This suggests that higher concentrations of Ni in the shell are found at locations with higher organic matter and low total concentrations of Ni.

A weak correlation was found between shell Hg concentrations and total Hg in the sediment (r=0.34). A regression was not attempted for Hg in the tissues as there were insufficient sample numbers. Few studies have addressed the relationship between Hg in shells and sediment-bound Hg, although, previous studies have shown a strong relationship between total Hg in the sediment, normalized for organic matter, and tissue concentrations for *M. balthica* in British estuaries (r=0.80, r=0.74) (Langston 1982; 1985). Organic matter in sediment is believed to be the most important factor influencing Hg concentrations in these tissues (Langston 1985; Rae and Aston 1982). Nickel and Hg were the only metals for which either a negative or a positive correlation between shell concentrations and total sediment metal was found.

Copper concentrations in the tissue and the shell were correlated positively with both RED and ER Cu, that is, Cu associated with Fe and Mn oxides (r=0.65, r=0.71). Hence, high concentrations in the tissue and the shell are more likely to occur at sites high in both RED and ER Cu. Tissue and shell Zn concentrations correlated positively with concentrations of ER Mn recovered from the sediment (r=0.62, r=0.39). Therefore, high Zn concentrations in *M. balthica* occurred at locations high in ER Mn.

To provide a general assessment of the potential impact of the observed metal levels, these concentrations can be compared with B.C. working and approved sediment criteria for the protection of freshwater and marine life (Nagpal 1994). This comparison is not straightforward, however. There is high spatial variability in metal concentrations in sediments and, therefore, a very large number of sediment samples must be collected to truly assess the spatial extent and significance of any exceedances of the criteria. In addition, the criteria are based on the assumption that all the metal is in a bioavailable form, so the bioavailable concentrations should be used if available. Also, the sediment type influences the metal concentration and availability, with metal concentrations being generally higher in silt and clay than in sandy sediments. Finally, salinity affects the availability of the metal. Therefore, when salinities in the estuary are below that of sea water, freshwater criteria as well as marine criteria should be used.

The maximum total concentration of each metal in the spring and summer is presented in Table 1. In addition, the maximum concentration of each metal in the labile fraction is presented to illustrate the discrepancy in some cases between what is potentially available to an organism and the concentration of the total metal. While it was expected that the station nearest the old WWTP discharge (A0) could still exhibit metal concentrations approaching criteria, this was not always the case. For instance, sediments off Brunswick Point (the point just south of Westham Island on the Main Arm, Fig. 1) approached the criterion for Zn and exceeded it for Cu, and off Westham Island, the criterion for Ni was exceeded. Considering that salinity is an important factor governing the availability of metals (Luoma 1983) and that salinity varied from freshwater levels (nearest the outflow of the Fraser River) to marine levels (Boundary Bay and sites farthest offshore), sediment quality criteria for both marine and freshwater systems are also presented. For example, Cu did not exceed the criterion for marine systems but exceeded that for freshwater systems by two times at A0 and at BPt-2 (located in Roberts Bank) (Table 1). The corresponding salinities at these locations near the outflow of the Fraser River were more characteristic of a freshwater environment (salinity of 10‰), indicating that freshwater criteria are more appropriate in some situations.

Nickel and Cu were the only metals that exceeded sediment quality criteria at more than one location (Hg did exceed criteria values but only at one location). Concentrations of Ni at over half of the sampling locations were in exceedance of both the marine and freshwater sediment quality criteria, while Cu exceeded the freshwater criterion at greater than half of the locations. Brewer *et al.* (1998) found that chromium (Cr), Mn, Fe, Ni and Cu exceeded the provincial criteria for the protection of freshwater life at all up-river reference sites, indicating that background levels of these metals are naturally high (see Schreier *et al.* [1999] for a discussion of natural sources of Ni in the Sumas watershed). However, both Ni and Cu are also released from small industrial plants, municipal WWTPs and wood preservation facilities, indicating that considerable inputs from industrial and municipal activities plus high background levels are contributing to the elevated levels measured.

Compared to levels reported in the few previous studies that have measured metal concentrations in the Fraser River intertidal area (Table 2) (Bindra and Hall 1977; McGreer 1979; Swain and Walton 1990), levels of all metals measured in our study have decreased or remained constant, except for Cd which increased in Boundary Bay. On Sturgeon Bank, near the original sewage discharge area, Cu, Pb, Ag, Hg and Cd have decreased from 12 to 56 per cent during the period 1977–1995; Zn has remained fairly constant. In addition, total organic carbon has decreased by 69 per cent (Thomas 1997) which could explain part of the decrease in metal concentrations because metals can be incorporated or sorbed to organic matter. In addition, McGreer (1996, pers. comm.) found that burrowing rates of *Macoma* in 1995, were normal compared to 1977 when rates were almost zero. This can be taken as an indication that the sediments are no longer toxic.

While concentrations of some metals (Cu and Ni) were higher than the LOEL (lowest observable effects level) in sediment, metal concentrations (except for Cu) in the Fraser River intertidal area are equal to or lower than levels observed in other estuaries (Table 2).

The common clam (*Macoma*) is an important link between primary producers and fish and shorebirds in the intertidal area. It has been used as a biomonitor of metal contamination because it feeds directly on deposited sediments and acts as a bioaccumulator and integrator of temporal fluctuations of metals in the sediment. However, there is usually high variability in tissue levels, which is associated with factors, such as age, size, season and sediment composition. Metal concentrations in tissues were measured only once in previous studies and only at the sewage discharge site on Sturgeon Bank in 1979 (McGreer 1979). Comparing McGreer's values to those in 1996 (Thomas 1997) indicates that tissue concentrations in the clam have decreased by two-fold for Cu and Zn, and 10-fold for Hg (Table 2). Since Zn in the sediments has remained relatively constant, but tissue concentrations of Zn in the clam have decreased by two times, this suggests that there has been a decrease in the bioavailability of Zn during this 17-year period. Comparing

Table 2. Ranges of total metal concentrations (µg/g dry wt) in various estuarine surface sediments and in the tissues of Macoma balthica.

LOCATION/AUTHOR	ESTUARY	Cd	Cu	Ni	Pb	Zn	Hg
Sediment - Fraser River Estuary							
Thomas 1997	Sturgeon Bank	0.03–0.43	10.7–54.1	30.0–41.3	5.3–19.0	46.6–95.8	0.02–0.22
	Roberts Bank	0.04–0.18	17.3–52.8	33.7–44.7	3.2–12.3	14.3–100.8	0.02–0.09
	Boundary Bay	0.06–0.25	4.4–23.4	8.0–23.0	1.3–7.8	21.2–53.0	0.01–0.04
Swain & Walton 1990	Boundary Bay	<0.10–0.17	4.3–26.6	-	1.7–14.2	21.0–96.4	0.008–0.05
McGreer 1979	Sturgeon Bank	<0.4–3.0	12.4–234	28–49	3–166	41–264	0.03–0.89
Bindra & Hall 1977	Sturgeon Bank	-	21.8–302		41.7–219	67.3–301	-
Grieve & Fletcher 1976	Sturgeon & Roberts banks	-	17.2	43.4	5.1	52.7	-
Sediment - Fraser and Other Estuaries							
Thomas 1997	Fraser River, Canada	0.03–0.43	4.4–54.1	8.0–49.0	1.3–19.0	21.2–100.8	0.01–0.22
Luoma *et al.* 1990	Suisun Bay/Delta, USA	0.2–0.8	12.0–95.0	-	16.0–120.0	50.0–245.0	-
Bryan & Uysal 1978	Tamar, U.K.	<0.1–5.4	111–521	33–64'	112–697	195–1150	-
Macoma *Tissue - Fraser R. Estuary*							
Thomas 1997	Fraser River, Canada (average)	0.15–1.5 (0.65)	9.5–308.4 (84.8)	4.2–26.9 (12.9)	0.5–13.5 (2.8)	86–527 (287)	0.15–0.27 (0.27)
McGreer 1979	Sturgeon Bank (near Iona Island)	D.L.	49–314	-	D.L.	392–743	0.74–6.76
Macoma Tissue - British Estuaries							
Bryan *et al.* 1985	Tees	0.2	152	0.3	5	414	1.03
	Mersey	0.2	134	1.7	7	747	-
	East Looe (upper)	0.7	208	7.7	36	1164	0.97
	Loughor (mid)	0.2	32	1.6	2	396	0.12
	Severn	9.4	224	12.7	19	1510	-
	Dovey (mid)	0.9	33	2.6	17	771	-
	Solway	0.6	35	1.5	5	365	-

- = metal not measured
D.L. = detection limit

the present tissue concentrations in the clam from the Fraser River intertidal region to other estuaries reveals that Pb and Zn are low, Cd and Hg are average, Cu is average to high, and Ni is high (Table 2).

Sewage Tracers

One metal that deserves special attention is silver. It is typically the most enriched metal relative to Cu, Pb and Zn in domestic sewage waste. For instance, the enrichment factor for Ag in sewage sludge is 200, which is several times higher than the factors for Cu, Pb and Zn. Hence, in the absence of metal smelter discharges, Ag is an excellent tracer for sewage contamination of sediments. It comes primarily from the photographic industry's discharges to the sanitary sewers. At present there is a Ag-enriched zone that extends several km north of the present-day submerged outfall diffuser (Fig. 2). It is similar to the northward-flowing bottom currents and the pattern observed for a dye tracer study of the effluent plume discharged from the outfall diffusers (Hodgins 1988). High Ag values extend further to the north (6 km) than to the south (1 km) of the diffuser and the highest values (~700 ppb) are found about 1 km north of the outfall. As a reference point, Ag values in average shale are ~70 ppb and would represent a pristine area. In contrast, the highest surface Ag values were ~1,300 ppb on Sturgeon Bank (at A0), the site of the pre-1988 sewage discharge. Ag concentrations rapidly decrease seaward and southward along the intertidal area (Fig. 2); in the latter case, this is due to the marked increase in sandy sediments south of the sewage discharge site. The lowest Ag concentrations were found in the course-grained sediments of A12 (Gordon 1997).

Plotting the ratio of metals to aluminum (Al) in a sediment core and comparing the ratio to that observed in shale gives a relative scale of enrichment or contamination. Figure 3 shows that Ag, Cu, Pb and Zn are enriched in the 2-10 cm interval. In fact, at 4 cm Ag is ~3,200 ppb, about three times the surface concentration. The peaks in metal concentrations and ratios may indicate the zone of deposition from 1962–1988, before the sewage discharge was moved offshore. This indicates that the sediments relatively close to the surface are considerably contaminated with metals and these contaminated sediments could be released by dredging in the area.

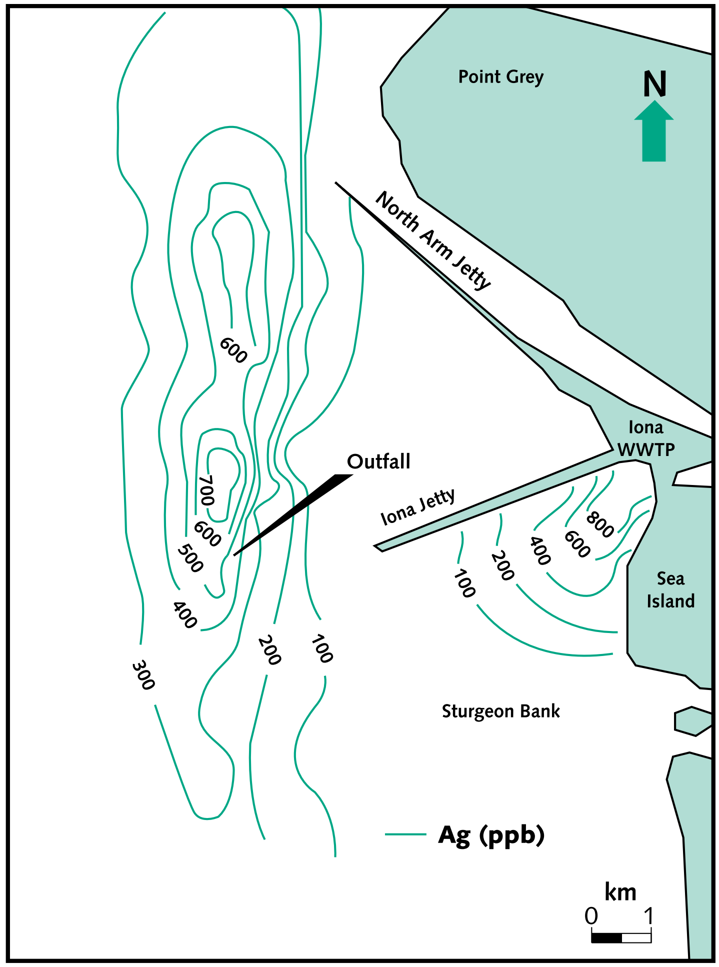

Figure 2. Distribution of Ag in surface sediments near the present Iona Island municipal WWTP deep-sea outfall and on Sturgeon Bank in the vicinity of the previously contaminated intertidal area (from Gordon 1997).

NUTRIENT DYNAMICS AND BENTHIC PRODUCTION

Nutrient Dynamics

In the Strait of Georgia and the Fraser River Estuary, nitrogen (N) concentrations may limit primary productivity (Harrison *et al.* 1983), in contrast to limitation by phosphorus in the river. Average daily nitrogen inputs to the Strait of Georgia include 1,500 tonnes from estuarine circulation which brings water of marine origin into the strait; 22 tonnes from sewage discharged in marine outfalls; 50 tonnes from the Fraser River (which includes N in sewage and in agriculture, urban and natural runoff) and 8 tonnes from atmospheric deposition (Mackas and Harrison 1997). While these values provide a general account of the annual sources of N loading to the offshore areas of the strait, the processes which control nitrogen uptake and recycling in any given season in the sand flats are as important and are discussed below.

Primary producers prefer to take up inorganic nitrogen sources such as nitrate (NO_3) and ammonium (NH_4), but some organic nitrogen compounds such as urea can also be utilized. When organic matter is deposited on or in the sediments, it is broken down (remineralized) by bacteria to NH_4, which is later oxidized to NO_3 by bacteria in the presence of dissolved oxygen. Thus the sediments usually have very high concentrations of NH_4 (several hundred times greater than the overlying water), which are usually correlated with the organic matter concentration of the sediments. Studies on nutrient cycling and dynamics showed that the concentrations of NO_3 and NH_4 in the water column on flood tides were lower than values on ebb tides, indicating that nutrients are released from the sediment during their immersion by sea water (Fig. 4). At the same time, phytoplankton chlorophyll *a* was higher on flood tides than on ebb tides, indicating phytoplankton biomass is lost to the sediment. Therefore, Sturgeon and Roberts banks represent an important nearshore nitrogen source for phytoplankton productivity in the Strait of Georgia, especially when ambient nitrogen in surface waters is low during summer, but represent a sink for phytoplankton production in the water column.

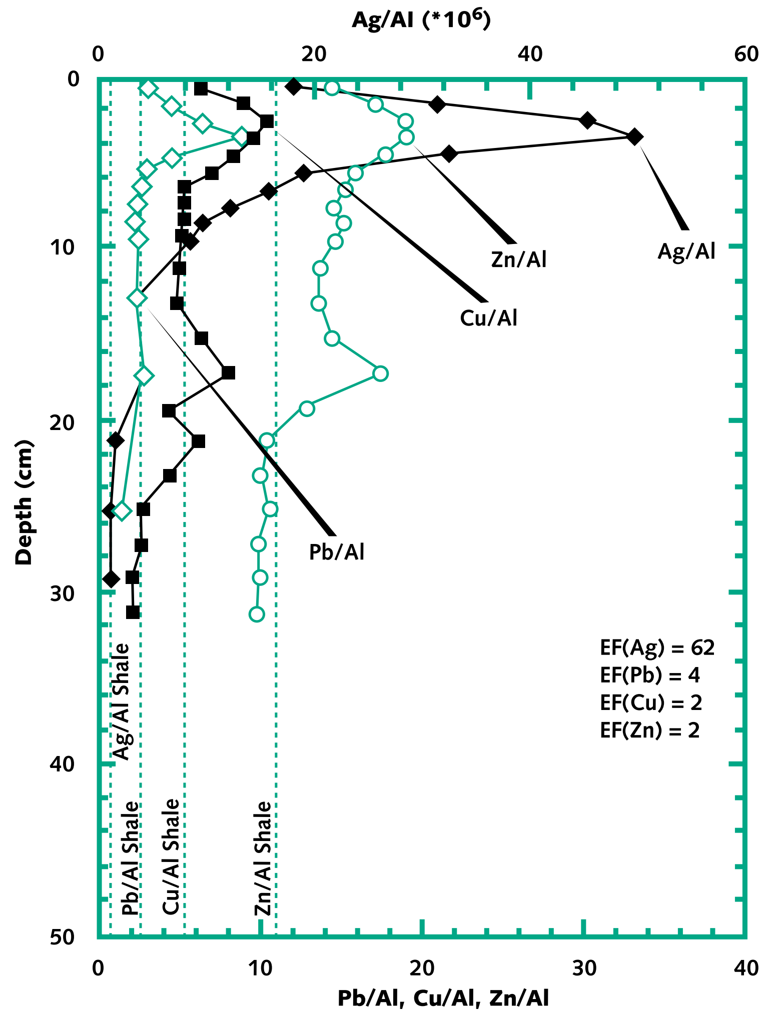

Figure 3. Vertical distribution of Ag/Al, Pb/Al, Cu/Al and Zn/Al in a 30 cm core from A0. The value of the ratios in average shale is indicated for comparison. Enrichment factors (EF) represent the maximum metal to Al ratio in the core normalized to the ratio in average shale (from Gordon 1997).

Pore water NH_4 concentrations in the top nine centimetres of sediment fluctuated between about 100 and 700 µM (1.4 and 9.8 mg NH_4-N/L) at A0 and between 20 and 700 µM (0.28 and 9.8 mg NH_4-N/L) at A14 (Fig. 5). In general, NH_4 concentrations in the pore waters were higher at A0 (near the previous sewage discharge site) than at A14 (on Roberts Bank). This is due to the higher organic matter content of the sediment at A0 and its subsequent decomposition and release of NH_4. In contrast, NO_3 concentrations in the pore waters were lower at A0 than at A14. Vertical profiles of pore water NH_4 at A0 showed various distribution patterns during the three-year study: an increase with depth or a decrease with depth, or a decrease in the top few centimetres and then an increase with depth. In contrast, at A14, the vertical distribution mostly showed a decrease with depth. Particulate organic nitrogen generally decreased with depth at both A0 and A14. These results indicate that particulate organic matter is deposited at the surface of the two banks and remineralized at the surface. Higher NH_4 concentration at the sediment surface suggests that NH_4 generated during remineralization could be released into the water column during flood tides, and it could also move downwards by diffusion because of the high concentration gradient. However, there is little accumulation of NH_4 in deeper pore waters and this suggests that denitrification (bacterial conversion of inorganic nitrogen to nitrogen gas) occurred in the sediments. The significance of denitrification to the nitrogen cycle on these tidal flats needs further study.

Chlorophyll *a* and Primary Productivity

The results show that the yearly average of benthic microalgal biomass, as measured by the amount of chlorophyll *a* (chl *a*), was lowest at A0 (84 mg chl *a*/m^2) and highest at A10 (2044 mg chl *a*/m^2), with A14 less than A12 and intermediate in biomass (235 and 882 mg chl *a*/m^2, respectively) (Fig. 6).

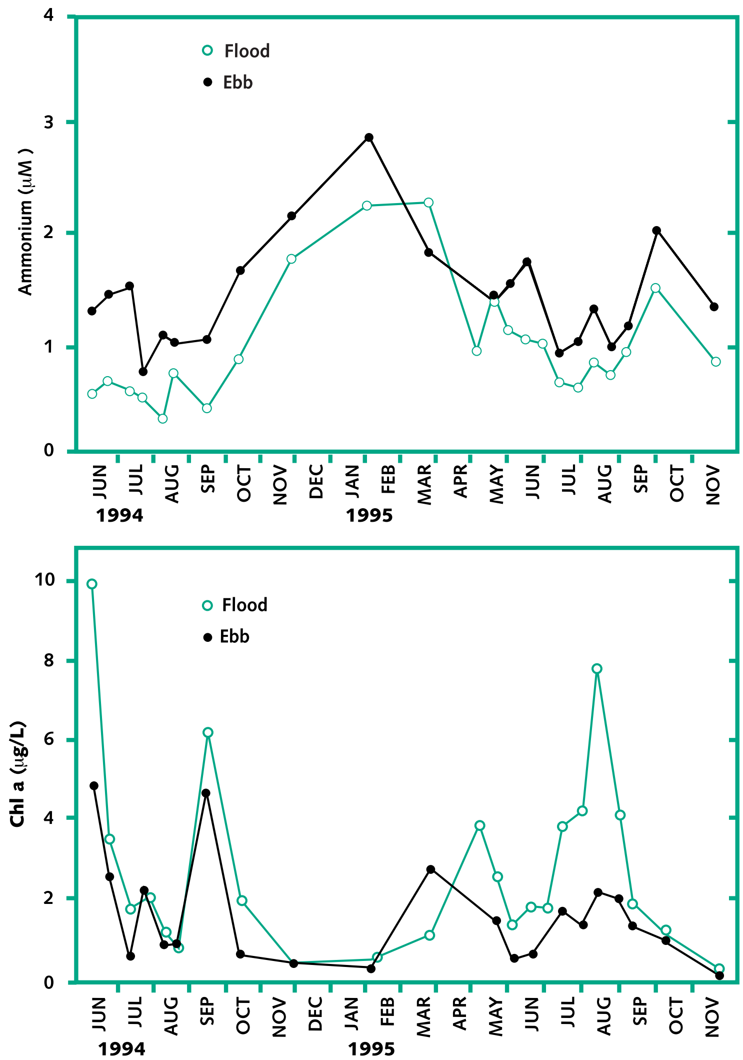

Figure 4. *Ammonium and chlorophyll* a *in the water column during flood and ebb tides in 1994 and 1995 (from Bendell-Young* et al. *in press).*

There are indications of the recovery from sewage pollution at A0 when benthic microalgal chl *a* biomass in this study is compared with biomass data from other studies at muddy sites on Sturgeon Bank. On the northern side of the Iona Jetty (see Fig. 1), Bawden *et al.* (1973) reported chl *a* values of 26–61 mg/m^2 in 1972; higher values (62.7 to 424 mg/m^2) were measured by Harrison (1981) in 1978–79. The latter higher values may be due to extracting chl *a* from a 5 cm sediment core instead of the top 1 cm extracted in the former case. Their sites were separated from the sewage discharge by the jetty and were not directly exposed to sewage effluents. A study conducted in 1974 by Otte and Levings (1975) showed very high chl *a* concentrations (up to 1,000 mg/m^2) at a site close to A0 and the algal biomass mainly consisted of dense blue-green algal mats. As blue-green algae commonly dominate the algal community in nutrient-rich lakes (Laws 1993) and estuaries (Jaworski 1990), the presence of blue-green algal mats was not surprising in this area close to the “freshwater” sewage discharge. The Otte and Levings (1975) study results agreed with those of another investigation in the same region and in the same year (BC Research 1975). The latter study observed the highest chl *a* concentration, 772 mg/m^2, at sites near the outfall and dominance by green algae. Chl *a* values in our study are comparable to Bawden *et al.*’s (1973) and Harrison’s (1981) values on the northern side of the Iona Jetty, but were lower than those measured in the other two studies (Otte and Levings 1975; BC Research 1975). These comparisons suggest that A0 has recovered to the level comparable to the sites on the other side of the Iona Jetty, which were at least 2 km farther offshore than A0 and may have been partially influenced by dispersed sewage effluent which drifted with tidal cycles. Another indication of the recovery is that the dominance by blue-green algae or green algae was not observed in our study; instead, diatoms were the main component of the benthic microalgae (Ross 1998).

Another sign of the recovery is that temporal fluctuations in benthic chl *a* biomass appeared to be parallel between A0 and A14 and between A10 and A12. The parallel fluctuations in chlorophyll between A0 and

A14 indicate that the two sites were subject to common environmental forcing, such as effects of the light period during tidal exposure and the influence of Fraser River discharge. This suggests that A0 has partially recovered to a level that responds to natural factors. However, in comparing our four sites, the lowest chl *a* level was found at A0. Whether this low chl *a* level indicates inhibition by contaminants (*e.g.* high levels of Cu, Ag, Hg or some organic chemicals, such as herbicides) or grazing pressure cannot be resolved with our data.

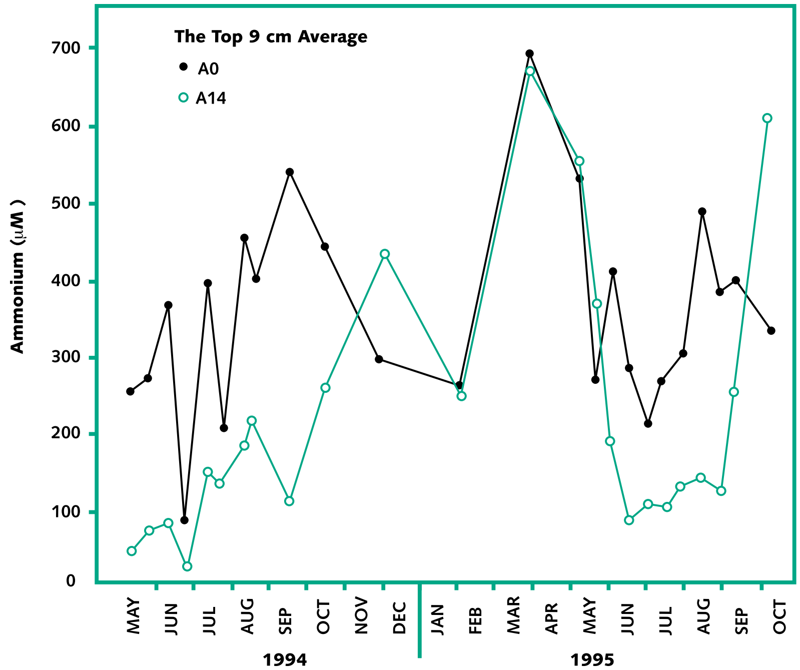

Figure 5. Average pore water ammonium concentration from a 9 cm sediment core at stations A0 and A14.

Chlorophyll *a* values in the sediment column at A10 and A12 were very high (1,000 to 3,000 mg/m^2), over one order of magnitude higher than those reported for other intertidal areas (*e.g.* ~300 mg/m^2 for Netarts Bay, Oregon [Davis and McIntire 1983] and ~120 mg/m^2 for North Inlet, S. Carolina [Pinckney and Zingmark 1993]). The vertical distribution of chl *a* showed large quantities of chlorophyll present below the surface of the sediment (up to 10 cm) and a low content of phaeopigments (initial breakdown products of chlorophyll), indicating that chlorophyll in the deeper sediments was recently buried. About 50 per cent of the chl *a* was in the top 1 cm at the muddier A0 and A14 sites, while only about 20 per cent of the chl *a* was in the top 1 cm at

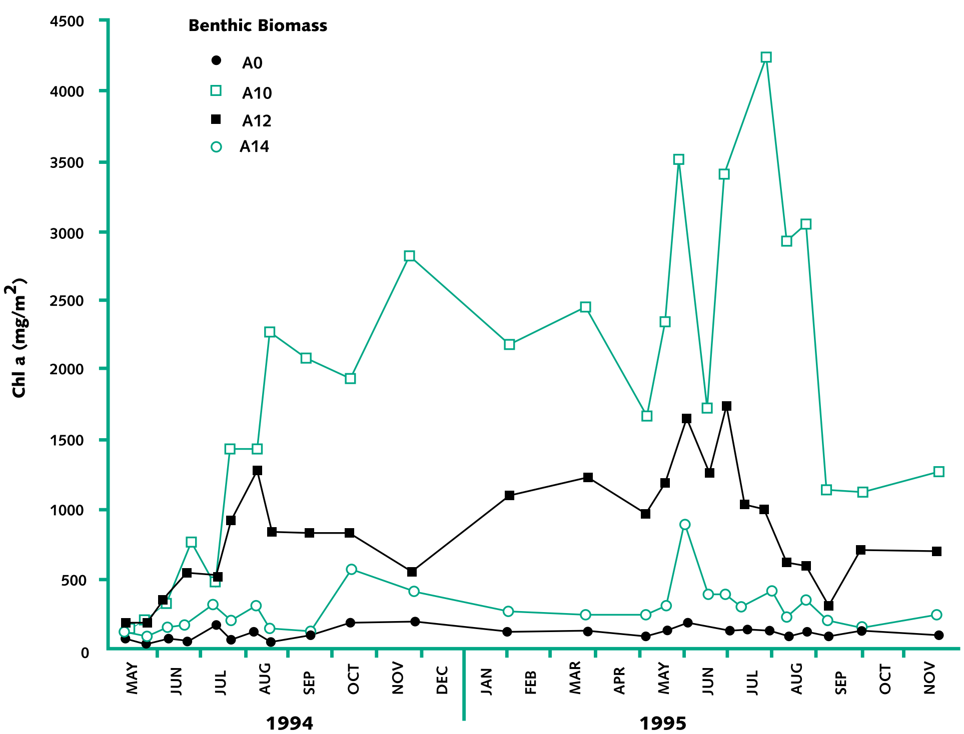

Figure 6. Seasonal changes in sediment chl a *at stations A0, A14, A10 and A12 (from Yin* et al. *submitted).*

the sandier A10 and A12 locations. Tidal drainage was most likely the major mechanism responsible for high chl *a* values at the two sandy sites; these sites experience tidal exposures at least once a day. The hypothesis is that microalgae grow in the benthic photic zone (usually the top 1–3 mm) and are drawn down through the large interstitial spaces of the well-sorted sandy sediment during drainage, which occurs during the ebbing tide. This process repeats over several daily tidal cycles and contributes to the accumulation of chl *a* in the deeper sediments. Large quantities of buried algal biomass indicated by these levels of chl *a* offer abundant food for benthic fauna and provide a source of NH_4 after remineralization by grazers and bacteria. In addition, it is important to stress the overall impact of the benthic algae reservoir in the sand flats for the whole delta-front secondary production: the muddy areas are not as rich in chl *a*, on a per square metre basis, and these muddy areas are only found in the upper intertidal zone.

Although sediment chl *a* (mainly benthic microalgal biomass) was lowest at A0, primary productivity per unit chl *a* was high, suggesting that perhaps grazing pressure was controlling the chl *a* standing crop more than inhibition by residual metals contamination in the sediments. The summer oxygen concentration in the water above the sediments at this site is near normal as opposed to the very low summer concentrations observed in the early 1980s (Birtwell *et al.* 1983). Reflecting these improved conditions, the top centimetre of the sediment column now exhibits the characteristic tan/brown colour of oxygenated sediments.

Secondary Production

Sturgeon Bank has exhibited several ecological changes stemming from the era of sewage deposition. These changes have included the alteration of population structure in dominant groups such as bivalves, amphipods, nematodes and polychaetes (Otte and Levings 1975; McGreer 1979), changes in sediment particle size (Grieve and Fletcher 1976) due to increased levels of organic carbon, and decreased oxygen concentration in the water overlaying the sediments (Birtwell *et al.* 1983). These were primarily due to the discharge of effluent onto the bank, compounded by physical habitat changes which caused the cessation of oxygen-rich and silt-laden freshwater flow from the North Arm over northern Sturgeon Bank. (The construction of the Iona to Sea Island causeway and the Iona Jetty breakwater also impaired effluent dispersion by sheltering the area from mixing during westerly and northwesterly winds, or by concentrating the plume against the jetty during southeasterlies.) Dissolved oxygen depletion occurred during the flood tide due to the high oxygen demand of the accumulated organic-laden sediment which was deposited when effluent was frequently pushed out of the channel and onto Sturgeon Bank by the heavier incoming seawater.

Benthic invertebrates in the area between the severely polluted zone and the contaminated zone include the polychaete worm *Manayunkia aestuarina* (Levings and Coustalin 1975; BC Research 1975), the polychaete *Eteone longa*, the common clam *Macoma balthica* (biomass up to 10.7 g/m^2 [Levings and Coustalin 1975]), the amphipods *Corophium salmonis* (biomass up to 2.9 g/m^2 [Levings and Coustalin 1975]) and *Corophium insidiosum,* and large numbers of harpaticoid copepods. The polychaete worm *Capitella capitata*, normally an indicator of organic pollution, occurred relatively infrequently when compared to other invertebrates in the area (approx. 2,200 individuals/m^2 at a distance 1,500 m southwest of the head of the effluent channel [BC Research 1975]). It was hypothesized that toxicity from heavy metals or chlorination, in combination with seasonal patterns of circulation, temperature, pH, dissolved oxygen and salinity, may have resulted in this distribution (BC Research 1975; Otte and Levings 1975).

Our objective was to monitor invertebrate production and population dynamics in the intertidal areas of Sturgeon and Roberts banks and to assess changes in these variables related to the cessation of sewage discharge onto the intertidal Sturgeon Bank. The analysis of invertebrate production, including measurements such as growth rates, secondary production and production to biomass ratios, is an important tool in estimating the availability of invertebrate food resources to their predators, and in assessing the health of an environment following an anthropogenic disturbance. Benthic invertebrates are better indicators of environmental quality than vertebrates (*e.g.* fish) because the post-larval stages of most invertebrates are relatively immobile and, therefore, must possess the inherent ability to survive in a contaminated location (Albright 1982).

Two suitable indicator species were selected based on previous research conducted by Levings and Coustalin (1975) at sites on Sturgeon and Roberts banks. The amphipod *Corophium salmonis*, and the clam *Macoma balthica* clearly fit within the guidelines of "moderate density" set forth by Pearson *et al.* (1983). Furthermore, these species form part of the basis of the food web for higher organisms in the area. For example, *C. salmonis* and *M. balthica* are common food sources of juvenile salmonids and flatfish (Cranford *et al.* 1985) and birds (Boates and Smith 1989).

When the sewage discharge was diverted in 1988, there were no animals living in the sediment at A0 (*i.e.* an azoic state). Several years later (Rebele 1994), various species of animals, such as the sewage tolerant polychaete *Manayunkia*, and *Corophium* and *Macoma*, began to recolonize this area. Another change that undoubtedly occurred was a reduction in organic matter through oxidation and the dispersal or dilution of the organic matter by sand or silt through sediment transport mechanisms. Such a reduction in sediment organic matter was even observed in the last three years (by about two-fold). This was attributed to a recent increase in sand content, and perhaps to the occurrence of less frequent sewage discharges onto the intertidal area during heavy rainstorms (due to the implementation of better control of stormwater flows in the sewerage system). Such sewage overflows are still occasionally discharged onto the old intertidal sewage channel during heavy rains or upsets at the treatment plant.

At the control station (A10), the density of *C. salmonis* was similar to that observed prior to the cessation of direct sewage discharge onto the intertidal zone (Arvai 1997). In terms of secondary productivity, values observed for *C. salmonis* (a maximum level of 0.78 g ash-free dry weight (afdw)/m^2/yr) in this study were lower than those observed for this and other species of the same genus in other research. The average productivity over the three years was similar at A0 and A14 (Fig. 7). At these two stations, which are still under the influence of past (A0) or present (A14) WWTP effluent disposal, high winter density, biomass, and productivity were observed. (Station A14 is exposed to WWTP discharges from Main Stem Fraser River WWTPs). It is likely that food availability (some in the form of organic matter

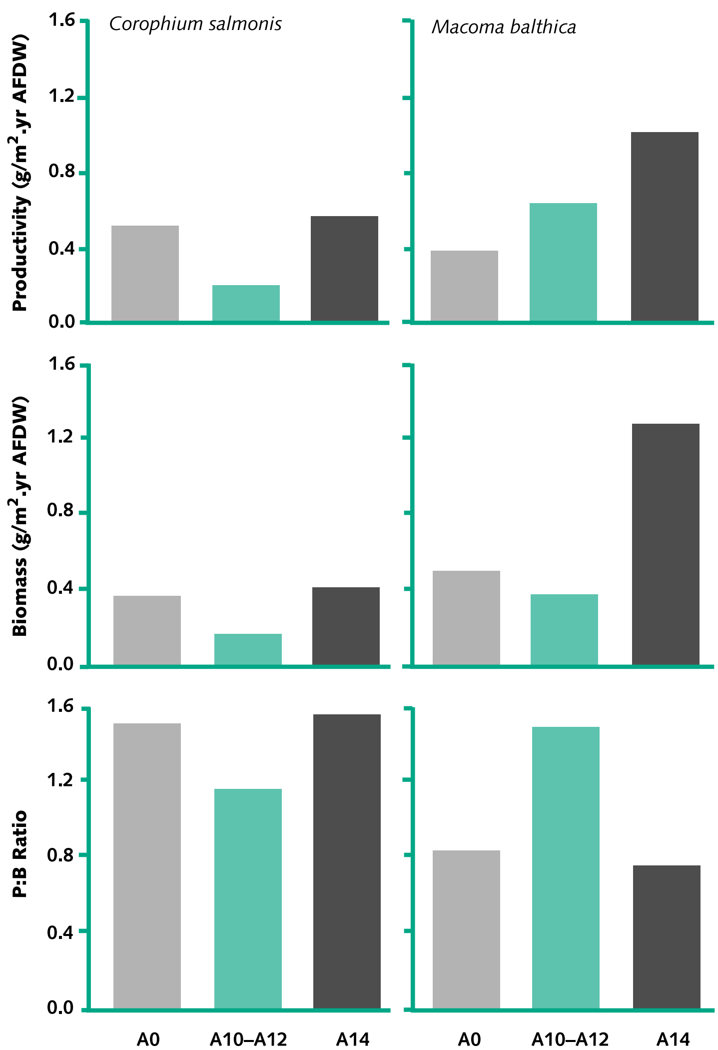

Figure 7. Three year averages of secondary productivity (g afdw/m^2/yr), mean biomass (g afdw/m^2/yr), and Productivity: Biomass (P:B) ratio for Macoma balthica *and* Corophium salmonis*. afdw=ash free dry weight (from Arvai 1997).*

from the WWTPs), longer potential feeding times and sediment-grain preference contribute to this finding at these stations. In general, densities observed at contaminated sites exceeded values found in previous studies at the same locations (BC Research 1975; Otte and Levings 1975).

Succession in the invertebrate community is complex, and can be affected by several interacting factors. For example, when sewage discharge is decreased, several factors such as the organic carbon, metal and nutrient contents in water and sediment change simultaneously and the importance of these individual factors may have different "weightings" in driving the biological community structure. For example, the preference of *Corophium* for fine grain sediments may be more important than a decrease in metal concentrations in the sediment (Arvai 1997). However, for station A0, the change from the original state of frequent anoxia was probably the most important factor influencing invertebrate recolonization of the area.

Secondary production of *M. balthica* (a maximum level of 1.9 g afdw/m^2/yr on Sturgeon and Roberts banks; Fig. 7 shows average of three years) was similar to values observed in other areas around the world (Beukema 1981; Cranford *et al.* 1985). Therefore, it was concluded that the *M. balthica* population and productivity have recovered dramatically from the previous azoic condition since effluent diversion. While most of the recovery is probably related to the reduction in contaminants and restoration of dissolved oxygen, some of the recolonization may be due to an increase in sand content or the decrease in organic matter. It is important to note, however, that in contrast to the situation with *Corophium*, both productivity and biomass values at A0 were less than half as high as those at A14, a site with similar sediment characteristics.

SHOREBIRD FEEDING ECOLOGY

Foraging conditions on the intertidal flats influence the time required for shorebirds to accumulate or replenish energy stores and this has a major influence on their survival and breeding success (Alerstam 1990), whether they are just passing through or spending the winter there. Until now, conservation has been hindered by a lack of understanding of intertidal food webs and the specific habitat and prey requirements of these birds.

In the Fraser River delta, two important shorebird species, Western Sandpiper and Dunlin, are potential indicators of coastal ecosystem health because the former uses the flats intensively in spring and late summer and the latter stays all winter.

Despite the extensive geographical range of Dunlin (*Calidris alpina*), populations of several subspecies of Dunlin worldwide (including our local subspecies, *pacifica*) appear to have declined in recent decades (Goss-Custard and Moser 1988; Paulson 1993). In Europe, studies on the wintering grounds have related these declines to human encroachment into wetland habitats and contaminated estuaries (Goede and De Bruin 1985a; 1985b). A possible explanation for their declining numbers in North America comes from Warnock and Gill (1996) who estimate the loss of Dunlin wintering habitat is between 30–91 per cent. Due to these concerns related to this species—35,000 to 60,000 of which overwinter in the Fraser River delta—Dunlin was selected for study in the Fraser delta. The behaviour and feeding of overwintering Dunlin were studied to evaluate this species' suitability as an indicator of ecosystem health of the intertidal flats and adjacent uplands (Shepherd 1997).

To understand what factors need to be considered when making inferences about the links between the health of the species and its food web, the feeding distribution in time and space was examined. By attaching radio transmitters to birds, information on site fidelity, habitat use, home ranges and activity budgets of male and female adult and juvenile Dunlin were obtained. Dunlin foraged both by day and night, as expected, but, surprisingly, they regularly fed at nearby agricultural fields at night. Males frequented these fields more than females. This was unexpected because a great deal more, and a greater diversity of, mudflat was available at night than during the day. Dunlin were rarely located in the fields during the day, likely due to the huge numbers of diurnal wintering raptors in the area. Hence, using Dunlin for monitoring toxic substances in the Fraser River delta would be complex since they forage in both marine and terrestrial areas and to differing degrees depending on their gender. This behaviour will add a layer of complexity to assessing the sources of contaminants in their diet. For example, contaminants in adult females would generally reflect uptake from the food web in the intertidal marine habitats, whereas males would reflect more of a mixture between marine and terrestrial sources.

This new information clouds the interpretation of the limited historical analyses of contaminants in Dunlin that were not separated into male and female samples. For example, in the late 1980s when the dioxin and furan discharges from upstream pulp mills were at their highest, a pooled sample of 18 Dunlin livers from both sexes was collected in December/January, 1989/90 (Whitehead 1997 pers. comm.). This analysis resulted in a concentration of 6.3 ng/kg of the most toxic dioxin congener, 2,3,7,8-tetrachlorodibenzo-*para*-dioxin, but it may have been higher in the females. Further, since all age classes were pooled, juveniles fresh from their arctic breeding grounds (where presumably exposure to these toxins is much lower) may also have lowered the concentrations. The eventual concentration accumulated by the end of the winter feeding would have likely been higher. However, as the levels of dioxins and furans in other wildlife (*e.g.* great blue heron) have declined dramatically (see Wilson *et al.* 1999), this threat should no longer exist for Dunlin.

In contrast to Dunlin, which reside on the delta for up to six months, Western Sandpiper (*Calidris mauri*) only utilize the delta tidal flats during spring and late summer migration. Predator/prey interactions of the Western Sandpiper were studied in order to determine the habitat requirements of a "typical" migratory shorebird and, also, the potential use of the species as a bioindicator of environmental health of the delta.

Western Sandpipers are one of the most common shorebirds in the western hemisphere. On the Pacific coast of North America, they migrate in large flocks between breeding grounds in Alaska and coastal wintering areas from California to Peru (Wilson 1994). The primary Pacific coast migration route is defined by a chain of critical stopover sites, including the Fraser River Estuary. Here, the northward migration is characterized by a peak in numbers from mid-April to late May (Butler *et al.* 1987), with maximum weekly numbers exceeding 500,000 shorebirds. On the Fraser River Estuary, the greatest overall densities of Western Sandpiper are found on Roberts Bank, west Boundary Bay and Sea Island (Butler 1994). In comparison, the southward migration is less intense and spread over a longer period, late June to early October, because of sex- and age-segregated movements by the shorebirds. In either migration, an individual might spend days to weeks on the delta.

Given their vast numbers and feeding demands, the expectation is that Western Sandpipers remove a high proportion of the available benthic production in the form of epibenthic and infaunal invertebrates from intertidal areas during their stay. Indeed, the arrival of migratory shorebirds on temperate mudflats in the Bay of Fundy has been associated with the decline in numbers of invertebrates (Daborn *et al.* 1993). In this context, the intense feeding pressure provided an opportunity to examine the link between the benthic community and at least one species of shorebird. This information will be useful in assessing the contaminant exposure risks this species and others at a similar trophic level might experience in the delta.

However, establishing the food web link quantitatively is difficult because of the spatial variability of invertebrates and difficulties in experimental design and analyses. Both factors have confounded quantitative interpretation and few studies have detected significant reductions in invertebrate numbers due to shorebird presence (Sewell 1996). Although Western Sandpipers are known to feed on invertebrates such as crustaceans, polychaete worms and clams living in or on the sediment, information on their natural diet is limited because of the difficulties in either observing what they eat or identifying prey remains in stomach contents. Taxonomic identification of the invertebrates and the lack of statistically valid sampling techniques are further problems. However, without a quantitative understanding of the distribution and abundance patterns of prey, coupled with an understanding of predator/prey relationships, actual prey and, thus, feeding habitat requirements cannot be ascertained. As a result, understanding the consequences of threats to habitat and subsequent provision of habitat conservation advice have been hindered.

The sandpiper study had two objectives. The first was to examine the spatial variability of invertebrate species across one area of the delta (Boundary Bay) as a prerequisite to designing sampling schemes to quantify prey density. A survey of macrofaunal invertebrates (>500 µm size) was conducted by collecting a series of 10x10 cm sediment cores across the intertidal flats of Boundary Bay. Between three and 10 invertebrate taxa per core were found with four numerically dominant taxa: gammarid amphipods (*Corophium* spp.), podocopid ostracods, polychaete worms (*Polydori ligni*) and epibenthic gastropod snails (*Batillaria zonalis*). For most taxa, there were significant differences between core densities at all scales except 1 km. With all taxa, there was considerable residual variation, suggesting patchiness at even smaller spatial scales than replicate cores collected 1 m apart.

The second objective consisted of a short-term exclosure experiment and random sampling (before/after predation) to assess reductions in invertebrate densities by Western Sandpipers at the Fraser River delta stopover during the spring migration. Surprisingly, there was little evidence for reductions in macroinvertebrate densities between the inside and outside of the exclosure cages even though several million Western Sandpipers had been feeding in the vicinity. In the exclosure experiment, a significant decline was observed only in a phyllodocid polychaete (*Eteone* spp.), but only at a site with little evidence of shorebird feeding. Thus, the decline could reflect the extreme patchiness in invertebrate distribution rather than shorebird predation. Results from the random sampling before and after shorebird migration revealed no differences that could be interpreted as being caused by shorebird foraging (Sewell 1996).

The trophic level(s) an animal feeds on and its connections as a prey itself to other trophic levels defines its place in the food web. The results of these studies have not elucidated the trophic position of Dunlin and sandpipers and highlight both the complexity of invertebrate spatial patterns and the uncertainties surrounding the feeding ecology of shorebirds. The apparent failure to detect the predatory “signature” of several million Western Sandpipers is enigmatic. One possible explanation is that the spatial variability of the macroinvertebrates is so high that the statistical power to detect effects is low (Sewell 1996). Another possibility is that Western Sandpipers may have been feeding on meiofaunal invertebrates and plants far smaller (<500 µm) than the macroinvertebrates that were assessed. Ongoing investigations utilizing a scanning electron microscope have revealed microstructures in shorebird bills that may act as sieves for meiofaunal prey. Other studies utilizing high-speed video technology have demonstrated that Western Sandpipers can indeed feed on meiofauna suspended in solution (Elner, unpub. results).

Overall, the research on the links between this species and the benthic food web has identified crucial gaps that need to be addressed before specific feeding habitat characteristics can be identified and mapped for conservation purposes. The prime question that remains is: from what prey do the Western Sandpiper and Dunlin, as well as other shorebirds, derive the bulk of their nutrition? If, indeed, meiofauna are the preferred prey, less biomagnification of contaminants by the intertidal-feeding shorebirds will occur than if the birds were feeding at a higher trophic level. Assessing contaminant accumulation in shorebird species that reside on the delta for extended periods and their prey is recommended to resolve this question.

CONCLUSIONS

While it is important to understand that the delta front comprises six fairly distinct habitats/zones differentiated by variable marine/freshwater ratio, sediment type and supply, several generalizations can be made about the status of this unique and internationally important ecosystem.

Sturgeon Bank is still influenced by the Iona Island municipal WWTP, although to a much lesser extent than before 1988, and by inputs from discharges to the North Arm upstream of the junction with the Middle Arm. Roberts Bank is directly influenced by the Main Arm and inputs from Lulu Island and Annacis Island municipal WWTPs. Boundary Bay reflects inputs delivered via the much smaller Nicomekl, Serpentine and Little Campbell rivers which carry an entirely different kind of contaminant mix. In general, metal concentrations were highest on Roberts Bank, and concentrations in Boundary Bay were much lower than those in Roberts and Sturgeon banks. However, the northern portion of Sturgeon Bank near Iona Island WWTP is still a contaminant hot spot, and it had the maximum occurrence of Cu, Cd, Pb, Zn, Ag and Hg. While greater than 50 per cent of each metal supplied by the Fraser River is in a bioavailable form, the iron generated by diagenesis in the pore waters of the estuarine sediments can reduce metals bioavailability and hence their bioaccumulation by organisms.

Nickel and copper exceeded provincial sediment quality criteria for the protection of freshwater or marine life at more than one location, and Hg exceeded the criterion near Iona Island WWTP.

On Sturgeon Bank, near the original sewage discharge area, Cu, Pb, Ag, Hg and Cd have decreased from 12 to 56 per cent during the period 1977–1995; Zn has remained approximately constant. The common clam *Macoma* was used as a biomonitor of metal contamination. Comparing 1979 with 1996 indicates that concentrations in the clam have decreased by two-fold for Cu and Zn and by 10-fold for Hg. Comparisons of *Macoma* tissue with those from other estuaries indicate that Pb and Zn are low, Hg and Cd are average, Cu is average to high, and Ni is high.

Silver proved to be an excellent tracer of the horizontal and vertical extent of contamination from the previous sewage discharge near the Iona Island WWTP. Vertical profiles of Ag in the sediments near Iona Island indicate that there are considerable deposits of contaminated sediments from about 2 to 10 cm below the cleaner sediment surface. These sediments should not be disturbed unless they can be removed without recontaminating a wider area. Thus, the proposal to reconnect McDonald Slough with the Strait of Georgia (*i.e.* dredge an opening through the artificial causeway) could cause significant trace metal contamination in adjacent areas or in dredge disposal zones.

Ammonium concentrations in the sediment pore waters at Iona Island were generally higher than at Roberts Bank (100 to 700 µM or 1.4 to 9.8 mg NH_4-N/L). This is due to the higher organic matter content of the sediment and the subsequent decomposition and release of ammonium. This supply of ammonium to the water column could help to maintain primary production through late summer when offshore levels of nitrate and ammonium are frequently near limiting concentrations for phytoplankton growth.

The yearly average benthic algal biomass was the lowest at A0—the site previously most impacted by the Iona Island municipal WWTP discharge—(84 mg chl *a*/m^2) and highest at A10—a control site (2,044 mg chl *a*/m^2)—with A14 (potentially affected by municipal WWTPs discharging to the Main Arm) and A12 (a control site) being intermediate in biomass. While the algal biomass at A0 is low, biomass has recovered to levels seen, in the past, at sites some distance from the previous sewage discharge channel. In addition, the previously observed blue-green algal mats have been replaced by benthic diatoms, which are present at lower biomass levels than the blue-green algae and are indicative of less nutrient-rich environments. Primary productivity per unit chl *a* was high at A0, suggesting that the metals in the sediments do not inhibit photosynthesis. The summer oxygen concentrations in the water above the sediments at A0 are now similar to other locations on the banks and the top centimetre of the sediment now exhibits the characteristic tan/brown colour of oxygenated sediments.

There were no animals living in the sediment at A0 when the sewage was diverted in 1988. Several years later, invertebrates such as the amphipod *Corophium*, the clam *Macoma* and the polychaete *Manayunkia* began to recolonize the area. This is likely due to an increase in the oxygen content of the sediment because of the decrease in the organic matter and a change from muddy to more sandy sediments.

As indicators of delta-foreshore ecosystem health and contamination, the use of Dunlin and Western Sandpiper was found to be somewhat problematic. Using Dunlin, which reside on the delta for up to six months, to monitor toxins in the Fraser River delta is complex since they forage in both marine and terrestrial areas (*i.e.* in nearby fields at night) depending on their gender. Contaminants in adult females, however, would generally reflect uptake from the intertidal food web. On the other hand, the Western Sandpiper, which utilize the delta tidal flats intensely during spring and late summer migrations, would possibly accumulate enough contaminants if sampled near the end of their spring migration stopover. Unfortunately, as meiofauna (*e.g.* nematodes and protozoans) are possibly the dominant food for this species, biomagnification of contaminants up the food chain would miss at least one magnification step and hence dampen the contaminant signal they could pick up in their short stay.

In conclusion, the delta foreshore is an ecosystem on the mend from about a century of contamination and habitat alteration. The sand flats are presently supporting large populations of shorebirds and waterfowl and the benthic food webs appear to be highly productive in comparison to other estuaries. However, this generality may not cover all areas as there are hot spots of past contamination. For example, a pollution episode that lasted for 26 years can still be detected a decade after it ceased. That area has now been recolonized by the former resident biota, and metals in the sediments have declined in concentration or their bioavailability. These responses should continue to improve over the next decade, especially as the frequency of combined sewer overflows at the outfall declines in response to the present program of sewer separation and management. Improvements should also be observed at the sites on Sturgeon and Roberts banks presently affected by contaminants originating in the Lulu and Annacis Island municipal WWTPs discharging to the Main Arm after secondary treatment is implemented fully in 1999. The improvements may be more subtle than those observed near the Iona discharge, however, as these sites were never exposed to the same intensity of pollution. The uncontaminated condition of Boundary Bay is noted but this could change as urban development intensifies in the Nicomekl and Serpentine River drainage basins in Surrey.

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