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# Fraser River Action Plan Resident Fish Contaminant and Health Assessment 

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## Summary

In this study for the Fraser River Action Plan (FRAP), a basin-wide survey of resident fish tissue contaminant levels was conducted along with an assessment of fish health. Adult peamouth chub (Mylocheilus caurinus) and adult mountain whitefish (Prosopium williamsoni) or juvenile starry flounder (Platichthys stellatus) were collected between July and November in 1994, 1995 and 1996, from up to 11 reaches in the Fraser basin. The Hansard, North Thompson, and Nechako reaches are located above all major effluent discharges and were chosen as reference reaches for this study. The Woodpecker, Marguerite, and Agassiz reaches on the Fraser River and the Thompson reach on the Thompson River are downstream of pulp mills and urban centres. Mission, Barnston, Main Arm, and North Arm are located furthest downstream in the highly urbanized and intensively agricultural Lower Fraser Valley.

Adult mountain whitefish were most abundant in the upper basin - in the Nechako River, Hansard reach, and North Thompson River - and were not present in the lower Fraser River downstream of Agassiz. Adult peamouth chub were distributed more evenly throughout the basin, with the highest catches in the estuary. Juvenile starry flounder were caught only in the estuary, since the species inhabits only marine and brackish waters.

In each reach, approximately 60 fish of each of two species were examined for external and internal abnormalities (necropsy), and their tissues - gill, liver, spleen, kidney, hindgut and pyloric caecae - were sampled for histological analyses. The necropsy was later converted to a numerical health assessment index (HAI). Activity of liver mixedfunction oxygenase (MFO) enzymes was measured, as ethoxyresorufin-o-deethylase (EROD) activity, on a sub-sample of these fish, as an indicator of exposure to certain organic contaminants. Contaminant analyses were conducted on composite liver, muscle, and bile samples, and included chlorophenolics, resin acids, organochlorine pesticides, PCBs, coplanar PCBs, dioxin/furans, PAHs, and metals.

Throughout the Fraser basin, dioxin and furan detections were dominated by 2,3,7,8tetrachlorodibenzofuran (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. Congeners in fish tissues from upstream of Hope reflected the dominance of pulp mill discharges in the upper basin. In the lower Fraser reaches, a wider array of dioxin and furan congeners was detected, due to the greater diversity of contaminant sources. TCDD was detected in 37 to $76 \%$ of mountain whitefish and peamouth chub tissues. Average within-reach concentrations were typically less than 1 $\mathrm{pg} / \mathrm{g} \mathrm{w} . \mathrm{w}$. for muscle and $2.4 \mathrm{pg} / \mathrm{g}$ w.w. for liver for both species. The highest levels were measured in the Thompson basin, both upstream and downstream of the pulp mill in Kamloops. Dioxin and furan levels have declined dramatically from the levels measured in 1988 because of recent process changes at the pulp mills.

Chlorophenolics are released from pulp mills, wood treatment facilities and municipal wastewater treatment facilities. Of the 47 chlorophenols and related compounds (for
example, guaiacols and catechols) analyzed, nearly all were at low concentrations (<1 $\mathrm{ng} / \mathrm{g} \mathrm{w} . \mathrm{w}$. ) or below detection in both muscle and liver. Higher chlorophenol concentrations were detected in bile downstream of pulp mill effluent sources. Two features are apparent in the patterns of chlorophenolics in bile. First, elevated concentrations indicate near-field, recent exposures, as was evident at Marguerite, where collection sites were as little as 20 kilometres downstream of the Quesnel mill effluent discharge. Second, total chlorophenolics concentrations were similar between the two reaches in the Thompson basin indicating that mountain whitefish, in particular, may be moving considerable distances over relatively short periods of time.

Resin acids are naturally occurring wood extractives that are released from cut timber, as from sawmills, chipping mills, and pulp and paper processing, and are also accumulated to high levels in bile. Resin acid concentrations in peamouth chub and mountain whitefish bile from reaches near sawmill operations (Hansard and North Thompson) were greatly elevated, indicating that sawmills are a significant source of these compounds in the basin.

Use of polychlorinated biphenyls (PCBs) has been severely restricted since the 1980s, but residues persist in the environment due to the resistance of the compounds to degradation. Measurements of a total of 84 of the 209 possible PCB congeners in fish tissues in 1994 and 1995 indicated that 1) contaminant sources (as indicated by congener patterns) do not appear to be localized to particular reaches, 2) concentrations seem to be similar throughout the basin, and 3) PCB concentrations and congener patterns are quite stable between years.

Use of persistent organochlorine pesticides has been banned or restricted throughout most of the world. Of the 24 pesticides determined in the analytical scan, only lindane (gamma-hexachlorocyclohexane) and endosulphan are still used in Canada, but many, including toxaphene and DDT, remain in use elsewhere in the world, particularly India, Central America and Russia. Consequently, sources in the Fraser basin are likely a combination of contamination from historical use, long-range transport in the atmosphere, and application of remaining stocks. In this study, concentrations of toxaphene and DDE were higher than other pesticides; both are highly stable, bioaccumulative and readily transported atmospherically. 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 found in the Agassiz reach. Inter-species differences in body burden were 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 that of mountain whitefish livers from the same sample reaches.

Although tissue levels of dioxins and furans, PCBs, and pesticides throughout the Fraser basin were well below the current tissue residue guidelines for consumption by humans, residues may still pose a risk to piscivorous wildlife. Dioxin and furan toxic equivalent quotients (TEQs), expressed in terms of recently-derived World Health

Organization 2,3,7,8-TCDD TEFs, far exceed the current CCME guideline level of $0.71 \mathrm{pg} / \mathrm{g}$ w.w. for protection of piscivorous wildlife in the Thompson basin. Residues of dioxin-like PCB congeners could further contribute to the total "dioxin" TEQs and exacerbate potential toxic effects. As well, toxaphene and DDE levels exceeded current CCME tissue residue guidelines for protection of wildlife.

Tissue concentrations of polycyclic aromatic hydrocarbons (PAHs) were measured only in peamouth chub tissues from the lower Fraser River reaches in 1994. PAHs are important indicators of urban and industrial contamination, and levels in the lower Fraser were highest in the North Arm, a reach that receives a high number of storm and industrial discharges.

Metals levels were below Canadian tissue residue guidelines for consumption by humans and within expected ranges for fish from uncontaminated sites in BC. Some levels of total selenium in mountain whitefish liver were slightly higher than the $3 \mu \mathrm{~g} / \mathrm{g}$ BC tissue residue guideline for the protection of human health, but are unlikely to be of concern to consumers.

EROD activity was induced downstream of urban centres and pulp mills in the Fraser basin, indicating exposure to organic contaminants. EROD induction in peamouth chub was highest in the estuary near the largest urban centre of Vancouver. Compared to reference reaches, up to 18 - and 78 -fold increases were seen in peamouth chub (Main Arm) and mountain whitefish (Woodpecker), respectively.

Fish condition indices, based on the length-weight relationship, energy reserves, and growth rates were used as indicators of the health of the fish. Peamouth chub condition indices, fat reserves and growth rates were all highest in the Nechako, Thompson and lower Fraser rivers, indicating better fish health in these reaches. Better fish health was associated with reaches having higher temperatures and lower sediment levels. Peamouth chub condition and growth rates were lowest at Marguerite in the central basin, a reach with both contaminant inputs from Prince George and Quesnel and high suspended sediment levels.

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, in 1995, at Agassiz compared to other Fraser reaches. Growth rates increased from north to south (upstream to downstream), indicating that latitude and altitude were the primary factors controlling growth. Mountain whitefish trends may be different from peamouth chub because mountain whitefish can migrate long distances and may not represent environmental factors at their capture sites. Comparison with mountain whitefish populations from other drainages, 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. However, genetic dimorphism in Fraser basin mountain whitefish populations, with some adults having a long, slender snout and thinner bodies ("pinnochios"), result in low condition indices which may bias estimates of fish health.

Relative gonad size (GSI) and age/size-at-maturity were measured as indicators of reproductive maturity and capacity. For both species, GSI were low and age/size-atmaturity were high in the central basin at Marguerite and in the Thompson River compared to upstream reaches, indicating impaired reproductive capacity. Apparent reproductive impairment may be caused by behaviour, habitat, and pulp mill and urban contaminant exposure from pulp mills and urban centres located immediately upstream.

Intersex, an indicator of endocrine disruption, was observed in the testes of starry flounder from the estuary. Because starry flounder were not sampled from reference sites, further study is needed to determined if the observed incidence is natural background for this species or was caused by endocrine disrupting compounds.

Elevation of the HAI has been linked to contaminant exposure and associated decreased growth and condition in other studies. However, 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, nor was the HAI associated with decreased condition or growth rates. 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. However, a high incidence of discoloured (highly pigmented) organs of both species downstream of Prince George and Quesnel at Woodpecker and Marguerite, respectively, may be associated with contaminant exposure.

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.

In conclusion, 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 low relative to existing environmental and human health guidelines, and have declined significantly from historical levels. Effects of contaminants on fish health in the Fraser basin could not be separated from effects of natural factors, because natural factors, such as flow, sediment loads and temperature, were highly variable over the large geographic area and the two to three years of the study.

## Sommaire

Dans le cadre du Plan d'action du Fraser (PAF), on a mené une étude à l'échelle du bassin pour connaître le niveau de contamination des tissus de poissons résidents, et on a évalué la santé des poissons. On a capturé des ménés deux-barres adultes
(Mylocheilus caurinus) et des ménominis des montagnes adultes (Prosopium williamsoni) ou des flets étoilés juvéniles (Platichthys stellatus) entre les mois de juillet et novembre en 1994, 1995 et 1996, dans des tronçons (jusqu'à 11) du bassin du Fraser. Les tronçons Hansard, North Thompson et Nechako, situés en amont de toutes les décharges majeures d'effluents, ont été choisis comme tronçons de référence dans cette étude. Les tronçons Woodpecker, Marguerite et Agassiz, sur le Fraser, et le tronçon Thompson sur la rivière Thompson, sont tous situés en aval d'usines de pâtes et papiers et de centres urbains. Les tronçons Mission, Barnston, bras principal et bras nord sont les plus loin en aval dans le bassin très urbanisé et soumis à une agriculture intensive de la vallée inférieure du Fraser.

Les ménominis des montagnes adultes étaient les plus abondants dans le bassin supérieur - dans les rivières Nechako et North Thompson, et dans le tronçon Hansard et n'étaient pas présents dans le bas Fraser en aval d'Agassiz. Les ménés deux-barres adultes étaient répartis plus uniformément sur tout le bassin, mais c'est dans l'estuaire qu'on en a capturé le plus grand nombre. Les flets étoilés juvéniles ont été capturés seulement dans l'estuaire, puisque cette espèce n'habite que les eaux marines et saumâtres.

Dans chaque tronçon, on a examiné environ 60 poissons pour chacune des deux espèces afin de déceler la présence d'anomalies internes et externes (nécropsie), et des échantillons de tissus- branchie, foie, rate, rein, intestin postérieur et caecums pyloriques - ont été prélevés pour l'analyse histologique. Les résultats de la nécropsie ont été convertis plus tard en un indice numérique d'évaluation de la santé (IÉS). On a mesuré l'activité enzymatique des oxygénases à fonction mixte (OFM) du foie, à partir de l'activité éthoxyrésorufine-O-déséthylase (EROD), dans un sous-échantillon de ces poissons, en tant qu'indicateurs de l'exposition à certains contaminants organiques. On a effectué des analyses de contaminants sur des échantillons composites de foie, de muscle et de bile pour y rechercher la présence de composés chlorophénoliques, d'acides résiniques, de pesticides organochlorés, de BPC, de BPC présentant une structure coplanaire, de dioxines et de furanes, de HAP et de métaux.

Dans l'ensemble du bassin du Fraser, les dioxines et les furanes détectés étaient dominés par le 2,3,7,8-tétrachlorodibenzofurane (TCDF), et, dans une moindre mesure, par la 2,3,7,8-tétrachlorodibenzo- $p$-dioxine (TCDD), qui est encore plus toxique; toutes deux sont des constituants caractéristiques des effluents des usines de pâtes et papiers. La présence de congénères dans les tissus des poissons en amont de Hope reflétait la dominance des déversements des usines de pâtes et papiers dans le bassin supérieur. Dans les tronçons du bas Fraser, on a décelé une gamme plus importante de congénères de dioxines et de furanes, ce qui est dû à la plus grande diversité des sources de contaminants. On a découvert des TCDD dans 37 à $76 \%$ des tissus du ménomini des
montagnes et du méné deux-barres. Les concentrations moyennes de contamination dans un même tronçon étaient généralement de moins de $1 \mathrm{pg} / \mathrm{g}$ (poids humide) pour les muscles et de $2,4 \mathrm{pg} / \mathrm{g}$ pour le foie, chez les deux espèces. On a retrouvé les concentrations de contaminants les plus élevées dans le bassin de la Thompson, autant en amont qu'en aval de l'usine de pâtes de Kamloops. Les modifications récentes apportées aux méthodes de production dans les usines de pâtes ont entraîné une réduction remarquable des concentrations de dioxines et de furanes par rapport aux niveaux mesurés en 1988.

Les usines de pâtes et papiers, les installations pour la préparation du bois et les stations de traitement des eaux usées municipales déversent des composés chlorophénoliques dans les cours d'eau. Parmi les 47 chlorophénols et autres composés apparentés (par exemple, les gaïacols et les pyrocatéchols) qui ont été mesurés, presque tous étaient en faible concentration ( $<1 \mathrm{ng} / \mathrm{g}$, poids humide) ou à un niveau non détectable, tant dans les muscles que dans le foie. Des concentrations plus élevées de chlorophénols ont été décelées dans la bile de poissons prélevés en aval des décharges d'effluents d'usines de pâtes. Deux caractéristiques ressortent de la présence de composés chlorophénoliques dans la bile. Tout d'abord, les concentrations élevées indiquent une exposition récente dans le champs proche, comme on l'a remarqué à Marguerite, où les sites de prélèvements étaient à 20 kilomètres seulement en aval de la décharge d'effluents de l'usine de Quesnel. Ensuite, les concentrations totales de composés chlorophénoliques étaient similaires pour les deux tronçons du bassin de la Thompson, ce qui indique que le ménomini des montagnes, tout particulièrement, parcourt des distances considérables sur une période relativement courte.

Les acides résiniques sont des extraits naturels du bois qui proviennent de coupes de bois, de scieries, d'usines de déchiquetage et de fabriques de pâtes et papiers; ils s'accumulent eux aussi à des niveaux élevés dans la bile. Les concentrations d'acides résiniques dans la bile du méné deux-barres et du ménomini des montagnes provenant de tronçons situés près de scieries (Hansard et North Thompson) étaient nettement élevées, ce qui indique que les scieries sont une source significative d'acides résiniques dans le bassin.

L'utilisation des biphényles polychlorés (BPC) est sévèrement réglementée depuis les années 80 , mais des résidus persistent dans l'environnement à cause de leur résistance à la dégradation. On a trouvé en 1994 et 1995, dans les tissus de poissons, un total de 84 congénères de BPC sur les 209 qui existent, ce qui indique que : 1) les sources de contaminants (comme le démontre la répartition des congénères) ne semblent pas être localisées dans des tronçons particuliers, 2) les concentrations paraissent similaires dans tout le bassin, et 3 ) les concentrations et la répartition des congénères de BPC sont assez stables d'année en année.

L'utilisation de pesticides organochlorés résis tants est prohibée ou réglementée dans la plus grande partie du monde entier. Parmi les 24 pesticides dont on a décelé la présence dans les analyses, seuls le lindane (gamma-hexachlorocyclohexane) et l'endosulphan sont encore employés au Canada. Toutefois, il y en a beaucoup, parmi lesquels le
toxaphène et les DDT, qui sont utilisés ailleurs sur la terre, particulièrement en Inde, en Amérique Centrale et en Russie. Ainsi, la présence des pesticides organochlorés persistants qu'on retrouve dans le bassin du Fraser est probablement le résultat d'une combinaison de contaminations dues aux utilisations antérieures, au transport atmosphérique à grande distance et à l'utilisation des stocks restants. Dans cette étude, on a également remarqué que les teneurs en toxaphène et en DDE étaient plus élevées que pour tout autre pesticide; la raison est que ces deux contaminants sont extrêmement stables, bioaccumulables et facilement transportables dans l'atmosphère. Les concentrations de toxaphène et de DDE dans le cours principal du Fraser augmentaient à partir des eaux d'amont et atteignaient les niveaux les plus élevés chez le méné deuxbarres et le ménomini des montagnes dans le tronçon Agassiz. Les différences interspécifiques dans la charge corporelle étaient probable ment liées principalement à la teneur en lipides des tissus, qui régit l'accumulation des composés organochlorés. L'effet de la teneur en lipides était particulièrement évident dans le foie des ménés deux-barres, dont le taux de lipides (12-30 \%) est généralement 2 à 6 fois plus élevé que celui du foie des ménominis des montagnes provenant des mêmes tronçons.

Même si les teneurs en dioxines et en furanes, en BPC et en pesticides dans les tissus, sur l'ensemble du fleuve Fraser, étaient bien au-dessous du seuil admis de résidus dans les tissus pour la consommation par les humains, les résidus peuvent tout de même poser un risque pour la faune piscivore. Les équivalents toxiques (ET) de dioxines et de furanes qu'on a calculés à partir des facteurs d'équivalence de la toxicité (FET) sur les 2,3,7,8-TCDD, récemment mis au point par l'Organisation mondiale de la santé, dépassent nettement, du bassin Thompson, le seuil de $0,71 \mathrm{pg} / \mathrm{g}$ (poids humide) établi dans les lignes directrices actuelles du CCME concernant la protection de la faune piscivore. Les résidus de congénères de BPC similaires aux dioxines pourraient accroître encore les ET des «dioxines totales » et exacerber les effets toxiques potentiels. De plus, les concentrations de toxaphène et de DDE dépassent le seuil établi dans les lignes directrices actuelles du CCME concernant les résidus dans les tissus et visant la protection de la faune.

C'est seulement chez le méné deux-barres provenant des tronçons du bas Fraser, en 1994, qu'on a mesuré les concentrations d'hydrocarbures aromatiques polycycliques (HAP) dans les tissus. Les HAP sont d'importants indicateurs de la contamination d'origine urbaine et industrielle, et les concentrations les plus élevées du bas Fraser se retrouvaient dans le bras nord, un tronçon qui reçoit de nombreux rejets d'eaux pluviales et d'effluents industriels.

Les concentrations de métaux se trouvaient au-dessous des seuils fixés pour la consommation humaine dans les lignes directrices canadiennes concernant les résidus dans les tissus, et étaient dans les normales pour des poissons provenant de sites non contaminés de la Colombie-Britannique. Certaines concentrations de sélénium total dans le foie des ménominis des montagnes étaient légèrement plus élevées que le seuil de $3 \mu \mathrm{~g} / \mathrm{g}$ fixé dans les lignes directrices de la Colombie-Britannique visant la protection de la santé humaine, mais elles ne devraient pas inquiéter les consommateurs.

On a observé l'induction de l'activité EROD chez les poissons prélevés en aval de centres urbains et d'usines de pâtes dans le bassin du Fraser, ce qui indique une exposition à des contaminants organiques. Chez le méné deux-barres, l'induction de l'EROD était à son plus haut niveau dans l'estuaire près du grand centre urbain de Vancouver. Les ménés deux-barres (bras principal) et les ménominis des montagnes (Woodpecker) présentaient une activité EROD 18 et 78 fois supérieure (respectivement) à celle mesurée chez les individus capturés dans les tronçons de référence.

Le coefficient de condition des poissons, basé sur la relation longueur-poids, les réserves énergétiques et le taux de croissance ont été utilisés comme indicateurs de l'état de santé des poissons. Le coefficient de condition, les réserves de gras et le taux de croissance du méné deux-barres étaient tous au maximum dans les rivières Nechako et Thompson ainsi que dans le bas Fraser, ce qui indique une meilleure état de santé pour les poissons de ces tronçons. L'état de santé des poissons est relié aux tronçons qui ont des températures plus élevées et un faible taux de sédimentation. Le coefficient de condition et le taux de croissance du méné deux-barres étaient les plus faibles à Marguerite, dans le bassin central, un tronçon qui a des apports de contaminants de Prince George et de Quesnel et un taux élevé de sédiments en suspension.

À l'opposé, le coefficient de condition et les réserves de lipides du ménomini des montagnes étaient généralement similaires dans les tronçons de la Nechako et du Fraser, et étaient maximales dans les tronçons du bassin Thompson. La condition des poissons était bonne à Woodpecker et, en 1995, à Agassiz, par rapport à celle des autres tronçons du Fraser. Les taux de croissance augmentaient du nord au sud (d'amont en aval), ce qui indique que la latitude et l'altitude étaient les facteurs principaux qui influençaient la croissance. Les tendances du ménomini des montagnes peuvent être différentes de celles du méné deux-barres, car les ménominis des montagnes se déplacent sur de longues distances, et ne représentent donc pas les facteurs environnementaux de leur site de capture. Si on compare les populations du ménomini des montagnes provenant d'autres cours d'eau à celles du bassin du Fraser, on découvre que ces derniers sont plus maigres que ce qui est optimal pour l'espèce, et qu'il y a peut-être des problèmes dans la nourriture et dans la relation trophique. Toutefois, le dimorphisme génétique dans les populations des ménominis des montagnes du bassin du Fraser (par exemple, certains adultes ont un museau fin et allongé et un corps mince - les «pinocchios») est responsable des faibles coefficients de condition, ce qui peut biaiser l'évaluation de l'état de santé des poissons.

La taille relative des gonades (indice gonado-somatique ou IGS) et le rapport âge/taille à la maturité ont été mesurés en tant qu'indicateurs de maturité et de capacité reproductives. Chez les deux espèces, l'IGS était faible et le rapport âge/taille à la maturité était élevé dans le bassin central, dans le tronçon Marguerite et dans la rivière Thompson, comparés à ceux dans les tronçons d'amont; cela indique une capacité reproductive réduite. La réduction apparente de la reproductivité pourrait être liée à des changements dans le comportement et à la dégradation des habitats ainsi qu'à
l'exposition aux contaminants d'usines de pâtes et de centres urbains provenant d'usines de pâtes et de centres urbains situés immédiatement en amont.

La présence de caractéristiques intersexuelles, qui est un indicateur de perturbation du système endocrinien, a été observée dans les testicules des flets étoilés provenant de l'estuaire. Puisque le flet étoilé n'a pas été prélevé dans des sites de référence, il faudra faire d'autres études pour déterminer si l'incidence observée est naturelle chez cette espèce ou si elle a été causée par des produits chimiques perturbateurs du système endocrinien.

Certaines études font le lien entre l'augmentation des indices d'évaluation de la santé (IÉS) et l'exposition aux contaminants, qui s'accompagne d'une réduction de la croissance et de la condition des poissons. Cependant, les IÉS n'ont pas augmenté en aval des centres urbains et des usines de pâtes dans le bassin du Fraser; ainsi, ils ne semblent liés ni à une exposition aux contaminants (activité EROD) ni à une réduction de la condition et des taux de croissance. En fait, les IÉS mesurés dans les tronçons de référence, où les concentrations de contaminants étaient faibles, étaient généralement supérieurs ou comparables à ceux des autres tronçons du bassin. On a découvert les IÉS les plus élevés dans la Nechako, ce qui était principalement causé par de graves infestations de parasites. Toutefois, la forte incidence de pigmentation prononcée des organes chez les deux espèces, observée en aval de Prince George et de Quesnel à Woodpecker et Marguerite, respectivement, peut être associée à une exposition aux contaminants.

Malgré les anomalies observées durant l'évaluation de la santé des poissons, un examen microscopique (histologique) des tissus indiquait que ceux-ci étaient généralement en bonne condition. Les anomalies histologiques, comme celles observées dans les IÉS, ne semblaient pas liées à une exposition aux contaminants, puisque les incidences étaient aussi élevées dans les sites de références que dans les sites d'aval. On a attribué la plupart des anomalies histologiques à des infestations parasitiques, qui sont très communes dans la Nechako.

En conclusion, le méné deux-barres et le ménomini des montagnes ont été utilisés avec succès comme poissons indicateurs d'exposition aux contaminants dans le bassin du Fraser - on les retrouvait partout dans le bassin et ils présentaient des différences dans les teneurs en contaminants et l'inductions des OFM entre les sites selon qu'ils se trouvaient en amont ou en aval des sources de contaminants. Les concentrations de métaux et de contaminants organochlorés mesurées dans les tissus étaient faibles par rapport aux seuils de contamination fixés dans les lignes directrices concernant la santé humaine et environnementale, et elles ont diminué de manière significative par rapport aux niveaux antérieurs. Les effets des contaminants sur la santé des poissons dans le bassin du Fraser ne pouvaient pas être séparés des effets de facteurs naturels parce que les facteurs naturels comme le débit de l'eau, les charges en sédiments et la température, étaient extrêmement variables dans la grande région géographique du bassin et pendant les deux à trois ans qu'a duré cette étude.

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## Introduction

The Fraser River Action Plan was initiated in 1991 to clean up pollution and restore the productivity of the Fraser River; the major cleanup goal was to reduce the discharge of pollutants, especially persistent toxic substances. The FRAP Environmental Quality Program aims to assess the effects of major sources of pollution and measure the success of cleanup efforts. A major goal of the Program was to assess the condition of the basin based on the contaminant levels, health, and community structure of selected indicators.

As one indicator of aquatic ecosystem condition, the FRAP Environmental Quality Program assessed the condition of resident fish, based on health and contaminant levels of mountain whitefish (Prosopium williamsoni), peamouth chub (Mylocheilus caurinus), and starry flounder (Platichthys stellatus). Previous fish studies in the Fraser Basin concentrated on the biology, habitat, and population sizes of commercially important species such as salmon and trout (Northcote and Burwash 1991). However, the health of resident fish populations is an important indicator of ecosystem condition because resident fish spend their entire life span in the river and better reflect local conditions than migrants such as salmon. FRAP provided a unique opportunity to conduct a basinwide survey of resident fish tissue contaminant levels coincident with an assessment of fish health.

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 polychlorinated biphenyls (PCBs) in fish tissue (Albright et al. 1975, Hall et al. 1991). Subsequently, monitoring for contaminants in fish tissue 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 run-off from lumber treatment facilities and effluent discharge from municipal waste water treatment plants (WWTPs) (Birch and Shaw 1995, Carey et al. 1988, Carey and Murthy 1988, Rogers et al. 1990, Rogers and Hall 1987, Rogers et al. 1992).

Information on contaminants in the upper basin has been collected more sporadically. Peterson et al. (1971) measured heavy metals in fish from lakes throughout BC and Derksen (1986) measured metals in rainbow trout and mountain whitefish in the Thompson River. In the 1980s, fears about chlorophenolics and 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 (Mah et al. 1989, Rogers and Mahood 1982, Rogers and Mahood 1983, Rogers et al. 1988, Servizi et al. 1993, Tuominen and Sekela 1992). Dioxin and furan surveys were repeated by the pulp mills from 1990 to 1992 (Dwernychuk et al. 1993).

Mountain whitefish were selected as an indicator species for this study because they are widely distributed and abundant in the Fraser basin, previous studies indicated that they accumulate dioxins and furans to higher levels than other species sampled (Dwernychuk et al. 1993, Mah et al. 1989, Pastershank and Muir 1995), and research is being conducted on their life history in the upper Fraser

River (McPhail and Troffe 1998). Peamouth chub are also widely distributed and abundant in the basin, were the 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 BC. Starry flounder is an estuarine species, which lives in contact with the sediments, a sink and source for contaminants. A previous study indicated that flounders do not migrate until over 17 cm in size (Nelson 1995), therefore effects on fish under this size would be the result of the local conditions.

The necropsy-based fish health assessment of Goede and Barton (Goede and Barton 1990) has been incorporated in this study because it is being widely applied in environmental studies (Adams et al. 1993, Hatfield Consultants Ltd. 1996a, Hatfield Consultants Ltd. 1996b, Healey 1997) and we wanted to test whether or not the health assessment abnormalities could be confirmed by histological analyses.

## Study Area

The Fraser River drains approximately one-quarter of the province of BC. The river arises in the Rocky Mountains at the Alberta-BC border near Jasper National Park, then flows north-west to Prince George, south through the arid Cariboo plateau, and west in southern BC through the Coast Mountains and Lower Fraser Valley into the Strait of Georgia. The Nechako and Thompson rivers, which are its major tributaries, flow into the Fraser River from the west at Prince George and the east in the mid-basin, respectively (Figure 1).

The Fraser basin supports one of the largest and most diverse salmon populations in the world and is a major part of the Pacific Flyway migratory bird route between South America and Siberia. The basin is also home to 2.5 million people (GVRD 1997) -- over 60\% of BC's population and approximately $80 \%$ of BC's economic production. The major urban centres in the basin are Prince George, Quesnel, Kamloops, and the Greater Vancouver Regional District. Fifty percent of BC's population resides in the Lower Fraser Valley.

There are eight pulp and paper mills which discharge into the Fraser basin: five located along a 150 km section of the river in the upper basin in Prince George and Quesnel, one on the Thompson River at Kamloops, and two on the lower river just upstream of Vancouver. These mills contribute more than half of the industrial effluent discharged into the Fraser basin. Consumption advisories were issued for mountain whitefish and largescale sucker in the Fraser, Quesnel, and Thompson rivers between 1989 and the early 1990s because of contamination with dioxins and furans. The advisories were lifted due to declining levels of dioxins and furans resulting from process changes in the pulp mills.

The hydrographs for four stations on the Fraser, Nechako, and Thompson rivers are shown in Figure 2 (Environment Canada 1997). The discharges at all stations are lowest in winter, when the northern reaches are under ice. Discharges increase in the spring during freshet, peak in summer, and decline through the fall. During this study, fish were sampled from the estuary between mid-July and mid-

August, during the peak discharge period. Sampling from upper reaches was conducted during declining flows in the fall. Flows during the fall of 1994 were lower than average at all reaches sampled (Figure 2).

Suspended sediment levels in the Fraser basin vary with discharge - sediment levels are high during high flows in the spring and early summer, and low during low flows in the winter. Long-term suspended sediment data are available only for the mainstem Fraser River (Environment Canada 1997) but suspended solids data are available for comparison among all reaches sampled for resident fish. Mean suspended solids concentrations at high (May-July) and low (January-March) flow at stations on the Fraser River and its tributaries (Hall et al. 1991) are presented in Table 1. Suspended solids concentrations are much higher at all sites on the Fraser mainstem than in any of the major tributaries.

Suspended sediment levels are highest at Marguerite. The percentage of silt and clay is highest (almost $100 \%$ ) during lowest discharges but falls to less than $40 \%$ at high discharges (Carson 1988). The clay fraction, which is most harmful to fish, generally accounts for about $20 \%$ of the silt-clay fraction, although no analysis has been done of the variation in this percentage.

Elevation gradients in the Fraser River mainstem are given in Table 2 - gradients are low above Prince George and below Hope, and high in the central basin.

Summer water temperature data collected at five of the Department of Fisheries and Oceans' continuous monitoring stations between 1994 and 1996 are presented in Figure 3. Water temperatures were lowest in the northern reaches at Shelley (Hansard reach) and the North Thompson River, intermediate in the middle Fraser mainstem (Marguerite reach), and highest at Hope (the lower Fraser River reaches) and in the Thompson and Nechako rivers. Over the three years, summer temperatures were highest in 1994 in the Nechako, Thompson, and North Thompson rivers, probably as a result of the low fall flows noted above. Water temperature data collected by the Federal-Provincial water quality monitoring program are also presented in Figure 4 to give a picture of the annual range of temperatures (Environment Canada 1997). Winter temperatures remain above zero in the Thompson River and the lower Fraser River (at Hope) where the rivers do not freeze over in winter. All other reaches are typically under ice for a period of several months during the winter.

## Resident Fish Life History

## Peamouth Chub

Peamouth chub are distributed through rivers and lakes of western Alberta, BC, Oregon, Washington, Idaho, and Montana in North America. In BC, peamouth chub are widespread -- in the upper Peace River basin, rivers on the Pacific slope from the Nass and Skeena rivers in the north to the Columbia and Fraser river basins in the south, and the Nanaimo River basin on Vancouver Island. Peamouth chub tolerate dilute sea water and are known to withstand brackish waters for a limited
time. Hence, the presence of peamouth chub in the Nanaimo River basin on the Vancouver Island has been attributed to a natural range extension (Clark and McInerney 1973).

Peamouth chub school in weedy shallows of lakes and rivers. They are most active during evening, night, and early morning hours. They spawn in shallow shore waters of lakes and rivers. Females are crowded by two or more males into an inch or two of water at the shoreline where the eggs and sperm are released. The grey-green, adhesive eggs settle to the bottom and become attached to stones and rubble. Spawning takes place from late spring to early summer. Water temperature at the time of spawning varies between $10.8^{\circ} \mathrm{C}$ and $22.2^{\circ} \mathrm{C}$ (Hill 1962, Shanbhogue 1976). Young-of-theyear peamouth chub are typically found in water less than 30 cm deep along rubble or gravel shores (Hill 1962). By age 3, most of the males mature ( $80.2 \%$ ), and most females mature at age 4 ( $97.7 \%$ ) (Hill 1962, Shanbhogue 1976).

In general, females grow faster than males and attain a larger size. Spawning females averaged 225 mm and breeding males averaged 198 mm in length (Schultz 1935). In Lake Washington, the sex ratio was equal for age 1 . Males were favoured at ages 2 to 3 . After age 3, a higher proportion of females were collected. Females were favoured when all age groups were combined (67.1\% females) (Shanbhogue 1976).

Peamouth chub feed on a wide variety of aquatic insects and their larvae, such as chironomids, mayflies, caddisflies, and some terrestrial insects. They also consume a variety of planktonic crustaceans, molluscs, and occasionally small fishes (Scott and Crossman 1973). In the Fraser River estuary, small peamouth chub ( $<75 \mathrm{~mm}$ ) feed mainly on chironomid larvae and crustaceans (Northcote et al. 1979). Large peamouth chub (>150 mm) feed on fish eggs, larvae, juveniles, and adults, and amphipods, invertebrates, molluscs, and insects. Prey abundance is one of many factors that determines the diet of fish, and hence, molluscs and crustaceans disappear from peamouth chub's diet as the chubs go further upstream from the estuary (Northcote et al. 1979). Peamouth chub feed predominantly at night during early spring and fall, and through most hours of the day during late spring and summer (Shanbhogue 1976).

Parasites are common in peamouth chub -- $94.6 \%$ of peamouth chub caught from 23 locations in BC were infested with parasites such as trematodes, nematodes, cestodes, acanthocephalans, and crustaceans (Bangham and Adams 1954).

## Mountain Whitefish

Mountain whitefish are distributed in lakes and larger streams only in western North America and are widespread in BC -- in the Liard, Peace, Columbia, and Fraser basins, and throughout the Pacific coastal drainages of the Bella Coola, Skeena, Nass, and Stikine basins (Scott and Crossman 1973). Its northern limit is the Liard River basin. They may inhabit small, turbid pools as well as cold, deep lakes, tending to frequent the upper 5 to 6 metres, seldom being found below 20 metres.

Spawning occurs in late fall or early winter over gravel or gravel and rubble. In BC, spawning occurs from October through February. Mountain whitefish are nocturnal spawners and the spawning activities begin at dusk. The water temperature at spawning is approximately $4^{\circ} \mathrm{C}$. Mountain whitefish eggs are adapted to a narrow temperature range during the incubation period -- $6^{\circ} \mathrm{C}$ is the upper optimum temperature for successful development and temperatures above $9^{\circ} \mathrm{C}$ result in disruption of egg development (Ford et al. 1992). The eggs are relatively large and, when water-hardened, average 3.7 mm in diameter. The number of eggs retained by females increases with increase in size; generally, the average number of eggs retained by a female is 5000 . No nest is constructed, and eggs incubate over the winter and hatch in early spring, at which time the emergent fry inhabit the margins of streams and backwaters downstream of spawning areas for several weeks. Whitefish then move to a favourable habitat and remain there through the mid-summer low flow period. Most mountain whitefish become sexually mature at age 3 or 4 , at which time, the mature adults may migrate to spawning habitats.

The growth rate of mountain whitefish varies with environmental factors, being slower with increase in altitude. The slow-growing populations of whitefish tend to live longer but still reach a maximum size similar to faster growing populations (Ford et al. 1992). The largest whitefish captured to date was more than 2 kilograms in weight and 572 mm in length. The range of average length observed for mountain whitefish caught from BC, Alberta, Utah, and California is 163 to 297 mm for age 3, 196 to 328 mm for age 4, and 221 to 330 mm for age 5 (Ford et al. 1992). There is no difference in growth rates between males and females (Scott and Crossman 1973).

Mountain whitefish feed primarily on benthic organisms, especially aquatic insect larvae, such as mayflies, stoneflies, caddisflies, and midges, and small molluscs and fishes. When bottom fauna are scarce, mountain whitefish will eat mid-water plankton and surface insects, and other fishes' and their own eggs. The mountain whitefish's diet in the lower Fraser River is composed mostly of insects (mainly chironomid larvae) and medium to large crustaceans (Northcote et al. 1979).

According to Bangham and Adams (1954), 84.2\% of mountain whitefish caught from 19 locations in BC were infested with parasites such as trematodes, cestodes, nematodes, acanthocephalans, crustaceans, and others.

A complete life history of mountain whitefish may be found in Ford et al. (1992).

## Starry Flounder

Starry flounder are distributed from southern California north to Alaska, west to the Bering, Chukchee, and Okhotsk seas, and south to central Honshu and Korea. They are found mainly in shallow waters but catches at depths greater than 275 m have been reported (Hart 1973). Starry flounder is a marine species able to tolerate low salinity. Young-of-the-year are commonly found in rivers where salinity is as low as 6 to 10 parts per thousand (Hart 1973). Starry flounder may have eyes and colour on either side -- 50 to $60 \%$ of flounders found from California to southeast Alaska are left-handed with the left-handedness increasing from south to north. Around the Alaskan
peninsula and Kodiak Island, approximately $68 \%$ of the flounders are left-handed, and in Japan, all are left-handed (Hart 1973).

Spawning takes place from February to April in Puget Sound, December to January in California (Hart 1973), and March to April in Fraser River estuary (Nelson 1995). They spawn in shallow coastal waters or tidal sloughs (Orcutt 1950). Spawning temperature is between 10.5 and $12.5^{\circ} \mathrm{C}$. Males mature at age 2 and females mature at age 3. The eggs are non-adhesive, and pale orange in colour. Egg diameter ranges from 0.89 to 0.94 mm and eggs are slightly lighter than sea water. Fecundity is estimated at $11,000,000$ for 565 mm (standard length) flounder (Orcutt 1950).

Larvae are about 2 mm in length on hatching, and are symmetrical. The newly hatched larvae initially float with their yolk sac facing up in the water and later turn around so that the yolk is facing the bottom. Young starry flounders ( 5 to 12 mm in length) eat mainly copepods and their nauplii, as well as barnacle larvae and Cladocera. They become asymmetrical at about 10.5 mm . They are demersal after the larval stage.

In the Fraser estuary, juvenile flounders ( $<100 \mathrm{~mm}$ ) feed primarily on medium to large crustaceans including mysids, amphipods, and isopods, and secondarily on insects, worms and small crustaceans. In the lower mainstem, dipteran insects made up most of their diet, followed by medium to large crustaceans and worms. Starry flounder adults ( $>100 \mathrm{~mm}$ ) consumed items similar to the juveniles with decreased numbers of small crustaceans. Molluscs and small fish also appeared in the diet of adults (Northcote et al. 1979).

Lengths-at-age for starry flounders found in the Fraser River estuary are: 91 mm at age $0+, 135 \mathrm{~mm}$ at age $1+, 223 \mathrm{~mm}$ at age $2+, 265 \mathrm{~mm}$ at age $3+, 230 \mathrm{~mm}$ at age $4+$, and 229 mm at age $5+$ (Nelson 1995). For male starry flounders in Monterey Bay, California (Orcutt 1950), lengths-at-age are approximately: 106 mm at age $1,235 \mathrm{~mm}$ at age $2,299 \mathrm{~mm}$ at age 3 , and 345 mm at age 4 . Females are longer than males at higher ages and they live longer.

Juvenile flounders, less than 200 mm , were sampled in this study because a tagging study conducted in the Fraser River estuary indicated that juvenile starry flounder are resident in the lower river (Nelson 1995).

## Methods

## Fish Capture

Adult mountain whitefish and peamouth chub and juvenile starry flounder were collected between July and November, in 1994, 1995, and 1996, from up to 11 reaches in the Fraser basin (Figure 1). The Nechako, Hansard, and North Thompson reaches are above all major effluent discharges on the Fraser and Thompson rivers, respectively, and were chosen as reference reaches for this study. The sampling periods are listed in Table 3.

Fish were captured using beach seining, gee traps, hoop nets, boat electroshocking, and bottom trawling as follows:

| Year | Location | Equipment |
| :---: | :---: | :---: |
| 1994 | Estuary | - Beach seines: 150-and 240- foot <br> - Bottom trawl <br> - Hoop net <br> - Mid-water trawl |
| 1994 | Upper reaches | - Beach seine: 30 m long by 4 m deep variable mesh, 7 mm mesh wings and 3.5 mm mesh bunt <br> - Boat electroshocker: Smith Root GPI 5.0 <br> - Large Vexar gee traps |
| 1994 | Middle and lower reaches | - Beach seine: 50 m long by 5 m deep variable mesh, 12.7 mm mesh wings and 9.5 mm mesh bunt |
| 1995 | Estuary | - Bottom trawl: mouth 3 m wide by 1 m deep, 6 m long, 20 mm mesh tapering to very fine meshed bunt |
| $\begin{aligned} & \hline 1995 \& \\ & 1996 \end{aligned}$ | All reaches | - Beach seine: 50 m long by 5 m deep variable mesh, 12.7 mm mesh wings and 9.5 mm mesh bunt |

Each reach was sampled until the target samples were obtained or until low capture success did not justify further effort. The target samples were 60 fish of each species in the size ranges: fork length 23 to 30 cm for mountain whitefish (upstream of estuary), fork length 18 to 25 cm for peamouth chub, and total length 15 to 20 cm for starry flounder (estuary). The numbers of fish collected for analyses in each reach are presented in Table 4. All fish species captured were identified and enumerated, and life history phase was noted as adult or juvenile. Target species of the desired sizes were placed into an on-board holding tank and transported to the mobile laboratory. All other fish were released.

Each site was identified by the name of the sample location, an alpha-numeric code representing the site, as well as the coordinates of the sample location obtained from a Global Positioning unit (GPS). The sites were then identified on a $1: 50,000$ scale topographical map.

Physical data collected at each sample site included water temperature, water velocity, substrate type, shore gradient, bank slope, hydraulic characteristics, and river bank characteristics. In addition, incidental observations regarding weather conditions were recorded at various sites. Water temperature was measured with a pre-calibrated hand-held thermometer. Dominant substrate was visually estimated in order of abundance as fines, gravel, cobble, boulder or bedrock. Water velocity was estimated as slow ( $0-0.5 \mathrm{~m} / \mathrm{s}$ ), moderate $(0.5-1.0 \mathrm{~m} / \mathrm{s})$, or fast $(1.0-1.5 \mathrm{~m} / \mathrm{s})$ and shore gradient and bank slope were either steep, moderate, or low.

## Mobile Field Laboratory - General Procedures

A mobile field laboratory was set-up on the shore near the sampling sites. Fish were held at the shore at these sites for a maximum of 12 hours in either mesh cages hanging in the river (estuary in 1994, only) or polyethylene tubs and aluminum tanks with pumps supplying a continuous flow of river water.

Immediately prior to processing, individual fish were dip-netted from holding pens or tanks and carried to the mobile lab in a stainless steel bucket of river water. In 1994 and 1995, fish were sacrificed by concussion. In 1996, live fish were anaesthetised using $0.01 \%$ phenoxyethanol before examination.

Fork lengths were measured for peamouth chub and mountain whitefish and total lengths were measured for starry flounder. The fish were weighed; the organs were removed, weighed, and preserved; then the gutted fish were re-weighed. Fish bodies and organs were weighed using K-tron balances -- models KS-1 and KS-10, respectively. After dissection, the head was removed, and the carcass was tagged with the fish number and placed in a stainless steel tray on dry ice. Otoliths were dissected from the head for later age determination.

The field laboratory crew consisted of four persons:

1. One person ("externalist") outside the mobile lab to collect fish from the holding tanks, perform the health assessment on external tissues, sacrifice the fish, preserve samples, and dissect otoliths.
2. One person ("recorder") inside the mobile lab to record all the data on field sheets.
3. Two "dissectors" inside the mobile laboratory to dissect sample tissues and assess internal fish tissues.

More details on field procedures are presented in the following sections that describe specific analyses.

## Sampling for Tissue Contaminants

Care was taken to ensure that samples were not contaminated during dissection at the field laboratory. After the fish were killed, staff handled fish with polyethylene-gloved hands and all materials contacting the external surface of the fish were washed with laboratory soap and rinsed with deionized water (DI). Dissections were completed on Teflon boards, and stainless steel dissecting tools were used for all internal contact. Fish bodies were frozen in stainless steel trays (wrapped in aluminum foil and plastic), gall bladders in glass bottles with aluminum- and Teflon-lined lids and livers in Teflon vials. All dissecting equipment and sampling containers were washed and rinsed as indicated in the table below.

Gutted fish, livers, and gall bladders were kept frozen on dry ice until they were received by the analytical laboratory. Samples remained at temperatures below $-15^{\circ} \mathrm{C}$ awaiting further preparation.

At the laboratory, fish were thawed and sorted by fish number. Skinless fillets of dorsal muscle were combined into composite homogenates of five randomly-chosen individuals of one species from one reach. In an effort to provide a true composite sample, each fillet was weighed, and aliquots drawn from each in proportion to the smallest fillet. Liver samples were combined as homogenates of up to 30 fish in order to provide sufficient tissue for the range of contaminant analyses. Each bile composite was prepared by choosing either five (lower Fraser in 1994) or approximately 30 usable bile samples
(gall bladder intact, full), and combined using a 100 mL stainless steel/glass barrel syringe which was cleaned with dichloromethane and methanol between uses. For the lower Fraser in 1994, the smallest volume of each five determined the aliquot taken so that equal portions from each gall bladder were combined into the final composite. For the remainder of the samples, all the bile was used because of the small sizes of the gall bladders.

| Item | Cleaning Sequence |
| :--- | :--- |
| Dissecting tools and boards (cleaned at <br> field lab) | -Lab soap wash in river or tap water. <br> -DI rinse, 3 times. <br> -Acetone rinse, 3 times. <br> -Hexane rinse, 4 times. |
| Amber glass bottles for gall bladder <br> samples for analysis of bile for organic <br> contaminants | -Lab soap wash and DI rinse. <br> -Oven bake for 6 hours at 325 <br>  <br> -Caps lined with oven-baked aluminum foil. |
| Glass bottles with Teflon liners or Teflon <br> vials for liver samples for analysis of <br> organic contaminants and trace metals | -Lab soap wash and DI rinse. <br> -Acid wash. <br> -DI rinse. <br> -Acetone rinse, 3 times. <br> -Hexane rinse, 4 times. |
| Stainless steel trays for fish bodies for <br> analysis of organic contaminants and <br> trace metals | -Lab soap wash and DI rinse. <br> -Acetone rinse, 3 times. <br> -Hexane rinse, 4 times. |

Muscle and liver composite samples were analyzed for a range of organic and metal variables. Axys Analytical Ltd., Sidney, B.C., determined concentrations of dioxins, furans, coplanar PCBs (IUPAC Numbers 77, 126, 169), PCB congeners, chlorophenolics, resin acids and polycyclic aromatic hydrocarbons. Levels of heavy metal contaminants in the tissues were measured by Quanta Trace for the lower Fraser sampling in 1994 and by the National Laboratory for Environmental Testing, Burlington, Ontario for the remainder of the samples.

Contaminant analyses conducted on fish tissues from sites in the Fraser basin in 1994 and 1995 are summarized in Table 5.

## Chlorinated Dioxin and Furan Analyses

Dioxin analyses followed closely methods outlined in Environment Canada (1992). Tissue samples (roughly 10 g muscle, $3-5 \mathrm{~g}$ liver) were first spiked with a surrogate standard solution of nine ${ }^{13} \mathrm{C}$ labelled dioxin and furan congeners. Each wet tissue sample was ground to a powder with $\mathrm{Na}_{2} \mathrm{SO}_{4}$ and extracted by column elution with 1:1 dichloromethane:hexane. A small subsample was removed for gravimetric lipid determination, prior to treatment of the extract by gel permeation chromatography for removal of dissolved lipids. Sample clean-up used a sequence of four columns (silica, alumina, carbon, alumina).

Analyses were conducted using a VG AutoSpec Ultima mass spectrometer equipped with a HewlettPackard 5890 GC, 60 m DB-5 chromatography column ( 0.25 mm i.d. x $0.1 \mu \mathrm{~m}$ film thickness) and a CTC autosampler and operated in single ion mode. Two ions were monitored for each group of isomers, two ions were used to monitor each of the ${ }^{13} \mathrm{C}$ - labelled surrogates, and five additional ions were monitored to check for interferences. Once identified, compounds were quantified using the internal standard method.

## Chlorophenolic Analyses

Tissue samples (approximately 10 g muscle, 5 g liver) were first spiked with a mixture of twelve ${ }^{13} \mathrm{C}$ labeled chlorophenolic compounds, then ground with $\mathrm{Na}_{2} \mathrm{SO}_{4}$ to a free-running powder. The sample was then transferred to a chromatographic column and extracted with 1:1 diethyl ether:dichloromethane. The eluate was concentrated, cleaned up using gel permeation chromatography, and the chlorophenolics converted to their acetate forms by reaction with acetic anhydride. The sample was back-extracted into hexane, loaded onto a silica gel column and eluted sequentially with hexane, toluene:hexane and finally isopropanol:toluene. An extract recovery standard solution was added prior to analysis by gas chromatography-mass spectrometry (GC/MS).

Bile samples were sonicated and hydrolyzed to release conjugated compounds prior to extraction.
Analyses were conducted using a Finnigan Incos 50 spectrometer with a Varian 3400 gas chromatograph, CTC autosampler and DG 10 data system. Chromatographic separation was achieved using a Restek ${ }_{x}-5$ column ( $30 \mathrm{~m}, 0.25 \mathrm{~mm}$ i.d., $0.25 \mu \mathrm{~m}$ film thickness). The mass spectrometer was operated in EI mode ( 70 Ev ) using multiple ion detection, acquiring two characteristic ions for each target analyte. The GC retention time and response factors were determined using a mixed calibration standard containing all target and surrogate compounds, derivatized to their acetate form. Compounds were quantified by integrated peak area relative to the peak area of the surrogate standard.

## Polychlorinated Biphenyls (Coplanar and Non-Ortho Congeners), Pesticides, and Toxaphene Analyses

Tissue samples (roughly 10 g for muscle, 3-5 g for liver) were spiked with a solution of ${ }^{13} \mathrm{C}$-labelled surrogates, then ground to a free-flowing powder with $\mathrm{Na}_{2} \mathrm{SO}_{4}$. The mixture was transferred to a glass chromatographic column containing 1:1 dichloromethane: hexane and eluted with additional solvent. The eluate was collected, concentrated and cleaned up on a calibrated gel permeation column (Biobeads SX-3) with 1:1 dichloromethane:hexane. This eluent was applied to a Florisil column, and eluted with hexane (F1 eluate) followed by 15:85 dichloromethane:hexane (F2 eluate) and $1: 1$ dichloromethane:hexane (F3 eluate). The F1 and F2 eluate were combined, then split gravimetically, with one portion being used for determination of coplanar PCBs and the remainder being used for analysis of PCB congeners, pesticides, and toxaphene. The F3 eluate was using for analysis of the most polar chlorinated pesticides.

The most polar chlorinated pesticides (eluate F3) were determined by high resolution GC with electron capture detector using a Hewlett-Packard 5890 gas chromatograph with a 60 metre ( 0.25 mm i.d., $0.10 \mu \mathrm{~m}$ film thickness) DB5 Durabond fused silica capillary column, ${ }^{63} \mathrm{Ni}$ electron capture detector, and a CTC autosampler with injection volumes from 1-100 $\mu \mathrm{L}$.

## Moisture and Lipid

A separate subsample of tissue ( $1-2 \mathrm{~g}$ muscle, 0.2 g liver) was weighed into a tared glass petri dish and dried to a constant weight at $105^{\circ} \mathrm{C}$ to determine moisture content.

Percent lipid determinations were made during either the dioxin/furan or PCB analyses. After tissue samples were ground to a powder with $\mathrm{Na}_{2} \mathrm{SO}_{4}$ and extracted by column elution with 1:1 dichloromethane:hexane, a small subsample was removed for gravimetric lipid determination, prior to treatment of the extract by gel permeation chromatography for removal of dissolved lipids.

## Polycyclic Aromatic Hydrocarbons (PAHs) Analyses

Tissue samples (about 10 g muscle tissue, $3-5 \mathrm{~g}$ liver tissue) were spiked with a surrogate standard solution and digested by refluxing in a methanolic KOH solution. The digest was transferred to a separatory funnel and extracted 3 times with pentane. The pentane layers were combined and dried over anhydrous $\mathrm{Na}_{2} \mathrm{SO}_{4}$, reduced and loaded onto a silica gel column (Biosil) for clean-up. The column was eluted with pentane followed by dichloromethane. The second eluate (containing the PAHs) was concentrated, transferred to a microvial and spiked with a recovery solution (containing perdeuterated homologues of acenaphthylene, fluoranthene and benz(a)fluoranthene) prior to analysis by GC/MS.

Analyses used a Finnigan Incos 50 spectrometer with a Varian 3400 gas chromatograph, CTC autosampler, DG 10 data system, and a Restek $\mathrm{k}_{\mathrm{x}}-5$ column ( $30 \mathrm{~m}, 0.25 \mathrm{~mm}$ i.d., $0.25 \mu \mathrm{~m}$ film thickness). The mass spectrometer was operated in EI mode ( 70 Ev ) using multiple ion detection, acquiring two characteristic ions for each target analyte and surrogate standard. Concentrations were calculated by comparison to internal standards using the mean relative response factors (RRFs) determined from calibration runs made before and after each batch of samples. RRFs had to agree within $\pm 10 \%$.

## PAH Metabolites in Bile Analyses

Concentrations of PAH metabolites in composite bile samples were determined by fluorescence detection after chromatographic separation by High Pressure Liquid Chromatograhy (HPLC) (Krahn et al. 1986).

Bile density was determined gravimetrically. Samples ( $10 \mu \mathrm{~L}$ volume) were injected into a LBK Pharmacia HPLC system consisting of a gradient pump, LBK Solvent Conditioner System, an LBK variable volume autosampler, and LBK 2215 integrator and LBK Fluorescence detector equipped
with respective pairs of UV filters for benz(a)pyrene, naphthalene and phenanthrene. Samples were analyzed in batches of 6 , bracketed with injections of standard solutions. Each sample was run once for each of the 3 wavelength pairs. Response factors, as determined by the ratio of the nanograms of standard injected per peak area, were monitored to measure instrument stability and calculate metabolite equivalents. Peak areas for calculation of PAH metabolite concentrations are the sum of all peaks within the select retention-time windows.

## Resin Acids in Bile Analyses

Samples were placed into a silanized centrifuge tube to which was added 0.5 M KOH in ethanol and an aliquot of surrogate standard (o-methylpodocarpic acid) and the sample pH adjusted to 11 . After hydrolysis for 3 hours at $70^{\circ} \mathrm{C}$, the sample was transferred to an Erlenmeyer flask and hydroxyamine hydrochloride added to the cooled solution. The pH was adjusted to 5 , and the sample was twice extracted with 20:80 diethyl ether:hexane. The collected organic layers were dried over anhydrous sodium sulphate and reduced to a small volume using a roto-evaporator. The extract was taken just to dryness under nitrogen, and solvent exchanged with methanol.

The sample was derivatized by addition of freshly generated diazomethane and allowed to react 30 minutes in a capped centrifuge tube. The extract was again reduced to dryness and exchanged to hexane.

Clean-up was on a basic silica column, eluted with hexane (discarded), followed by 5\% diethyl ether in hexane (collected). The extract was reduced to a small volume under a nitrogen blowdown. A aliquot of recovery surrogate ( $\mathrm{d}^{12}$-chrysene) was added and the extract was transferred to a microvial for analysis by GC/MS.

Samples were analyzed using a Finnigan Incos 50 mass spectrometer equipped with a Varian 3400 gas chromatograph, a CTC autosampler and a DG data system. Chromatographic separation was accomplished using a 30 metre DB- 5 column ( 0.25 mm i.d. x $0.25 \mu \mathrm{~m}$ film thickness). Data were collected in EI-mode using multiple ions detection to enhance sensitivity. Three characteristic ions were monitored for each target analyte and surrogate standard compound.

## Quality Assurance Measures for Organic Contaminant Analyses

Quality of the analytical data was assessed through a combination of measures conducted by the analytical laboratory, inter-lab comparisons and blind duplicate submissions.

Each batch was prepared and extracted as lots of at most 9-12 samples which included one appropriate certified reference standard material, one matrix spike, one procedural blank and one duplicate tissue analysis in which the sample was subsampled and each portion extracted and analyzed separately. Concentrations and sample detection limits of target analytes were calculated using ${ }^{13} \mathrm{C}$-labelled surrogates. When surrogate recoveries were particularly low, samples were reanalyzed if sufficient sample remained.

Roughly $10 \%$ of the samples were split or reanalyzed as blind duplicates. In addition, 10 samples were analyzed independently by the Institute of Ocean Sciences, Sidney, B.C., for chlorinated dioxins and furans and PCB congeners for comparison with results provided by Axys Analytical Ltd.

## Trace Metal Analyses

Analyses of metals in tissue collected in the lower Fraser from Mission downstream in 1994 were conducted by Quanta Trace Laboratories, Inc., Burnaby, B.C. Samples were acid-digested $\left(\mathrm{HNO}_{3} / \mathrm{H}_{2} \mathrm{O}_{2}\right)$ and the digestate analyzed for a suite of metals (cadmium, chromium, cobalt, copper, iron, manganese, nickel, and zinc) by ICP-AES (Inductively Coupled Plasma-Atomic Emission Spectroscopy). Mercury was determined by cold-vapour AES and arsenic, lead and selenium determined by graphite furnace AES.

All other samples were submitted to the National Laboratory for Environmental Testing. Arsenic and selenium were measured by ICP, mercury by cold-vapour atomic absorption spectrophotometry, and all other metals by atomic absorption spectrophotometry.

Trace metal QA/QC included blind duplicate analyses and use of certified reference materials.

## Contaminant Data Summary and Presentation

Contaminant analysis data were screened for quality assurance and quality control (QA/QC) indicators such as surrogate recoveries, procedural blanks, matrix spikes, lab and blind duplicate results and general trends. Where suspect data were identified, the values were either excluded from future analysis or flagged and used in further analysis with caution.

A wide array of approaches have been suggested for analysis of data with many observations below the limit of analytical detection (Helsel 1990), none of which result in completely satisfactory or unbiased results. The approach used here was a pragmatic compromise between utility and reality, and has been used by Niimi (1996). When an analyte was detected in at least one composite sample within a reach, non-detects of that compound were assigned $1 / 2$ DL with the presumption that one or more occurrence indicates the presence within the area. Where the compound was not detected in any analysis within a reach, NDs were assigned a value of zero. An exception was total PCBs, where all NDs were set at $1 / 2 \mathrm{DL}$.

Statistical analyses were conducted using either SYSTAT 5.0 (Wilkinson 1990) or SigmaStat 1.0 (Jandel 1997). Principal components on dioxin and furan congener proportions were calculated from a correlation matrix after first normalizing the data to a unit sum by sample.

## Hepatic Mixed Function Oxygenase (MFO) Induction

Liver tissues for analysis of MFO induction were dissected from the fish, rinsed in 1.15\% potassium chloride solution, and frozen in liquid nitrogen within minutes of sacrificing the fish. Approximately $20 \%$ of the liver samples were split before freezing to prepare blind duplicates for submission with the samples in 1995 and 1996.

In the laboratories, samples were transferred from liquid nitrogen to $-80^{\circ} \mathrm{C}$ freezers, then thawed on ice at the time of analysis.

## 1994 Method

In 1994, the hepatic 3-cyano-7-ethoxycoumarin-o-deethylase (ECOD) and 7-ethoxyresorufin-odeethylase (EROD) catalytic activities were measured by the Institute of Ocean Sciences, Sidney, B.C.

Livers were homogenized in 5 mL of cold $1.15 \%$ potassium chloride then centrifuged at $10,000 \mathrm{~g}$ at $4^{\circ} \mathrm{C}$ for 15 minutes to produce the post-mitochondrial supernatant. The supernatant was then centrifuged at $100,000 \mathrm{~g}$ for 40 minutes to produce the microsomal pellet. The pellet was resuspended in 0.1 M sodium phosphate buffer ( pH 7.5 ). Protein concentration was determined in each of these samples using the Bradford assay (Bradford 1976).

The following deviations from this method occurred:

1. For peamouth chub and mountain whitefish collected from the Main and North arms, Hansard, Woodpecker, and Nechako, the post-mitochondrial supernatant was prepared and frozen at the field laboratory.
2. ECOD and EROD activity in starry flounder livers was determined on the post-mitochondrial supernatant, not the microsomal pellet.

The catalytic assay for EROD activity was performed by placing 2 mL of 0.1 M sodium phosphate buffer ( pH 7.5 ) in a fluorometer cuvette and adding NADPH to a final concentration of $200 \mu \mathrm{M}$. An appropriate volume ( 50 to $300 \mu \mathrm{~L}$ ) of microsomal protein was added and the cuvette was mixed well by gently shaking. The fluorescence was recorded for 30 seconds using a Perkin-Elmer Model 204 fluorescence spectrophotometer with excitation at 510 nm and emission wavelength at 585 nm . This fluorescence was used as a control, i.e., there should be no increase in fluorescence with time. At this point the substrate ethoxyresorufin was added to a final concentration of $1 \mu \mathrm{M}$, and the increase in fluorescence with time was recorded for at least one minute. A known amount of resorufin, the product of this enzyme reaction, was added as an internal standard. The increase in fluorescence was then expressed in terms of pmols of resorufin produced per minute per mg of protein. The ECOD activity assay followed the same method, except the excitation and emission of the product 3-cyano-7-hydroxycoumarin was 408 nm and 450 nm , respectively.

## 1995 and 1996 Method

Analyses were conducted at the National Water Research Institute, Burlington, Ontario. Liver samples ( $0.01-0.05 \mathrm{~g}$ ) were homogenized in $500 \mu \mathrm{~L}$ HEPES-KCL buffer. Crude microsomal preparations were prepared by centrifugation at 9000 g for 20 min at $2^{\circ} \mathrm{C}$. The supernatant was centrifuged at $105,000 \mathrm{~g}$ for 60 minutes at $2^{\circ} \mathrm{C}$. The resulting microsomal pellet was re-suspended in $20 \%$ glycerol in HEPES. The catalytic assay of the conversion of the substrate ethoxyresorufin to resorufin was conducted in micro-titre plates. Each sample was assayed in triplicate, and a set of external positive ( $B$-naphthoflavone) and negative (unexposed fish) controls was also assayed with each plate. Fifty $\mu \mathrm{L}$ of liver homogenate and $50 \mu \mathrm{~L}$ of 7ER/HEPES buffer were added to each well and the reaction was started with $10 \mu \mathrm{LNADPH}$. The progress of the reaction was followed at oneminute intervals over a period of 13 minutes using a Cytofluor 2300 micro-plate fluorometer with excitation at 530 nm and emission wavelength at 590 nm . The fluorometer data files were imported to spreadsheets where fluorescence was converted to concentrations of resorufin based on standards included with each plate. Slopes of lines relating concentration to time were calculated and crude activity was expressed as pmoles resorufin per minute. Total protein was measured using commercial protein kits and bovine serum albumin as a standard on a micro-plate spectrophotometer, which automatically calculates protein concentrations from the standard included with each micro-plate. Final activity was calculated by dividing activity by protein concentration, to give EROD activity as pmole resorufin per mg protein per minute.

Some archived whole liver samples collected in 1994 from the Mission reach were submitted for analysis in 1996, to compare results attained by the two methods.

## Ageing

Ages were read using surfaces and burnt cross-sections of the sagittal pair of otoliths, by the Department of Fisheries and Oceans Fish Ageing Laboratory, Pacific Biological Station, Nanaimo, BC. Fifteen percent of the otoliths were re-read by a second reader, as a precision test. Ages given indicate whole years of growth.

## Fecundity Estimates

Mature ovaries were preserved in $10 \%$ buffered formalin for analysis of fecundity in 1996. Five replicate sub-samples were weighed and counted from each ovary pair. Sub-samples contained approximately 100 eggs each. Total fecundity for a fish was estimated as the mean of replicates calculated as follows:

Total Fecundity $=$ Sub-Sample Weight / Total Gonad Weight * \# Eggs In Sub-sample.
Eggs in the largest size class were assumed to be maturing eggs and only these eggs were included in the fecundity estimate. A minimum of 50 eggs in this large size class were also measured to estimate mean diameter.

## Gut Content Analyses

For estuary reaches in 1994, stomach samples for gut contents analyses were preserved in 10\% buffered formalin. All material was removed from each gut and sorted, identified, and counted. Each identified group was blotted damp dry on paper towel, and weighed separately to 0.001 g . Parasites were excluded from counts and weights. When an organism was first encountered, it was placed in a separate vial for an identification check by an independent expert, and for future use as part of a reference collection.

## Necropsy and Health Assessment Index

A necropsy (Goede 1993, Goede and Barton 1990) was conducted by assessing organs against the scales of colour and morphological variability in Table 6. External assessments (eyes, gills, thymus, and pseudobranch) were conducted on live fish. The presence of gills parasites was not considered an abnormality unless there was other associated damage. For this study, we modified the original necropsy coding scheme by adding a skin lesion assessment (Barton 1994). In addition, calculation of the health assessment index (Adams et al. 1993) was modified as follows:

- The thymus and pseudobranch assessments were deleted because the assessments were difficult and inconsistent. Involution of the thymus usually occurs by the time of sexual maturity, making this organ difficult to locate (Roberts 1978).
- The liver rating ' C ' was considered normal for peamouth chub. The liver is a lipid storage site for cyprinids; therefore the amount of fat in the liver is more likely a function of amount of lipid stored than ill health.
- Almost all peamouth chub kidneys were coded OT because they were swollen and pale, therefore the baseline HAI for peamouth chub is 30 .

Due to the subjective nature of the necropsy, the assessments were confirmed by the recorder, so that two people would agree on the assessment before it was recorded. In addition, a quality assessment test of variability among assessors was conducted in 1996. Assessors \#1 and \#2 are:

| Reach | Assessor \#1 | Assessor \#2 |
| :--- | :--- | :--- |
| Main Arm | Recorder A + Externalist A | Recorder B + Externalist B |
| Hansard | Recorder A + Externalist A | Recorder B + Externalist B |
| Woodpecker | Recorder A + Externalist A | Recorder C + Externalist B |

The 2 recorders conducted their internal assessments simultaneously inside the mobile laboratory. The external assessments were done outside the mobile laboratory by two other assessors, who reported data to their respective recorders, independently. The external and internal assessors were always paired the same, except at Woodpecker, a replacement recorder completed the internal assessments.

## Histopathology

Samples of gill, liver, spleen, hindgut, and kidney tissues were preserved in Davidson's Solution for histological examination. In 1995 and 1996 immature gonads were also taken for positive identification of sex. In 1995 and 1996, approximately $10 \%$ of the tissue samples were split to prepare blind duplicates for submission with the samples.

Histological assessments were carried out by Dawna Brand at the Department of Biology, University of Victoria for 1994 samples and by Dr. John Bagshaw at the Department of Fisheries and Oceans' Pacific Biological Station in Nanaimo for 1995 and 1996 samples. One $5 \mu \mathrm{~m}$ section was prepared from each tissue following standard histological procedures (Humasen 1979). Sections were stained with a standard Harris' Haematoxylin and counter-stained with eosin. The histological evaluation noted only the presence of specific abnormalities without quantification of the number and severity. In 1995 and 1996, approximately $15 \%$ of the slides were re-evaluated by the second experienced histologist (Dawna Brand, Department of Biology, University of Victoria) as a precision test.

Slides from peamouth chub and starry flounder collected in 1995 and 1996 from the estuary (Main Arm and North Arm) were sent to Dr. Kelly Munkittrick at the National Water Research Institute for examination for intersex. These reaches were selected because they have the highest input of sewage effluent. Sewage effluents contain endocrine disrupting chemicals which would be expected to cause endocrine effects such as intersex.

## Results and Discussion

## Fish Catches

A list of species caught in each reach sampled in the Fraser basin from 1994 to 1996 is presented in Table 7. In all tables and figures, sample reaches along the Nechako and Fraser rivers are reported from the north to the southwest (i.e., Nechako and Hansard, farthest north, are reported first, through to the Main Arm and North Arm, at the mouth). The Thompson basin reaches are reported last. The species diversity, measured as the number of species per reach, increased from the upper to the lower reaches. The numbers of fish species caught in the reference reaches (Hansard - 12 species, Nechako - 13 species, and North Thompson - 12 species) were lower than in the downstream sites. The greatest species diversity was observed in the Main Arm (20 species), due to the combined presence of both freshwater and marine species. McPhail (1998) found the same pattern in his examination of the distribution of fishes in the Fraser basin.

Relative fish abundance between reaches was estimated by standarizing the catch data for each reach by the number of fish caught per seine set, electrofishing pass, trawl, hoop net set, and gee trap (catch per unit effort). Catch per unit effort (CPUE) estimates for the most common fish species are presented in Table 8 to Table 10. Beach seining was by far the most efficient means of sampling adult mountain whitefish and peamouth chub, and bottom trawls caught most of the starry flounder. Boat electroshocking was not very successful. Large Vexar ${ }^{\mathrm{TM}}$ "gee" traps were useful in the turbid upper

Fraser River waters, but proved to be unsuccessful in clear-flowing sites. The CPUEs, summarized by reach and year, for the six most common species of fish captured by beach seining (mountain whitefish, peamouth chub, suckers, squawfish, redside shiner, and chinook salmon) are presented in Figure 5.

CPUE should be proportional to the abundance of fish in the stock. However, season, water temperature, water level, turbidity, currents, capture method and other variables can influence CPUE (Hubert 1983, Swanson et al. 1992). In this study, the sites and habitats where peamouth chub, mountain whitefish, and starry flounder were abundant were fished until the desired samples were obtained. For example, fall aggregations of mountain whitefish at the mouths of creeks were targeted; shallow and weedy areas of the reach were fished to obtain peamouth chub; and bottom trawls were used to collect starry flounder in the lower reaches of the estuary. This sampling strategy likely biased the CPUE - increasing catches of mountain whitefish, peamouth chub, and starry flounder, while decreasing the catches of other species.

CPUE varied greatly between years, with catches generally highest in 1995, intermediate in 1994, and lowest in 1996. For example, peamouth chub beach seine CPUEs in 1994-5 in the estuary ranged from 19.89 to 73.82 , but in 1996, CPUEs were only 4.07 to 6.97 ; and mountain whitefish CPUE were much higher in 1995 than in 1994 in the upper Fraser ( 10.93 in 1994 versus 31.95 in 1995). Inter-annual variation in CPUE was likely the result of a number of factors including sampling dates, flow conditions, water levels and habitat differences. For example, unusually high flows in 1996 washed out the most productive peamouth chub sampling site and moved the Naver Creek confluence with the Fraser River at Woodpecker. As a result, even considerable sampling effort in 1996 resulted in only 38 mountain whitefish and 55 peamouth chub.

Mountain whitefish were most abundant in the upper reaches of the Fraser basin (CPUE of 10.93 to 31.95 in the Nechako and at Hansard) and declined downstream (CPUE of 1.38 to 3.92 at Marguerite and Agassiz). Only a few juveniles were caught downstream of Agassiz in the estuary. Mountain whitefish were abundant in the North Thompson and Thompson rivers (CPUE of 10.35 to 43.48).

When fall aggregations could be located in a reach, the mountain whitefish samples were easily obtained. For example, in 1994, CPUE for mountain whitefish in the North Thompson (43.48) far exceeded that in the Thompson (10.35), because dense aggregations were found at the mouths of the Barriere River and Louis Creek. A summary of the distributions of mountain whitefish populations sampled in each reach is given in Table 11. Mountain whitefish catches were lowest in the Marguerite reach. In 1994, most of the side channels were dry, and gravel bars were large with shallow water extending well out into the river, decreasing the effectiveness of beach seines (Triton Environmental Consultants Ltd. 1994). Even during higher flows in 1995, aggregations of fish were absent from the mouths of tributaries where sampling efforts were concentrated. It is possible that sampling occurred after migration of the whitefish up the tributaries to spawn, particularly during the later sampling of 1994.

Alternatively, the Marguerite reach has the highest suspended sediment concentration (Carson 1988) and river gradient in the basin, and lacks resting areas for fish such as side channels and backwaters. Suspended sediment and higher currents may have driven mountain whitefish to alternative habitats.

Peamouth chub were present in all reaches sampled, including the estuary, because they are tolerant of dilute seawater. Although peamouth were distributed throughout the basin, abundance generally increased from upstream to downstream reaches. Abundance was highest in the estuary (CPUE of 19.89 to 73.82, except in 1996 ) and lowest in the North Thompson (CPUE of 0.99 to 5.98). Peamouth chub were distributed diffusely within reaches, making them more difficult to capture than mountain whitefish (Triton Environmental Consultants Ltd. 1994, 1996a, 1996b).

Suckers were most abundant in the Nechako and middle Fraser reaches (Woodpecker, Marguerite and Agassiz), and relatively rare in the estuary and Thompson basin. It is interesting to note that the suckers were abundant in the Marguerite reach, despite the high gradient and suspended sediment levels. Suckers lie close to the bottom where the currents are low (Scott and Crossman 1973), which may favour their abundance in this reach.

Squawfish and redside shiners had similar distribution patterns in the Fraser basin, and were both dominant in the Nechako, and intermediate in the Woodpecker and Marguerite reaches. Both species were rarer at Hansard and in the lower Fraser. In the Thompson basin, however, redside shiners were very abundant whereas squawfish were rare.

Most of the chinook salmon captured in this study were juveniles. CPUEs were highest in the upper and middle mainstem (Hansard to Marguerite) and in the North Thompson River.

Starry flounder is a marine species, therefore was caught only in the lower reaches of the Fraser River. The highest bottom trawl CPUEs were in the Main Arm (68 to 93) followed by the North Arm (9 to 46), Barnston (2.10) and Mission (0.15). No starry flounder were caught upstream of New Westminster, which is the limit of marine intrusion in the Fraser River.

Habitat data, including bottom substrate characteristics, hydrology, bank slope, water temperature, water velocity, shore gradient and GPS coordinates, were collected for each reach and are included with the electronic raw catch data files. As more than one substrate type has been identified for most sampling sites, a definite relationship between CPUE and habitat substrate was not explored.

In conclusion, mountain whitefish were most abundant in the upper reaches - Nechako River, Hansard, and North Thompson. Peamouth chub were evenly distributed amongst all the reaches, with the highest catches observed in the estuary. The starry flounder catches were highest in the Main Arm. Some variables that may have contributed to the fishing success are sampling time, site selection, habitat type, hydrology, river flow, shore gradient, and sampling gear and technique.

## Contaminants in Tissues

A wide array of organochlorine, resin acid and trace metal contaminants were analyzed in fish tissues and bile from reaches throughout the Fraser basin in 1994 and 1995. A total of 11 river reaches were sampled in 1994 and 9 in 1995.

Graphical presentation and interpretation is used throughout the following discussion, with minimal statistical analysis. The reasons are twofold. First, the total sample numbers were small, with a maximum of five composite tissue sample analyses per species within each reach. Second, the variability in concentrations within a reach was frequently very high. Sophisticated statistical analysis of patterns in geographic distribution of contaminant levels are obscured by the low statistical power resulting from this combination of small sample size and high variability in concentrations.

For simultaneous visual comparison of tissue levels of all three species over reaches and years, overlaid histograms are employed (Figure 6). Species pairs collected within each sampling reach in 1994 are shown as background bars with an associated colour or pattern. Summary results of analyses from the same reach and species in 1995 are shown as overlaid narrow bars. When reading these plots, it is important to remember that 1) only peamouth chub were collected in all reaches, 2) in the two lower estuary reaches, whitefish were replaced with starry flounder, 3 ) in the Barnston and Mission reaches, no samples were collected in 1995, 4) of the 11 reaches shown, the first is the Nechako and mainstem Fraser from upstream to downstream, while the last 2 (N. Thompson and Thompson) are tributary reaches upstream and downstream of Kamloops.

## Contaminant Residue Data Quality

## Procedural Blanks

Target analytes in procedural blanks were, with a very few exceptions below detection. Where contaminants were detected in blanks, levels were either far below those of sample tissues in the same batch or the compounds detected in blanks were not present in sample analyses. Sample results were not blank-corrected.

## Surrogate Recoveries

Surrogate recoveries provide a general indicator of the success of the laboratory extraction and clean-up procedures and possible sample matrix interactions. Results are used in calculations of analyte concentrations by the isotope dilution method, so even relatively low recoveries will yield acceptable results as measured by other QA measures. With few exceptions, surrogate recoveries for the 1994 and 1995 data were within acceptable ranges (Table 12, Table 13).

## Lab Duplicates

Lab duplicate analyses were satisfactory for most parameter groups (Table 14, Table 15). Results were excellent for the relatively unreactive organochlorines, such as the dioxins and furans, PCBs and organochlorine pesticides, with average relative standard deviations (RSD $=$ standard deviation/average * 100, expressed as a \%) between pairs of analyses generally less than $10 \%$. Results from chlorophenolic analyses were generally poorer, with widely ranging RSDs - due both to sporadic detections near the detection limit and analytical difficulties due to the reactive nature of many chlorophenolic congeners. Dioxin and furan duplicate results were excellent for all measured congeners except the octachlorinated dioxins, for which the between-sample RSDs averaged $40.5 \%$. Mean RSD values for duplicate trace metal analyses for all but Mn and Ni were less than $10 \%$. For Mn and Ni, RSDs averaged to 20 and $25 \%$ respectively.

## Blind Duplicates

Selected tissue samples were split and resubmitted to the analytical laboratory as blind duplicates as a measure of the precision of the analyses. Results for the more stable organochlorines such as the PCBs, dioxins and furans, and pesticides were good to excellent despite concentrations frequently at or near detection limits. Apparent differences in concentrations were most often attributable to variation in detection limits than to evident analytical errors.

The notable exceptions were in determinations of PAH metabolites in bile, where levels in blind duplicates often bore little relation to concentrations measured in the original analyses. The cause was traced to inconsistent sample injections, which would produce acceptable replication within batches but poor replication between batches. These data are discussed in these results, but only with caution and because they represent the first such data from the Fraser River.

Mean RSD values for blind duplicate trace metal analyses for all metals were less than 10\% (Table 16).

## Certified Reference Materials and Matrix Spikes

At least one certified reference material was analyzed with each batch of samples, where such reference materials were available. In other cases, samples were spiked with known quantities of target analytes as an indication of analytical accuracy.

Results for each analyte group were excellent. Expressed as a percentage of the expected values, either as matrix spikes or certified reference concentrations, average results were: dioxins and furans $97 \%$, chlorophenolics $104 \%$, organochlorine pesticides $95 \%$ in 1994 and 1995, PCBs greater than $96 \%$ in 1994 and 1995. Octachlorodioxin recoveries tended to be relatively poor compared to recoveries of lesser chlorinated dioxins and furans.

Dioxins and furan results from the Regional Dioxin Laboratory at the Institute of Ocean Sciences (IOS) were in very close agreement with that from Axys Analytical (Table 17). Here again, the
octachlorodioxins tended to be relatively low in samples analyzed by Axys relative to IOS, and the results would suggest that, in general, the contribution of this component in the total dioxins and furans may be underestimated. From a toxicological perspective, this is of relatively minor consequence because of the low toxicity of these highly chlorinated congeners.

## Chlorinated Dioxins and Furans

## Background

Chlorinated dioxins and furans (Figure 7) are of concern in the Fraser basin due to production and release of these compounds as a by-product of chlorine bleaching processes in Kraft-process mills (Mah et al. 1989). Even trace levels in effluent are of concern because of their extreme toxicity, resistance to environmental degradation, and the potential of some congeners to bioaccumulate and bioconcentrate in tissues of aquatic organisms. Of the 210 possible dioxin and furan congeners (CEPA 1990), the tetrachlorinated dioxin with chlorine substitution at the 2,3,7 and 8 positions ( $2,3,7,8-\mathrm{TCDD})$ is the most toxic. The high toxicity of this particular compound is due to the planar aspect and molecular dimensions which allows free passage through cell membranes (McFarland and Clarke 1989). Toxicity of other dioxin and furan congeners have been compared to $2,3,7,8-\mathrm{TCDD}$, and toxic equivalency factors (TEFs) have been derived in order to evaluate the significance to human health of the spectrum of dioxins and furans typically represented in environmental samples. Two commonly-used sets of TEFs have been derived, one set for human risk assessment (NATO/CCMS 1988) and a more recent set for other taxa, which recognizes the different sensitivities and metabolic abilities of mammals, fish and birds (van den Berg et al. 1998).

Bleached Kraft pulp mill effluents (BKME) are dominated by tetrachlorinated dioxins (TCDD) and tetrachlorinated furans (TCDF) (Cleverly et al. 1997, Dimmel et al. 1993). In addition to the effluent of pulp and paper mills, dioxins and furans may be released from a number of other sources, both anthropogenic and natural (Cleverly et al. 1997). Combustion sources such as wood burning and municipal incinerators produce a wide spectrum of dioxin and furan congeners, with a particular predominance of the penta- to octachlorinated congeners (Bacher et al. 1992, Cleverly et al. 1997, Zitko 1992). Chlorophenate wood preservatives are particular sources of the higher chlorinated dioxins and furans, especially the hepta- and octachlorinated congeners (Hagenmaier and Brunner 1987). Additionally, chlorophenolic wood presevatives (Jones 1981) were a significant source of dioxins from burning of pentachlorophenol (PCP)-contaminated wood chips (CEPA 1990). Vehicle emissions, such as from vehicles using unleaded gas and diesel trucks, are particularly rich in octachlorinated dioxins (Cleverly et al. 1997). Therefore, analysis of the occurrence patterns of particular dioxin and furan congeners or homologues can provide "source signatures" and point to particular contaminant sources.

High dioxin and furan concentrations have been found in previous surveys of fish tissues in the Fraser basin. For example, a 1988 survey of dioxins and furans in fish tissues near mill discharges in the Fraser basin (Mah et al. 1989) showed levels far in excess of recommended safe concentrations for human consumption. These results led to fish consumption advisories for both the Fraser and

Thompson rivers (Dwernychuk et al. 1996). Pulp bleaching process changes implemented by mills throughout BC through the early 1990s have resulted in dramatic decreases in dioxin and furan levels in both effluents and in the environment (BCMELP 1995, Krahn 1995). Similar trends in dioxin and furan levels have been recorded elsewhere in Canada (Luthe 1998).

## Dioxin and Furan Results

A suite of 15 individual dioxin and furan congeners and group totals (Table 18) were measured in whitefish, peamouth chub and starry flounder liver and muscle from 11 sites in 1994. Tissues were not analyzed for dioxins and furans in 1995. Results of the 1994 analyses are summarized in Table 19 to Table 22.

## Furans

Total furans were dominated by $2,3,7,8-\mathrm{TCDF}$, which was detected in $96 \%$ of peamouth chub liver and muscle and $80 \%$ of whitefish muscle and $87 \%$ of whitefish liver samples (Table 23, Figure 8, Figure 9). TCDD and TCDF are characteristic components of BKME (Cleverly et al. 1997, Trudel 1991), and their presence throughout the basin is probably due to pulp and paper mill discharges. Pentachlorinated furan and hexachlorinated furan congeners were detected sporadically at low levels, particularly in peamouth chub and starry flounder liver downstream of Hope (Figure 9) and in whitefish muscle from the North Thompson (Figure 8). Both of these congeners are released from a wide range of combustion sources.

Total TCDF in peamouth chub muscle from the mainstem Fraser River ranged from below detection ( $<0.10 \mathrm{pg} / \mathrm{g}$ w.w.) near Hansard to a high of $0.73 \mathrm{pg} / \mathrm{g}$ w.w. at Barnston Island (Table 19). In contrast, the highest TCDF concentration in whitefish muscle ( $2.2 \mathrm{pg} / \mathrm{g}$ w.w.) in the Fraser River was measured at Woodpecker, downstream of pulp mill effluents from Prince George (Table 20).

TCDF in fish muscle was much higher in the Thompson basin compared to the Fraser, with mean concentrations of up to 4.20 and $13.6 \mathrm{pg} / \mathrm{g}$ w.w. in peamouth chub and whitefish muscle, respectively compared to a maximum of $1.19 \mathrm{pg} / \mathrm{g}$ w.w. from Fraser reaches (Table 19, Table 20). Trace levels of TCDF were measured in peamouth chub upstream of mill discharges in both the Fraser and Thompson rivers, suggesting possible upstream movement of peamouth chub from as far as Prince George and Kamloops, respectively (Figure 8). Maximum values in whitefish particularly in the Thompson basin were high, with concentrations up to 16 and $22 \mathrm{pg} / \mathrm{g}$ w.w., in the North Thompson and Thompson reaches. There was no difference in TCDF levels between the reference reach on the North Thompson and the Thompson River reach, downstream of Kamloops, a reflection of movement and probable exchange of unexposed and exposed fish between these two areas.

Mean TCDF concentrations in peamouth chub liver were lowest in the upper and middle Fraser reaches and increased four-fold downstream of Marguerite, from less than $2 \mathrm{pg} / \mathrm{g}$ w.w. to greater than $8 \mathrm{pg} / \mathrm{g}$ w.w. in the lower river (Figure 9; Table 21). Mean concentrations of TCDF in peamouth chub liver in the estuary were similar amongst reaches, ranging from 9.4 to $9.5 \mathrm{pg} / \mathrm{g}$ w.w. In whitefish,

TCDF in liver ranged from below detection ( $<0.60 \mathrm{pg} / \mathrm{g}$ w.w.) at Hansard to $2.3 \mathrm{pg} / \mathrm{g}$ w.w. downstream of Quesnel. These relatively elevated levels in peamouth chub in the estuary probably reflect a combination of high liver lipid content and the combined effluents of the five upstream mills. Contamination from the Thompson River in particular is implicated, since levels were highest in these reaches ( 11.0 to $37.5 \mathrm{pg} / \mathrm{g}$ w.w.) and tend to increase in the mainstem Fraser downstream of the Thompson-Fraser confluence. The Thompson River at the time of sampling probably functioned as a "point-source" discharge of these contaminants to the mainstem Fraser.

In starry flounder, trace ( $0.12 \mathrm{pg} / \mathrm{g}$ w.w.) levels of TCDF were found in muscle from only the Main Arm reach in the estuary. TCDF levels in liver were nearly identical between the North and Main Arm reaches, at 2.6 and $2.7 \mathrm{pg} / \mathrm{g}$ w.w. respectively.

Interspecies differences in TCDF burden are apparent, particularly in the elevated levels in whitefish relative to peamouth chub muscle in the Thompson reaches, and between liver tissues in both species throughout the Fraser basin (Figure 8, Figure 9). This pattern is, in part, a reflection of differences in tissue lipid content, since the differences shrink when concentrations are lipid-normalized (Figure 10, Figure 11). Levels in peamouth liver were higher than those measured in whitefish, doubtless due to the much higher lipid content in the former species.

## Dioxins

The most toxic of the dioxin congeners, $2,3,7,8-\mathrm{TCDD}$, was detected in fish tissue in all but the Hansard reach (Figure 8, Figure 9). Concentrations in peamouth chub muscle from the upper Fraser reaches (upstream of Mission) ranged from below detection ( $<0.10 \mathrm{pg} / \mathrm{g}$ w.w.) to $0.10 \mathrm{pg} / \mathrm{g}$ w.w. (Table 19). In the estuary, levels in peamouth ranged from below detection ( $<0.10$ ) to $0.26 \mathrm{pg} / \mathrm{g}$ w.w. in a single sample from the North Arm. Concentrations in whitefish muscle were considerably higher than in peamouth chub, but still averaged less than $1.0 \mathrm{pg} / \mathrm{g}$ w.w. through the mainstem Fraser in the upper basin reaches at Woodpecker and Marguerite.

In starry flounder, 2,3,7,8-TCDD was detected in liver, but not in muscle (Figure 8, Figure 9). Concentrations were low ( $<0.35 \mathrm{pg} / \mathrm{g}$ w.w.), and were roughly $1 / 3$ those detected in peamouth chub from the same reaches.

Patterns in total TCDD in liver in the mainstem Fraser were similar to muscle, with the lowest levels in the upper basin and generally increasing downstream to the estuary (Figure 8, Figure 9). The highest levels were in whitefish liver ( $2.9 \mathrm{pg} / \mathrm{g}$ w.w.) in the Woodpecker reach. TCDD in peamouth chub liver was elevated in the estuary, ranging from $1.70-3.0 \mathrm{pg} / \mathrm{g}$ w.w. (Table 21, Table 22).

Total TCDD in both whitefish and peamouth chub muscle tended to be highest in the Thompson reaches. Concentrations in whitefish muscle were quite variable, even within reaches. For example, in samples from the Thompson reach downstream of Kamloops, 2,3,7,8-TCDD ranged from 0.30 to $3.4 \mathrm{pg} / \mathrm{g}$ w.w. (Table 20). Values in peamouth chub muscle were much lower, from non-detectable ( $<0.10 \mathrm{pg} / \mathrm{g}$ w.w.) to $0.40 \mathrm{pg} / \mathrm{g}$ w.w. In contrast, TCDD was not detected in whitefish liver in the

Thompson drainage and peamouth chub liver levels were comparable to those measured in the estuary (1.25-1.95 pg/g w.w.).

The efficacy of whitefish, relative to other species, in accumulating TCDD has been noted in other studies (Owens et al. 1994a) and appears to relate to factors other than simple differences in lipid content (Figure 10). This is particularly evident in those tissues collected from reaches in the middle Fraser, where lipid-normalized levels in whitefish far exceed those in peamouth chub (Figure 10). Influences other than trophic position would seem to contribute, since both species appear to have similar feeding patterns and habitat preferences. Anecdotal observations of peamouth feeding suggest these fish may take a relatively larger proportion of terrestrial insects in turbid reaches, perhaps reducing intake of water-borne dioxins and furans (McPhail 1997).

Other dioxin congeners were detected in peamouth chub liver and muscle throughout the basin (Figure 8, Figure 9). Octachlorodioxin in particular, was the congener at highest concentration at a number of sites (Table 19, 20 and 21) with measured levels, in an unusual case, up to $17 \mathrm{pg} / \mathrm{g}$ w.w. in peamouth chub muscle at Mission (Figure 8, Table 19). Lipid-normalized octachlorodioxin patterns show clearly the elevated levels in the estuary reaches, particularly in starry flounder (Figure 12). Flounder tissues have very low in lipid content ( $<1 \%$ ), and even trace detections result in high lipidnormalized concentrations.

## Dioxin and Furan Congener Profiles

Congener profiles from liver tissue analyses (Figure 9), and to a lesser extent muscle (Figure 8), show clearly that as the diversity and density of potential effluent sources increases, so too do the diversity of detected dioxin and furan congeners. This pattern is particularly evident in comparison of analyses of peamouth chub liver from the upper Fraser with those collected from reaches in the estuary (Figure 9). In upstream reaches, only one or two congeners were detected, with 8 to 9 congeners detected in the lower Fraser Valley and estuary reaches. Congener profiles in flounder and peamouth chub liver collected in the same reaches present similar patterns (Figure 9), with a wide spectrum of congeners being detected at sites in the highly industrialized estuary area. While higher chlorinated congeners contribute little to the overall toxicity due to dioxins and furans, their presence is important in indicating particular contaminant sources.

Principal component analysis of the congener data shows a clear separation of reaches immediately downstream of mills on the Fraser River and in the Thompson basin and those elsewhere (Figure 13). This again, is due to the elevated TCDD/TCDF in these reaches relative to other congeners.

Detections at some sites were quite anomalous, and may indicate previously unknown aspects of the biology of the target fish species. The presence of TCDF and heptachlorinated furan in peamouth chub tissues from "reference" areas upstream of Prince George on the Fraser River, and upstream of Kamloops in the Thompson drainage, is particularly interesting. Since the measured TCDF contamination in the Fraser River "reference" area is probably due to exposure to pulp and paper mill effluents downstream of Prince George, the occurrence suggest greater mobility in the species than
was previously suspected. The presence of heptachlorofuran in liver upstream of Prince George suggests another source, perhaps chlorophenate wood preservatives (Cleverly et al. 1997, Hagenmaier and Brunner 1987). This is corroborated by other measurements, particularly elevated PCP and high concentrations of resin acids in bile (considered in section on resin acids in bile).

## 2,3,7,8-TCDD Toxic Equivalents

Dioxin and furan concentrations were expressed in terms of 2,3,7,8-TCDD toxic equivalents (TEQ) using both NATO toxic equivalent factors (designated $\mathrm{TEF}_{\text {NATO }}$ and $\mathrm{TEQ}_{\text {NATO }}$ ) from (CEPA 1990) and more recent taxon-specific TEFs (van den Berg et al. 1998) (Table 24). Both sets of TEQs are included for completeness (Table 19 to Table 22), and the van den Berg et al. (1998) TEQs are included in Figure 8 and Figure 9. The NATO TEF values are the standard accepted values for evaluating risk to humans of exposure to dioxins and furans, and the newer set are more correctly designed to assess the risk to wildlife. These current TEFs better reflect the very different sensitivities of mammals, fish and birds ( $\mathrm{TEQ}_{\text {Mammal }}, \mathrm{TEQ}_{\text {Fish }}, \mathrm{TEQ}_{\text {Bird }}$ ) to individual dioxin and furan congeners. In addition, the van den Berg et al. (1998) TEF values form the basis of interim CCME tissue residue guidelines for piscivorous wildlife. In their application, the most significant difference between the 1998 TEFs, particularly with respect to the Fraser River data, is in the avian value for 2,3,7,8-TCDF which is considered as equivalent in toxicity to $2,3,7,8-\mathrm{TCDD}$ (Table 24).

Average $\mathrm{TEQ}_{\text {NATO }}$ in peamouth chub muscle in the mainstem Fraser River ranged from 0.02 to 0.13 $\mathrm{pg} / \mathrm{g}$ w.w., with only minor differences among reaches. In peamouth chub liver, $\mathrm{TEQ}_{\text {NATO }}$ were low ( $<0.2 \mathrm{pg} / \mathrm{g}$ w.w.) through the upper basin, and elevated through the estuary reaches ( $2.64-2.98 \mathrm{pg} / \mathrm{g}$ w.w.), largely owing to the presence of $2,3,7,8-\mathrm{TCDD}$ in these reaches (Figure 9). $\mathrm{TEQ}_{\text {NATO }}$ in peamouth chub tissues were higher in the Thompson basin, with mean muscle levels of 0.51 to 0.60 $\mathrm{pg} / \mathrm{g}$ w.w. and liver levels to $5.7 \mathrm{pg} / \mathrm{g}$ w.w.. The elevated TEQ ${ }_{\text {NATO }}$ in the Thompson was a consequence of measurable levels of 2,3,7,8-TCDD and relatively high concentrations of 2,3,7,8TCDF (Figure 8, Figure 9). In peamouth chub muscle tissues in the Thompson basin, $\mathrm{TEQ}_{\text {NATO }}$ were similar upstream and downstream of the Kamloops pulp mill (Figure 8), while TEQ ${ }_{N A T O}$ in liver were considerably higher downstream.

In whitefish liver tissues, $\mathrm{TEQ}_{\text {NATO }}$ was highest in the Woodpecker reach ( $2.55 \mathrm{pg} / \mathrm{g}$ w.w.) and declined ( $1.73 \mathrm{pg} / \mathrm{g}$ w.w.) in the Marguerite reach, downstream of Quesnel, owing to a reduction of TCDD relative to TCDF (Figure 9). TEQ $_{\text {NATO }}$ in whitefish liver from the Thompson basin was highest in the North Thompson (Table 22). In whitefish muscle tissue, $\mathrm{TEQ}_{\text {NATO }}$ was highest in the Thompson basin ( $\sim 2 \mathrm{pg} / \mathrm{g}$ w.w.), here again, from a combination of high TCDF levels and the presence of TCDD residues (Figure 8).

Since the WHO mammal and fish TEF values are relatively similar to the $\mathrm{TEF}_{\text {NATO }}, \mathrm{TEQ}_{\text {Mammal }}$ and $\mathrm{TEQ}_{\text {Fish }}$ tended to be very similar to $\mathrm{TEQ}_{\text {NATO }}$. Pronounced departures were seen in $\mathrm{TEQ}_{\text {Bird }}$ values, largely due to the presence of $2,3,7,8-\mathrm{TCDF}$. Elevated $\mathrm{TEQ}_{\text {Bird }}$ are apparent in the Thompson sampling reaches, with average values of 11.48 to $13.10 \mathrm{pg} \mathrm{TEQ}_{\text {Bird }} / \mathrm{g}$ in whitefish muscle (Table 20) and 11.95 to 39.45 pg TEQ Bird $/ \mathrm{g}$ in peamouth liver (Table 21).

Calculated TEQ $_{\text {NATO }}$ values for all tissues throughout the basin were well below the $20 \mathrm{pg} / \mathrm{g}$ w.w. Health and Welfare Canada guideline for human consumption.

Exceedences of the CCME tissue residue guideline for protection of wildlife $(0.71 \mathrm{pg} \mathrm{TEQ} / \mathrm{g}$ wet tissue weight; CCME 1999) were found in a number of reaches and tissues. Values in excess of the guideline in peamouth muscle were seen in the Thompson drainage and the North Arm of the Fraser, only in terms of TEQ Bird (Table 19). Similarly, substantial exceedences, greater than 10x of the guideline level, for any of the three TEQs in whitefish muscle were seen only in TEQ $_{\text {Bird }}$ in tissues from the Thompson drainage.

The particularly elevated $\mathrm{TEQ}_{\text {Bird }}$ in tissues from the Thompson basin may be of concern. The high calculated levels in peamouth liver (to $39 \mathrm{pg} \mathrm{TEQ} / \mathrm{g}$ ) may pose less risk than levels in whitefish muscle since this tissue would represent the bulk of the body mass consumed by piscivorous birds such as osprey. The measured levels may be having an effect on wildlife, since studies of fledgling success in osprey upstream and downstream of Kamloops showed greater success at upstream sites, possibly attributable to lower contaminant burden (Wilson et al. 1999).

## Comparison with Other Studies

Process changes and upgrades at pulp and paper mills in the early 1990s have resulted in dramatic reductions in dioxin and furan contamination in fish tissues throughout the Fraser basin. Decline in 2,3,7,8-TCDD in whitefish following these process changes was rapid and dramatic. Prior to process changes, concentrations of over $140 \mathrm{pg} / \mathrm{g}$ w.w. were measured in whitefish tissues (Mah et al. 1989), while present totals are generally less than $3 \mathrm{pg} / \mathrm{g}$ w.w. (Figure 14). Results in 1995 (Dwernychuk et al. 1996) were similarly low, with most values ranging from below detection to $0.50 \mathrm{pg} / \mathrm{g}$ w.w. A similar sharp decline in TCDD in fish tissues occurred in the Thompson basin after process changes at the Weyerhauser pulp and paper mill in Kamloops (Figure 14).

The situation was similar for the chlorinated furans. For example, in 1990 Hatfield Consultants Ltd. (1994) found 2,3,7,8-TCDF levels in individual whitefish muscle and liver samples up to 290 and $660 \mathrm{pg} / \mathrm{g}$ w.w., respectively, downstream of Quesnel. Composite tissue samples analyzed from the same river reach during this study (1994) showed maximal TCDF levels in whitefish of only $1.0 \mathrm{pg} / \mathrm{g}$ w.w. in muscle and $2.3 \mathrm{pg} / \mathrm{g}$ w.w. in liver.

There remains a concern about the extreme variability in TCDD burden among individual fish. For example, in monitoring levels in fish tissues downstream of Marguerite even as late as 1995 one individual whitefish had a 2,3,7,8-TCDD concentration of $68 \mathrm{pg} / \mathrm{g}$ w.w. (Dwernychuk et al. 1996). The situation in the Thompson reaches in this study are probably similar. In the four composite whitefish muscle samples analyzed for TCDD, three of four averaged $<0.30 \mathrm{pg} / \mathrm{g}$ w.w., but $3.4 \mathrm{pg} / \mathrm{g}$ w.w. in the last. Since these analyses were on composites of 5 individual fish, this result could represent contamination of the composite by a single fish with as much as $16 \mathrm{pg} / \mathrm{g}$ w.w. TCDD. Both these results suggest that the question of dioxin contamination may not be solved completely.

Other river basins elsewhere in North America, where similar reductions in effluent contaminants have been achieved have shown varying successes in translating process changes into declines in fish tissue burdens (Johnson et al. 1996, Owens et al. 1994a). When reductions in tissue dioxin and furan burden in fish are seen, they may be translated rapidly into declines in higher trophic levels (Whitehead et al. 1992).

## Dioxin and Furan Summary

Levels of chlorinated dioxins and furans throughout the basin are low compared to historical measurements, but may still, in some cases, pose a risk to piscivorous wildlife. Congener profiles in fish tissues from the Fraser River reaches upstream of Hope showed a clear dominance of characteristic pulp-mill-related TCDD and TCDF.

In the lower Fraser reaches, the wide array of potential contaminant sources is reflected in the diversity of detected dioxin and furan congeners. Concentrations measured in 1994 were a small fraction of historical levels, and levels may continue to decline with any further improvements in effluent process controls. Non-pulp and paper sources of dioxins and furans, particularly those related to combustion and possibly the continued use of PCP, probably contribute to the total burden in tissues in the estuary reaches. However, penta to octachlorinated congeners predominate from these sources (Cleverly et al. 1997). These less toxic dioxin and furan components, although at measurable levels in the environment, have low TEFs and would thus contribute only a small amount of the total toxicity.

The similarity of dioxin and furan concentrations and congener patterns in fish tissues from the North Thompson and Thompson rivers suggest considerable movement and mixing of fish between the two areas with consequent upstream distribution of pulp mill-related contaminants.

## Chlorophenolics

## Background

BKME contain a wide spectrum of chlorophenolics formed by reaction of natural lignins and phenolics with chlorine during the bleaching process (McLeay and Associates Ltd 1986). The result is a veritable soup of organic compounds, including various chlorophenols, chloroguaiacols, chlorocatechols, chlorovanillins, chlorosyringols and chlorosyringaldehydes (Figure 15) (Brumley et al. 1996, Paasivirta et al. 1992). These compounds are hydrophobic to varying degrees, and some will accumulate in tissues of aquatic organisms exposed to effluent streams. Absorption is principally by diffusion across respiratory surfaces and to a lesser extent through ingestion during feeding (Suedel et al. 1994).

Pulp and paper mills have historically been the major contributor of chlorophenolics to the Fraser basin. Chlorophenols have a wide array of applications as feedstock for chemical production and as biocides for a range of products. Jones (1984) lists 100 commercial products containing chlorophenolics. Some of these may be released by sources such as wood treatment facilities and municipal effluents (Norecol 1993). For example, basin-wide sales of PCP, used in limited application as a heavy-duty wood preservative, during 1995 were in the order of 123 tonnes (Norecol 1997). Commercial PCP formulations also include an array of tetrachlorophenols and other chlorinated compounds including dioxins and furans (Hagenmaier and Brunner 1987, Jones 1981). Congener detection patterns may provide clues to general contaminant sources. For example, the presence of 3,4,5-trichloroguaiacol is characteristic of pulp mill effluents (Hall et al. 1991), while measurable levels of more highly chlorinated chlorophenols (particularly PCP) may be more indicative of discharges from wood treatment facilities or drainage from treated waste storage areas.

In most cases, chlorophenolics are readily metabolized to their polar conjugates and rapidly excreted either in bile or urine (Wesen 1990). Studies have shown a close relationship between ambient and tissue levels of chlorophenolics in fish (Servizi et al. 1988), and as a result, as distance from an effluent discharge increases, levels rapidly drop to non-detectable levels. Chlorophenolics and their metabolites are sequestered in bile prior to excretion, resulting in high concentrations of both parent compounds and metabolites (Oikari and Holmbom 1986). These relatively high levels make bile a convenient and sensitive matrix for monitoring exposures to even low levels of effluent (Brumley et al. 1996, Wachtmeister et al. 1991). For example, studies have shown bioconcentration of chlorophenolics in bile at levels from 10 to 100,000 times ambient concentrations, with half lives on the order of one week (Owens et al. 1994b).

## Chlorophenolic Analysis Results

A total of 47 chlorophenolic congeners were measured; 16 chlorophenols, 10 chloroguaiacols, 9 chlorocatechols and an additional 12 phenolics such as chlorosyringaldehydes and chlorovanillins (Table 25). In 1994, muscle and liver from 9 reaches and two species per reach were analyzed. Bile samples were collected in 1994 from peamouth chub and whitefish from five reaches, three in the upper Fraser and two in the Thompson River drainage. In 1995, bile only from 7 reaches in the basin was analyzed for chlorophenolics.

## Chlorophenolics in Liver and Muscle Tissues

Chlorophenolic analysis results are summarized in Table 26. Chlorophenolics group totals for all tissues are summarized in Table 27. Congener detection profiles for each species and tissue matrix are presented in Figure 16 and Figure 17 and Table 28 to Table 33.

As with other surveys of chlorophenolics in fish tissue downstream of pulp mills (Owens et al. 1994b), detections in muscle were erratic and at low concentrations. Only a relatively few compounds were measured at average within-reach levels greater than $1 \mathrm{ng} / \mathrm{g}$ w.w. (Table 26). Total chlorophenolics in fish muscle throughout the basin were low, averaging less than $2 \mathrm{ng} / \mathrm{g}$ w.w. in most
reaches (Figure 16). However, individual muscle composites ranged as high as $6.2 \mathrm{ng} / \mathrm{g}$ w.w. for peamouth chub at Marguerite and to $13 \mathrm{ng} / \mathrm{g}$ w.w. for whitefish from the North Thompson (Table 27). Despite potential contamination by effluent discharges from three upstream pulp and paper mills, total chlorophenols in peamouth chub and whitefish muscle were only slightly higher at Woodpecker ( $1.6 \mathrm{ng} / \mathrm{g}$ w.w. peamouth, $1.3 \mathrm{ng} / \mathrm{g}$ w.w. whitefish) compared to the upstream "reference" reach (1.1 $\mathrm{ng} / \mathrm{g}$ w.w. peamouth and whitefish), and showed only a modest increase in the Marguerite reach (2.9 ng/g w.w. peamouth, 2.2 whitefish) downstream of Quesnel (Table 27, Figure 16). In the Thompson reaches, total chlorophenol concentrations in peamouth chub muscle were similar to those measured from upstream Fraser reaches, while levels in whitefish were somewhat higher. In addition, mean total chlorophenols in both peamouth and whitefish muscle were higher in the North Thompson ( $1.8 \mathrm{ng} / \mathrm{g}$ w.w. peamouth, $4.7 \mathrm{ng} / \mathrm{g}$ w.w. whitefish), compared to the Thompson reach ( $1.0 \mathrm{ng} / \mathrm{g}$ w.w. peamouth, $2.2 \mathrm{ng} / \mathrm{g}$ w.w. whitefish) (Figure 16; Table 27).

Of the suite of chlorophenolics, only 2,4/2,5-dichlorophenol was detected in muscle and liver from all three species. Fifteen congeners were not detected in either muscle or liver (Table 26). Detection frequencies for most compounds were less than $10 \%$. The most frequently detected compounds in muscle were 2,4/2,5-dichlorophenol (peamouth chub detection frequency $48 \%$ (21/44) and whitefish $64 \%$ (14/22)) and 3,5-dichlorophenol (peamouth chub detection frequency $41 \%$ (18/44), and whitefish $68 \%$ (15/22) Table 26).

Pentachlorophenol, 4-monochlorophenol and a range of dichlorophenols were commonly detected at trace levels in both peamouth chub and whitefish muscle from the upper Fraser (Hansard and Marguerite) and Thompson reaches (Figure 16, Table 28 and Table 29). The presence of these compounds in tissues probably reflects contamination by nearby wood treatment facilities in these reaches.

Chlorinated syringaldehydes (CSA) were detected in peamouth chub muscle tissues upstream and downstream of pulp mill effluent sources. On both the Fraser and Thompson basins, fish collected upstream of mill effluents showed elevated 2,6-CSA while downstream, the dominant syringaldehyde was 2-CSA. In the whitefish muscle from the Thompson reaches, in particular, levels of total chlorophenols were dominated by 2,6-CSA. Chlorosyringaldehydes can result from bleaching of hardwood pulp (Brumley et al. 1996), and the feed-stock for the Prince George and Kamloops mills producing this result may bear further examination. Other sources may be important, since CSAs were also detected in bed sediments in the North Thompson, upstream of pulp mill sources (Brewer et al. 1998).

As with muscle tissue analyses, chlorophenolic detections in liver tissues were sporadic but were of somewhat higher concentration (Table 26). Total chlorophenol levels were uniformly low (average < $3 \mathrm{ng} / \mathrm{g}$ w.w.) in the far upper basin reaches at Hansard and Woodpecker, and only slightly higher than in muscle. Total chlorophenol concentrations in liver were 5-15x higher downstream of Quesnel, relative to upstream sites. Through the lower Fraser reaches, total chlorophenols in liver ranged from 8 to $14 \mathrm{ng} / \mathrm{g}$ w.w. , and showed little variation either between sites or between species (Figure 17, Table 30, Table 31). Of particular interest in the lower Fraser reaches are elevated levels of 4-
chlorophenol in peamouth liver (2.5-5.6 ng/g w.w., Table 30, Figure 17), the source of which is unknown.

Total chlorophenols in both muscle and liver follow similar spatial patterns, being lowest in the upper reaches and in the lower Fraser, highest in the middle Fraser downstream of mill effluent discharges. The exception to this general trend is in whitefish muscle, where total chlorophenolics in the North Thompson were higher than those measured in the Thompson downstream of Kamloops (Figure 16). This is likely a function of the upstream movements of exposed fish to the North Thompson reach and the very high dilution of effluents in the passage through Kamloops Lake to the Thompson reach.

The spectrum and pattern of chlorophenolics in muscle and liver were quite different in both peamouth and whitefish (Figure 16, Figure 17). At least two factors likely contribute to the differences. Chlorophenolic concentrations were low and much of the apparent difference between tissues may be attributable simply to differences in sample detection limits related to tissue lipid-levels and available tissue mass for analysis.

## Chlorophenolics in Bile

Results are summarized in Table 32 and Table 33 and presented graphically in Figure 18 and Figure 19. Individual analyses of bile composites were, in some cases, highly variable. For example, in peamouth chub bile from Woodpecker and Marguerite in 1994 total chlorophenolics ranged from 32-420 and 482-2500 ng/mL, respectively. This high variability between samples was also found in whitefish from the Athabasca River (Owens et al. 1994b), and doubtless results from the rapid metabolic turnover coupled with highly variable exposure conditions.

Total chlorophenols levels in bile from whitefish and peamouth chub from the upper Fraser in 1994 ranged from a low of $36.8 \mathrm{ng} / \mathrm{g}$ w.w. in the Hansard reach to $3680 \mathrm{ng} / \mathrm{g}$ w.w. in whitefish immediately downstream of mill discharges from Quesnel (Table 32). Levels at Woodpecker were low compared to Marguerite, a result probably related to the proximity of the sampling area to the effluent discharges. Concentrations in samples from these same reaches in 1995 were both considerably lower than measured in 1994, and in whitefish were nearly identical in the Woodpecker and the Marguerite reaches (Figure 19). Although total concentrations declined, component profiles for the two sets of bile analyses (Figure 18, Figure 19) indicate the similar patterns in contaminants over the two sampling years.

Totals for both whitefish and peamouth chub in the Hansard reach in both 1994 and 1995 were dominated by chlorophenols, particularly PCP and 2,3,4,6TeCP (peamouth chub) and 2,4/2,5dichlorophenol (whitefish), suggestive of release from wood preservation (Figure 18, Figure 19). That the contamination is consistent with emissions from wood processing and treatment is corroborated by other analyses, particularly the presence of heptachlorodioxins in peamouth chub muscle and resin acids in bile, data for which are considered elsewhere in this report. The relatively high dichlorophenol level in whitefish and its absence in peamouth chub may suggest metabolic transformations in the whitefish not occurring the peamouth chub.

The influence of municipal and industrial discharges from Prince George is evident in the striking increase in the spectrum of detected contaminants, particularly chlorophenols and pulp-mill related chloroguaiacols in fish collected downstream. Downstream of both Prince George and Quesnel in the Woodpecker and Marguerite reaches, total chlorophenols in bile of both species are dominated by characteristic pulp mill chloroguaiacols, with relatively minor contribution from a range of chlorophenols in the Woodpecker reach (Figure 18, Figure 19). A marked increase in total chlorophenols occurs in the Marguerite reach, owing to elevated levels of both 4,5- and 3,4,5chloroguaiacols.

In the Thompson reaches, total chlorophenolics in bile showed remarkable similarity in both average concentration and in congener composition between 1994 and 1995 (Table 32 and Table 33). In whitefish, total chlorophenols in the North Thompson were about $50 \%$ of those in the Thompson downstream of Kamloops. At the upstream reach, the dominant chlorophenolic was 2,4/2,5dichlorophenol while downstream of the Kamloops mill, the dominant congener was 5-chloroguaiacol (Figure 18, Figure 19). Totals in peamouth chub bile were low (average $\leq 27 \mathrm{ng} / \mathrm{g}$ w.w.) in both years in the Thompson basin. The contrast between peamouth chub collected from upstream and downstream of the pulp mill is apparent in detectable levels of 5-chloroguaiacol in fish downstream of the mill discharges.

Chlorophenol totals in starry founder and peamouth chub bile collected in the estuary reaches in 1995 ranged from 30 to $144 \mathrm{ng} / \mathrm{g}$ w.w. (Table 33). In both species, the elevated levels of PCP and 2,4,6trichlorophenol indicate clearly contamination by wood treatment facilities (Figure 19). Minor contributions of 4,5-dichloroguaiacol and 5-chloroguaiacol suggest contamination by both local pulp and paper mill effluents in the lower Fraser and releases from the upstream mills.

## Comparison to Guidelines

There are presently no Canadian tissue residue guidelines for protection of aquatic life with which to compare these results. A particular concern with respect to human use of fish species is tainting of flesh, and BC Environment (Warrington 1993) have established congener-specific criteria for chlorophenols related to tainting of fish flesh for flavour impairment. Criterion levels typically range from $10 \mu \mathrm{~g} / \mathrm{g}$ for 2 -chlorophenol to $80 \mu \mathrm{~g} / \mathrm{g}$ for 2,3-dichlorophenol. An exception is 2,4-DCP, at 0.2 $\mu \mathrm{g} / \mathrm{g}$ (Warrington 1993). Maximum concentrations in all tissues were, in most cases, several orders of magnitude below these criteria for most congeners (Table 34).

## Comparison with Previous Studies

Previous studies conducted in the 1980s measured chlorophenols in fish tissues in the Fraser River estuary, with a focus on chlorophenols resulting from wood preservation in a period prior to deregistration of PCP (Birch and Shaw 1995, Carey and Murthy 1988, Rogers et al. 1990, Rogers and Hall 1987, Rogers et al. 1992), with generally less effort on other related compounds such as the guaiacols and other more "exotic" congeners. Concentrations measured in starry flounder in 1994 are
far below levels recorded in starry flounder collected in the North Arm in 1988 (Birch and Shaw 1995). For example, total chlorophenols in composites of flounder liver in 1988 ranged from 108 to $527 \mathrm{ng} / \mathrm{g}$ w.w., compared with the $8-12 \mathrm{ng} / \mathrm{g}$ w.w. measured in this study.

Similar declines are evident in fish from reaches upstream, due both to changes in pulp mill bleaching process and reduction in use of chlorophenate wood preservatives. Sampling of trout in the Thompson reach in 1990 showed PCP levels in muscle of $10-33 \mathrm{ng} / \mathrm{g}$ w.w. (Norecol 1997), compared to maximum total chlorophenolic levels measured here of $3.4 \mathrm{ng} / \mathrm{g}$ w.w. (Table 26). MacDonald et al. (1997) found no detectable chlorophenolics in tissues from sturgeon captured in the vicinity of Prince George in 1994. Chlorophenolic levels in juvenile chinook salmon sampled in 1988 by Rogers et al. (1989) from sites in the upper Fraser were typically 10 to 100x higher than those found in this study.

Measurements of chlorophenolics in bile from mountain whitefish from the Wapiti River in Alberta were made by Swanson et al. (1993). Although only one sample was analyzed, the levels declined from about $10,000 \mathrm{ng} / \mathrm{ml}$ w.w. near the outfall to $1000 \mathrm{ng} / \mathrm{ml}$ w.w. at a point 230 km downstream. Levels measured here were far lower (max $3,600 \mathrm{ng} / \mathrm{g}$ w.w.) than those measured in Alberta, likely due to the much higher effluent dilution in the Fraser River.

Declines in chlorophenolics in tissue beyond those measured here are likely. Derksen (1997) presented a history of decline of 3,4,5-trichloroguaiacol after the shift to $\mathrm{ClO}_{2}$ substitution at the Northwood Mill, in Prince George. In 1988, 3,4,5-TCG in effluent averaged $41,000 \mathrm{ng} / \mathrm{g}$ w.w., declining to $3,400 \mathrm{ng} / \mathrm{g}$ w.w. by 1993 and was below detection in 1996 sampling.

## Chlorophenolics Summary

Chlorophenolic levels in muscle and liver tissues were at low concentrations in fish throughout the basin. Detections of chlorophenols, particularly PCP, were primarily attributed to contamination by nearby sawmills and wood treatment facilities in reaches in the upstream Fraser and estuary and in the Thompson basin. Elevated guaiacols, particularly 4,5-dichloroguaiacol in liver tissue in the Marguerite reach, was associated with exposure to BKME. The utility of bile as a matrix for chlorophenolic measurements and as an indicator of exposure was demonstrated clearly in comparison to muscle or liver analyses. Dramatic declines in total chlorophenol concentrations are apparent throughout the basin when compared to available historic data.

## Resin Acids in Bile

## Background

About 10 naturally-occurring resin acids are commonly released from wood processing and pulping operations (Figure 20). Together, they contribute a large proportion of the acute toxicity of these effluents (Holmbom and Lehtinen 1980, Leach and Thakore 1975, McLeay and Associates Ltd
1986). Resin acids loads and the patterns of resin acid components depend on a number of factors, including effluent process, mill operation, and tree species (Taylor et al. 1988). Like many organic components of wood pulp, the parent compounds may react during chlorine bleaching to produce chlorinated resin acids which are considerably more hydrophobic, toxic and persistent than their parent compounds (McLeay and Associates Ltd 1986). In particular, mono-and dichlorohydroabietic acid have been associated with a variety of sublethal toxicological symptoms in fish (Kennedy et al. 1995, Tana 1988).

The principal route of accumulation of resin acids, as of many contaminants in fish, is through absorption from ambient waters through unprotected skin, particularly the gill tissues (Randall et al. 1998). Resin acids absorbed into the bloodstream are rapidly conjugated in the liver, sequestered and excreted in the bile (Oikari et al. 1984). Residence time in fish is short, with resin acids levels in bile typically dropping to below-detection in fewer than four days after exposure (Niimi and Lee 1992). Biliary contaminant concentrations vary with dose and duration of exposure, but may reach levels many thousands or tens of thousands of times ambient concentrations (Oikari and Niittayla 1985). In chronic effluent exposures, there is a linear relationship between ambient and bile concentrations of resin acids (Oikari and Niittayla 1985). Rapid uptake, high measurable concentrations and a short half-life are characteristics which have led to application of resin acids in bile as indicators in fish of recent exposure to wood-waste effluents (Leach and Thakore 1975, Oikari and Holmbom 1986, Oikari and Kunnamo-Ojala 1987).

## Resin Acids Results

Bile was analyzed for eight naturally-occurring and two chlorinated resin acids (Table 35). Samples in 1994 were collected from peamouth chub from two sites in the upper Fraser and two sites in the Thompson basin, and from whitefish at Woodpecker. In 1995, bile samples were collected from peamouth chub at 7 sites, whitefish from 2 sites in the upper Fraser and from flounder at two estuary sites. Analytical results for 1994 and 1995 are summarized in Table 36 and Table 37, respectively.

In 1994, the highest total resin acid concentrations (average $425 \mu \mathrm{~g} / \mathrm{mL}$ ) were from peamouth chub collected in the Hansard reach (Figure 21, Table 36). These levels are almost certainly the result of local contamination from nearby saw mills at Upper Fraser and Sinclair Mills. Evidence from other analyses, such as elevated PCP in bile and heptachlorodioxins in tissues, provide additional support. Potential contamination from these sources has been noted in previous studies in the area (Rogers et al. 1989). Sampling in 1995 focussed on upstream areas nearer McBride where there are fewer major wood processing facilities. The change is reflected in 1995 total resin acid concentration in bile, which was roughly $1 / 100^{\text {th }}$ the levels measured in the Hansard reach in 1994. Contaminant profiles in the two years are also very different, with 1994 samples being dominated by abietic and sandaracopimaric acids (together comprising approximately $65 \%$ of the total), and by abietic and dehydroabietic acid in 1995 (Table 36 and Table 37).

As a consequence of site selection for the Hansard reach in 1994, levels in peamouth downstream of the Prince George effluents ( $11.8 \mu \mathrm{~g} / \mathrm{mL}$ ) were only a fraction of the upstream "reference" area. In

1995 however, levels were low in the Hansard $(5.7 \mu \mathrm{~g} / \mathrm{mL})$ reach with a doubling in concentration downstream of Prince George at Woodpecker ( $12.6 \mu \mathrm{~g} / \mathrm{mL}$; Table 37). In contrast, total resin acids in whitefish at Hansard ( $36.8 \mu \mathrm{~g} / \mathrm{mL}$ ) were higher than at Woodpecker $(24.5 \mu \mathrm{~g} / \mathrm{mL})$, indicating either a local source or recent movements from downstream areas. Concentrations in peamouth chub from the Woodpecker reach showed striking similarity between years, probably due to both the constant distance from and uniformity of effluent discharges.

Chlorinated resin acids, measured as mono- and dichlorodehydroabietic acid, were at low levels (max $0.10 \mu \mathrm{~g} / \mathrm{mL}$ ) in bile samples downstream of Prince George (Table 36 and Table 37) and in the estuary reaches (Table 37). An increase in chlorinated resin acids in the Fraser River resulting from of exposure to upstream BKME was expected but not observed. Similar increases were expected in samples from the Thompson basin but were also not observed (Table 37).

Resin acids in the Thompson basin were also highest in the North Thompson "reference" reach in 1994 and 1995, upstream of discharges from the pulp mill at Kamloops (Table 36 and Table 37). Local sawmills on the North Thompson River are probable sources. Some of the resin acids may be from downstream sources, since the presence of chlorinated resin acids in bile from the North Thompson reach in 1994 (Table 36) suggests fish movement from areas downstream of Kamloops. Data for dioxin and furan levels in fish from the North Thompson in 1994 further suggest movement of these 'resident' fish and the possibility that upstream fish were exposed downstream of the Kamloops pulp mill.

In both the Fraser and Thompson rivers, total resin acids in bile in peamouth chub were dominated by abietic and dehydroabietic acid, which on average represented $62 \%$ of the total resin acid burden in 1995. The dominance of other components varied through the basin, but in all cases as few as 4 resin acids typically constituted greater than $85 \%$ of the total resin acids. In reaches where evidence suggested uptake from local sawmill effluent sources (Hansard 1994, Main Arm and North Arm 1995), other less common components such as sandaracopimaric acid (SAA) tended to be relatively elevated. This is the case particularly in the North Arm, where SAA comprised $20 \%$ in peamouth chub and $30 \%$ in flounder of the total resin acids.

With the exception of pimaric, isopimaric, and the two chlorinated resin acids, concentrations of most resin acids in whitefish were similar to those of peamouth chub. For those four exceptions, levels were slightly higher in whitefish compared to peamouth chub. Resin acid profiles were similar in both species, reflecting both a similar source environment and similar uptake mechanisms.

Total resin acids in bile of starry flounder from the estuary reaches ranged from 2 to $29 \mathrm{ng} / \mathrm{g}$ w.w., with a clear dominance of abietic, dehydroabietic, isopimaric and sandaracopimaric acids (Table 37). Average levels in the Main Arm were roughly 3x those in the North Arm, in contrast to peamouth collected in these same reaches. In peamouth, average levels in the North Arm were roughly twice those in the Main Arm.

High resin acid concentrations in peamouth chub and mountain whitefish bile at sites upstream of mills on both the Fraser and Thompson basins are unlikely to reflect natural background levels. In view of the very short half-life of resin acids in fish bile and tissues, the measured concentrations probably reflect recent exposure to runoff or effluents from saw mills in the vicinity of the sample reaches. Collections at the Hansard site in particular, were near the Northwood saw mill at Upper Fraser, and it is probable that peamouth chub move easily between areas upstream and downstream of the facility.

Some concentrations measured in this study are comparable to near-field levels found by Oikari (1986), who measured resin acid levels in roach (Rutilis rutilis) at varying distances downstream of pulp and paper mill effluents. Within 9 km of the effluent discharge, bile concentrations dropped about $90 \%$, from $240-500 \mu \mathrm{~g} / \mathrm{mL}$ to $40-60 \mu \mathrm{~g} / \mathrm{mL}$ for the four measured resin acids. Similar results were found by Tavendale et al. (1996) for effluent-exposed goldfish, where levels dropped almost $90 \%$ in a distance of only 2 km .

Most of the sampling in this study was in areas distant from effluent discharges, and, as such, measured levels in Fraser River fish tend to be far below those concentrations measured by Oikari or Tavendale. Sampling in the Hansard area in 1994 was an exception, and the proximity to a resin acid source is reflected clearly in the high resin acid levels.

## Resin Acid Summary

Results of these analyses show that fish are being exposed to waterborne resin acids in some parts of the basin but that the zone of greatest influence is probably restricted to near-field areas. Profiles of resin acid components show that the highest levels were due to non-chlorinated compounds, probably originating from nearby sawmills, such as in the Hansard reach in 1994 and in the North Thompson and Fraser River Main and North arms in 1994/1995. Contributions from pulp and paper processing are apparent in samples downstream of Prince George and Quesnel in both 1994 and 1995. Chlorinated resin acids were detected only at low levels and only downstream of BKME.

## Polychlorinated Biphenyls (PCBs)

## Background

Polychlorinated biphenyls are highly stable, lipophilic and relatively non-volatile organochlorine compounds which are nearly ubiquitous in the environment. Low chemical reactivity and resistance to degradation at high temperature made them ideally suited to their principal use as dielectric fluids in high-voltage transformers and capacitors (Strachan 1988). Prior to their removal from the North American marketplace, PCBs saw wide application from hydraulic fluids to wood sealants to flame retardants (Safe 1994, Strachan 1988) to microscopy.

On the naked biphenyl molecule (Figure 22) are 10 potential sites for chlorine substitution, with a total of 209 possible different congeners. Each of these congeners are assigned numeric designations
in a system devised by Ballschmitter and Zell (1980) and subsequently adopted by the International Union of Pure and Applied Chemistry (IUPAC). Tables showing the IUPAC designation and corresponding chemical structure are presented in a number of publications such as McFarland and Clarke (1989).

The most common commercial formulations in North America were the various Arochlors produced by the Monsanto Corporation, each containing upwards of 80-100 individual congeners (Figure 23). In the environment, de-chlorination (Bedard and May 1996), volatilization, selective condensation (Wania and Mackay 1996), congener-specific metabolism and elimination (Brown 1994, Coristine et al. 1996, Safe 1994), and differential partitioning to environmental compartments (Turrio-Baldassari et al. 1993) occur, all of which may serve to alter the measured proportions of each congener from that of the original formulation. In all, about 100 PCB congeners have been detected in environmental samples (Schulz et al. 1988). Comprehensive analysis of congener patterns may provide information about contaminant sources and can yield clues to mechanisms of congener metabolism (Bright et al. 1995, Brown 1994, Elkus et al. 1994, McFarland and Clarke 1989, Pim et al. 1997).

The acute toxicity of most PCB congeners is low, and the modes of action in situations in which detrimental effects have been seen are poorly understood (Safe 1994). Exceptions are those PCB congeners with chlorine substitution in the ortho positions, a structure in which the molecule assumes a planar configuration and mimics the action of the highly toxic 2,3,7,8-TCDD (McFarland and Clarke 1989). Although the four ortho or coplanar PCBs (cPCB) constitute less than $0.1 \%$ of the total content of commercial PCB mixtures, they probably contribute most of the dioxin-like toxicity (Brown et al. 1995). While some environmental contamination by cPCBs is a consequence of commercial PCB release, Brown et al. (1995) suggested that cPCBs detected in environmental samples result from combustion sources. Because of their high toxicity, the three cPCBs, IUPAC numbers 77, 126 and 169, were measured at levels to $1 \mathrm{pg} / \mathrm{g}$ w.w. wet weight, a detection limit an order of magnitude lower than that of the non-ortho-substituted congeners.

## PCB Analysis Results

In 1994 and 1995, muscle and liver tissues from peamouth chub at all 11 reaches, from whitefish at 7 upstream reaches and from starry flounder at two reaches in the estuary were analyzed for a total of 84 individual non-ortho PCB congeners or co-eluting congener groups and three coplanar PCBs (Table 38). Two additional sites in the lower Fraser, Mission and Barnston, were sampled in 1994 but not in 1995. Sample detection limits vary with sample weight but were near $1 \mathrm{ng} / \mathrm{g}$ w.w. for the non-ortho PCB congeners and $1-2 \mathrm{pg} / \mathrm{g}$ w.w. for the coplanar PCBs.

## Total PCBs in Tissues

Summary total PCB ( $\Sigma \mathrm{PCB}$ ) concentrations in fish muscle and liver are presented in Table 39 and Table 40. Basin-wide comparisons of total PCBs in fish tissues are shown in Figure 24.

Total PCBs in fish muscle in both 1994 and 1995 averaged less than $6 \mathrm{ng} / \mathrm{g}$ w.w. wet weight at most locations in the upper basin and showed little difference between peamouth chub and whitefish (Figure 24). In all cases, levels in 1995 were lower than in 1994. In peamouth chub muscle in the Marguerite reach in particular, $\Sigma$ PCB concentration in 1994 averaged $16.6 \mathrm{ng} / \mathrm{g}$ w.w., but dropped to $1.8 \mathrm{ng} / \mathrm{g}$ w.w. in 1995 (Table 39). The factors causing this elevated level in 1994 will be considered later in this section. Total PCBs in muscle were higher in the lower Fraser compared to upstream areas (Figure 24), with concentrations in the Agassiz reach in 1994, for example, in excess of $25 \mathrm{ng} / \mathrm{g}$ w.w. in both peamouth chub and whitefish (Table 39, Figure 24).

Two features of $\Sigma$ PCBs in fish muscle in the lower Fraser are of particular note. First, the high concentrations measured in whitefish at Agassiz were very similar between years, compared to sites elsewhere in the basin. In peamouth chub in the same reach, however, levels were much lower in 1995 ( $10.5 \mathrm{ng} / \mathrm{g}$ w.w.) than in 1994 ( $27.4 \mathrm{ng} / \mathrm{g}$ w.w.) (Table 39). Second are the elevated $\Sigma$ PCBs in peamouth chub muscle in both the Agassiz and Mission reaches in 1994, suggesting contamination either from a local source or local process resulting in increased PCB accumulation (Figure 24).

In the two Thompson reaches, $\Sigma$ PCB concentrations in muscle samples were similar both upstream and downstream of Kamloops and showed no difference between years (Figure 24). Levels in peamouth chub were similar to the upper Fraser reaches (3.4-5.5 ng/g w.w.) and somewhat lower than levels in whitefish ( $7.2-12.2 \mathrm{ng} / \mathrm{g}$ w.w.). Measured $\Sigma$ PCBs in whitefish were higher in the Thompson than in the upper Fraser sites in both 1994 and 1995 (Table 39).

The basin-wide pattern in $\Sigma$ PCB concentrations in peamouth and whitefish liver mirror those in muscle, with low levels in the upper Fraser reaches and elevated levels in the estuary (Table 40, Figure 24). Differences in lipid content between the upstream and estuary reaches may account for the pattern, but levels in the lower Fraser River and estuary remain elevated, even when concentrations are normalized to tissue lipid content (Figure 25). Concentrations as high as $271 \mathrm{ng} / \mathrm{g}$ w.w. were measured in peamouth chub liver from the North Arm in 1995. However, in both 1994 and 1995 the average level through the estuary was much lower, about $170 \mathrm{ng} / \mathrm{g}$ w.w. (1994/1995 average, Barnston, Main Arm, North Arm reaches: Table 40). Levels in peamouth chub liver through the upper Fraser reaches averaged $23-25 \mathrm{ng} / \mathrm{g}$ w.w. in 1994 and $11-14 \mathrm{ng} / \mathrm{g}$ w.w. in 1995. Total PCBs in liver from Thompson reaches were similar to those in the upper Fraser basin, with levels of $21-27$ and $27-36 \mathrm{ng} / \mathrm{g}$ w.w. in the North Thompson and Thompson, respectively, with only minor differences between years.

In whitefish liver, $\Sigma$ PCBs, with the exception of Agassiz, ranged from 3.2 to $16.2 \mathrm{ng} / \mathrm{g}$ w.w. (Table 40) and were similar or slightly (roughly 1-2 ng/g w.w.) higher in 1995 than in 1994. At Agassiz in 1994, $\Sigma$ PCB in the single composite was $22 \mathrm{ng} / \mathrm{g}$ w.w.; in 1995 the measured level averaged $9 \mathrm{ng} / \mathrm{g}$ w.w. (Table 40).

Total PCBs in starry flounder tissues were elevated in the North Arm relative to the Main Arm, doubtless a consequence of the higher density of industrial discharges and storm drains to this reach (Brewer et al. 1998). Levels were roughly 10 times greater in liver tissue than in muscle in both 1994
and 1995 , probably reflecting the relative lipid content of the two tissues. Levels in starry flounder in 1995 were only half those measured in 1994. In comparison with peamouth chub collected from the same reaches, $\Sigma$ PCBs in starry flounder were about $30 \%$ of levels in peamouth.

Inter-species differences in PCB burden are apparent, primarily reflecting the relative tissue lipid levels among the three target species. This is particularly true in the comparisons of peamouth and whitefish liver tissues, where levels were 2-10 times higher in peamouth than in whitefish. As a consequence, levels tended to be highest in peamouth chub when compared to the other sampled species, either whitefish or flounder, in each particular sampling reach. Other factors, such as habitat selection and feeding preferences are probably of relatively minor importance in comparisons since when concentrations are normalized, interspecies differences all but disappear in 1995 data (Figure 25). In areas of sediment contamination, the close substrate contact of demersal fish may encourage uptake of pollutants such as PCBs, as for example, is seen in starry flounder muscle from the Main and North arms (Figure 25).

## PCB Congener Profile Patterns

Of the 84 target PCB congeners or co-eluting sets, up to 78 in 1994 and 83 in 1995 were detected in at least one sample. In 1995, in particular, all 83 congeners were detected in a single peamouth liver composite from the Fraser River north arm.

Plots of the total numbers of detected congeners are presented in Figure 26. In both peamouth chub liver and to a lesser degree in muscle, there is a clear step-increase in the number of detected congeners from samples upstream compared to samples downstream of Marguerite (Figure 26). In both peamouth chub and whitefish from the two reaches in the Thompson basin, the higher number of congeners in muscle were detected in the North Thompson reach, while the higher number in liver were found downstream of Kamloops (Figure 26).

The combination of a low tissue lipid content and relatively small tissue mass in whitefish liver is particularly evident in numbers of detected congeners. Total numbers of congeners detected in whitefish liver were lower than in peamouth chub, ranging from 5 to 25 and showed little variation throughout the basin (Figure 26).

A plot of the detection frequency of the 84 congeners over all samples is presented in Figure 23. The pattern, when compared to proportional representation of congeners in commercial Arochlors, suggests contamination by both Arochlor 1260, contributing the hexachloro- and higher substituted members, and Arochlor 1254, which may account for the tetra- and pentachloro-substituted congeners. Five congeners, PCB numbers 153, 149, 138/163/164 and 180, which were each detected in more than $90 \%$ of both muscle and liver samples are dominant components in Arochlors 1254 and 1260. In addition, the congeners not detected in any muscle tissue sample, PCBs 207, $205,198,130,19$, or any liver sample, PCBs 198, 189, 130, 46, are either absent or at low levels in the Arochlors 1254 and 1260.

Concentration profiles of individual PCB congeners measured in muscle and liver for all three fish species are presented in Figure 27 to Figure 30. The similarity in source contamination and a common process of accumulation is evident in the very similar profiles across both tissues, reaches and species. While minor PCB components differ between sites due to small variations in sample detection limits, dominance of particular congeners is evident throughout the basin. Consistently high peaks of PCB numbers 101/90, 110, 118, 138, 149 and 153 are apparent in nearly all plots, corresponding again to dominant congeners in Arochlors 1254 and 1260 (Figure 23). In the entire set of analyses in each of the sampling years, these 6 congeners alone accounted for about $43 \%$ of the $\Sigma$ PCBs. Dominance by these particular PCB congeners in environmental samples is common due partly to their relatively high representation in common commercial Arochlors and the particular resistance of these components to chemical breakdown or metabolic transformation (Niimi 1996).

Trace concentrations and spotty detections of PCB congeners in peamouth chub and whitefish tissues from the Nechako and Hansard reaches (Figure 27 to Figure 30) reflect the low level of municipal and industrial activity within these reaches. At least some of the contamination within these reaches is probably due to atmospheric transport from distant sources. Concentrations of individual congeners in muscle of both peamouth chub and whitefish from these two reaches were low, usually less than $0.9 \mathrm{ng} / \mathrm{g}$ w.w., and dominated by PCBs 138, 149 and 153 in addition to traces of tetrachlorinated congeners, PCBs 33 to 74. A similar pattern was seen in liver tissues from both peamouth chub and whitefish, although the higher lipid content of the peamouth chub liver resulted in accumulation of both much higher levels and wider spectrum of congeners than was found in whitefish (Figure 29, Figure 30).

In all tissues collected in the mainstem Fraser River, both the spectrum and concentrations of PCB congeners increased downstream to the estuary reaches. Profiles of PCB congeners in peamouth chub muscle and liver in 1994 suggest a source in the vicinity of Prince George rich in the tetrachlorinated members (Figure 27, Figure 29) which was apparently absent in 1995.

Derksen (1997 DRAFT) measured PCB congeners in a number of industrial effluents in the Prince George and Quesnel area through 1994 and 1995. While PCB congeners were low to nondetectable in most cases, the pattern of congeners in effluents from the Northwood Pulp and Paper Mill in Prince George in November 1994 is of particular interest. The profile and, in particular, the predominance of tetrachlorinated congeners bears striking resemblance to that found in fish tissues downstream at both Woodpecker and to some extent Marguerite (Figure 31). In subsequent sampling of this same effluent (October 1995), PCBs were low to non-detectable, consistent with tissue congener patterns measured in 1995. Should this scenario be correct, it is clear that relatively local contamination sources are important in determining tissue PCB burden and that effluent changes may translate to rapid changes in uptake by biota.

## Coplanar PCBs in Tissues

Coplanar PCBs (PCBs 77, 126 and 169) show patterns similar to those of the total non-ortho PCBs (Table 41, Table 42). Tissue concentrations of all three compounds were, in general, lowest in the
headwater sites and highest in the lower Fraser (Figure 32). PCB 77 had the highest levels, with average concentrations in peamouth muscle up to $34 \mathrm{pg} / \mathrm{g}$ w.w., and in peamouth liver to over 400 $\mathrm{pg} / \mathrm{g}$ w.w. from reaches in the lower Fraser (Table 41, Table 42). The highest levels in both peamouth and starry flounder liver were measured in the Fraser River North Arm.

Concentrations in 1994 were generally higher than in 1995 in most reaches. In some cases, such as in PCB 77 and 126 in peamouth muscle downstream of Marguerite, and for PCB 169 in starry flounder liver, the between-year difference is dramatic (Figure 32). Some areas showed remarkable constancy between years, such as is evident in peamouth liver from the North Arm (Table 42)

Total concentrations of the measured coplanar PCBs were clearly related to total concentrations of non-ortho PCBs (Figure 33), suggesting that there is probably no preferential accumulation of these toxic congeners.

## PCB Toxic Equivalents

To estimate the toxicological hazard of the measured PCB residues, 2,3,7,8-TCDD toxic equivalents were calculated using mammalian, fish and bird TEFs for non-ortho and coplanar congeners (Table 43) published by van den Berg et al. (1998). Fish tend to be the least sensitive to dioxin-like activity, mammals considerably more so and birds are most sensitive, as evidenced by the higher TEFs for each taxon. The TEQs calculated here are likely to be lower than the true total toxicity, since some of the congeners for which TEFs have been derived (particularly PCB 81), were not measured.

Average within-reach TEQs for peamouth chub and whitefish muscle in the upper Fraser and Thompson reaches were generally less than $1 \mathrm{pg} / \mathrm{g}$ w.w. for all species and TEQ classes (Figure 34). Particular exceptions were in TEQ $_{\text {PCB-Bird }}$ in the lower Fraser reaches in 1994, where levels greater than $2 \mathrm{pg} / \mathrm{g}$ w.w. were measured. TEQs in muscle were higher in 1994 than in 1995 for all three fish species (Table 44).

TEQs in liver were higher than those in muscle. Both $\mathrm{TEQ}_{\text {PCB-mammal }}$ and $\mathrm{TEQ}_{\text {PCB-Fish }}$ in liver from all three fish species were generally less than $1 \mathrm{pg} / \mathrm{g}$ w.w. throughout the basin (Figure 35, Table 45). Reaches in the lower Fraser were the exceptions, with TEQ $_{\text {PCB-mammal }}$ in 1994 and 1995 of 3.6-6.8 pg TCDD/g in peamouth liver, and surprising consistency between years (Table 45). Particularly extreme values were seen in the more sensitive $\mathrm{TEQ}_{\text {PCB-Bird }}$ values, which in 1994 ranged from 5-23 pg TCDD/g in peamouth liver. Levels in these reaches showed sharp declines in 1995, due to lower measured levels of PCB 77 (Table 42).

As a result of recent improvements in pulp and paper mill effluent quality, the contribution of PCB congeners to dioxin-like toxicity in tissues is becoming increasingly important. In reaches where both PCB congeners and dioxin/furan congeners were measured in 1994, the contribution of PCBs to the total estimated health hazard due to dioxin-like activity in some lower Fraser reaches exceeded that due to dioxin and furan congeners (Figure 36). In peamouth chub liver from the lower Fraser, Thompson and North Thompson reaches, in particular, total TCDD TEQs (both TEQ ${ }_{P C B-\text { Bird }}$ and
$\mathrm{TEQ}_{\text {Dioxin-Bird) }}$ ) ranged from 25 to 40 pg TCDD/g tissue (Figure 36). In reaches where dioxins and furans continue to be elevated, such as in the Thompson and North Thompson, PCBs continue to comprise a relatively small proportion of the total TEQs.

## Comparison with Historical PCB Levels

Comparative historical data for PCBs in peamouth chub and whitefish in the upper basin are sparse, and, where present, represent a period when contamination was more prevalent due to the active use of PCBs. Total PCBs were measured in mountain whitefish downstream of pulp and paper mills on the Athabasca River in 1992 (Pastershank and Muir 1996). Levels measured in upper Fraser are in close agreement or lower than the $6-10 \mathrm{ng} / \mathrm{g}$ w.w. measured in upstream reference areas on the Athabasca River where PCB contamination is most probably due to atmospheric transport. Lower PCB levels in whitefish muscle have been reported in the Peace River, where concentrations averaged about $1.3 \mathrm{ng} / \mathrm{g}$ w.w. in 1994 (Pastershank and Muir 1996).

Somewhat more comparative data are available for locations in the estuary (Table 46), and without exception, these data document a steady decline in tissue PCBs from the early 1970s to present day. Even with due consideration to differences in analytical methods, the difference between maximal measured levels in peamouth chub muscle in 1973 of greater than $500 \mathrm{ng} / \mathrm{g}$ w.w. compared to the maximum of $23 \mathrm{ng} / \mathrm{g}$ w.w. measured in $1994 / 5$ is positive evidence of the success of control measures. Levels in peamouth chub liver have declined from the maximum of $1050 \mathrm{ng} / \mathrm{g}$ w.w. measured by Swain and Walton (1989), but even now show concentrations up to $270 \mathrm{ng} / \mathrm{g}$ w.w. (Table 46).

Measurements in this study are among the first data on environmental levels of coplanar PCBs in fish tissues in the Fraser River, data which are long overdue (Rogers et al. 1992). Rantalainen et al. (1998) measured coplanar PCBs in whole fish composites of long-nose sucker from sites in the lower Fraser and estuary in 1991 and 1992. Their sample sizes were very small (1-3), but the overall pattern of contamination in the area was similar to that seen in the FRAP program - lowest levels in the Agassiz area, and the highest levels in the North and Main Arms (Table 47). Variability among sites within the equivalent FRAP reaches was high, with fish from the upper North Arm (Burnaby) having 2-3x higher levels that fish from the downstream (MacDonald Beach) site.

In comparison with other west coast sites, levels of the three coplanar PCBs in the Fraser estuary are probably quite low. Petreas et al. (1992) measured coplanar PCBs in flounder in the Sacramento area and found levels in muscle of 80,27 and $3.5 \mathrm{pg} / \mathrm{g}$ w.w. for PCBs 77, 126 and 169. Levels measured in this study are, with the exception of those in peamouth chub liver tissue, well below those of Petreas et al. (1992).

The FRAP data show clearly both the presence of coplanar congeners in biota and the toxicological significance of the residues. Historical data from the Fraser River are lacking, but the close relationship between coplanar PCBs and total PCBs (Figure 33) suggests that levels of these components will have declined over time in concert with drops in other PCB components.

## Comparison to Guidelines

In terms of existing Health and Welfare Canada maximum residue guidelines for consumption by humans, total PCB levels measured in fish in 1994 and 1995 fall far below the current $2 \mu \mathrm{~g} / \mathrm{g}$ (2000 ng/g w.w.) limit.

The recent CCME TEQ ${ }_{P C B}$ guideline level of 0.79 pg TCDD TEQ $/ \mathrm{g}$ wet weight has been set for protection of wildlife consumers of aquatic biota (CCME 1999). With the exception of $\mathrm{TEQ}_{P C B-B i r d}$ values in the lower Fraser, this guideline was not exceeded in fish muscle (Table 44). Exceedence was common in peamouth liver throughout the basin for both $\mathrm{TEQ}_{\text {PCB-Bird }}$ and $\mathrm{TEQ}_{\text {PCB-mammal }}$ values, particularly in 1994 (Table 45).

## PCB Summary

PCB concentrations throughout the basin were low in both 1994 and 1995 and have declined over historical levels. Even between 1994 and 1995, levels showed an apparent decline at many locations on the mainstem Fraser River, perhaps due to improved effluent quality in a discharge in the Prince George area. Some hot spots remain in areas of the lower Fraser, particularly in the vicinity of Agassiz and in the North Arm.

Recent work in residue guideline development and in derivation of toxic equivalency factors for individual congeners will probably, in the future, provide a better indication of the health risks to both wildlife and humans of low-level exposure to PCBs.

## Organochlorine Pesticides

## Background

This suite of contaminants includes a wide variety of harmful and potentially harmful persistent organochlorine compounds and their degradation products. All are soluble in lipid to some extent and display varying tendencies to accumulate and bioconcentrate in both terrestrial and aquatic ecosystems (Suedel et al. 1994). More than $90 \%$ of the pesticides considered here have for many years been banned for general use in North America, some since the early 1970s. However, many of the parent compounds or primary metabolites are quite stable, and low level contamination continues through volatilization and atmospheric transport from distant source areas where their application continues (Barrie et al. 1992, Wania and Mackay 1996) or from local residual contamination (Szeto and Price 1991). Evidence of global contamination by these chemicals has long been known (Ballschmitter et al. 1981). Initial concerns about the carcinogenicity of many of these chemicals has been compounded by their possible role as environmental disrupters of normal endocrine processes, particularly sexual differentiation and development (Arnold et al. 1996, Soto et al. 1994).

## Pesticide Results

A total of 24 organochlorine pesticides were measured in fish tissues sampled in 1994 and 1995 (Table 48). These analytical results and their associated sample detection limits are summarized in Table 49 to Table 52.

All 24 target pesticides were detected in one or more samples (Figure 37). Some were of very low frequency, as for example aldrin and heptachlor which were found in only a single sample in two years sampling. In addition, methoxychlor was detected in fewer than $8.5 \%$ of analyses in 1994 and $3 \%$ of all analyses in 1995. Others were more commonly encountered. Dieldrin, $p, p^{\prime}-\mathrm{DDE}$, hexachlorobenzene, trans-nonachlor were nearly ubiquitous in the basin, being detected in 82 to $100 \%$ of tissue analyses in both 1994 and 1995. Detection patterns between years in both muscle and liver, overall, showed strong similarity (Figure 38).

Concentrations of most of the target pesticides showed a strong similarity between sampling years (Figure 39). Levels were low to trace, with the majority of compounds measured in both muscle and liver from the three species being less than $5 \mathrm{ng} / \mathrm{g}$ w.w., and much of it less than $1 \mathrm{ng} / \mathrm{g}$ w.w. (Figure 39). There were some exceptions, which shall be considered in more detail below.

## Drins - Aldrin, Dieldrin, Endrin

Aldrin, endrin and dieldrin are closely related cyclodiene pesticides (Figure 40), widely used as soil fumigants and household/agricultural insecticides. Historically, the three had fairly high use in North America and dieldrin, in particular, was among the most widely used domestic pesticides (CCME 1988).

All three are closely related chemically. Dieldrin is an epoxide derivative of aldrin. In the environment aldrin is transformed to dieldrin with measured conversions consistently greater than $90 \%$ (CCME 1988). Endrin is a stereoisomer of dieldrin (Barrie et al. 1992). Dieldrin is particularly persistent, and has the highest carcinogenic potency of any of the major organochlorine pesticides (Barrie et al. 1992).

Aldrin was detected in only one tissue analysis in the two years of sampling (detection limits 0.2-0.4 $\mathrm{ng} / \mathrm{g}$ w.w.). Endrin was detected sporadically throughout the basin in both peamouth chub and whitefish at levels less than $1 \mathrm{ng} / \mathrm{g}$ w.w. (Table 49 to Table 52).

Dieldrin, in contrast, was detected in $80 \%$ of samples in 1995 and $95 \%$ of samples in 1994, at levels to $2 \mathrm{ng} / \mathrm{g}$ w.w. in peamouth chub liver (Table 51). Spatial differences in levels in peamouth liver are related to patterns in tissue lipid content through the basin. Likewise, between-species differences in tissue lipid content accounts for disparate dieldrin burdens amongst the three fish (Figure 41). A basin-wide pattern is weak. Liver dieldrin concentrations in whitefish throughout the basin and starry flounder in the two estuary reaches were less than $0.5 \mathrm{ng} / \mathrm{g}$ w.w. Levels in peamouth chub showed a general increase from $0.5-1.0 \mathrm{ng} / \mathrm{g}$ w.w. in the upper basin to $1-2 \mathrm{ng} / \mathrm{g}$ w.w. through the estuary
reaches. Concentration in muscle tissue throughout the basin averaged less that $0.2 \mathrm{ng} / \mathrm{g}$ w.w. (Figure 41), with minor differences among species and between years.

## Endosulphan

Endosulphan is one of only two organochlorine pesticides measured in this study which are currently registered for application. Sales of the active ingredient totaled over 270 kg in 1991, mostly in the Thompson basin and lower Fraser Valley (Norecol 1993). In the technical formulation are two endosulphan isomers, endosulphan I (also known as alpha-endosulphan) and endosulphan II (also known as beta-endosulphan) in a 70/30 mixture (Burgoyne and Hites 1993). In the environment, endosulphan-I tends to predominate, although both endosulphan I and endosulphan II isomers transform to endosulphan sulphate (Figure 42) (CCME 1988). Environmental residence time varies, but endosulphan is known to persist in soils for at least 2 years (NRCC 1975). Some finds its way into local watercourses. For example, Wan et al. (1995), in 1991, found levels of endosulphan sulphate in ditch waters draining agricultural areas in the lower Fraser Valley of up to $13.4 \mathrm{ug} / \mathrm{L}$.

Endosulphan congeners were detected throughout the basin in both 1994 and 1995 in 30 to $80 \%$ of the samples. Concentrations of the most commonly detected congeners, endosulphan sulphate and alpha-endosulphan were low, with mean levels in muscle in all species rarely exceeding $0.2 \mathrm{ng} / \mathrm{g}$ w.w. (Figure 43 and Figure 44). A slight increase in tissue concentration in the estuary sites relative to the upstream sites is evident in Figure 43, as are sporadic spikes in mean concentrations. The obvious peaks in both endosulphan sulphate and alpha-endosulphan in peamouth chub muscle from the North Arm Fraser River reach are due to measured levels of 2.6 and $2.9 \mathrm{ng} / \mathrm{g}$ w.w., respectively, in a single tissue composite. These elevated levels correspond very closely to sites in which Wan et al. (1995) measured consistently the highest concentrations of endosulphan (to $13.4 \mathrm{ug} / \mathrm{L}$ ) in ditch water of 7 localities sampled in the lower mainland.

Endosulphan isomers were generally at higher concentration in liver than muscle tissues, and consistently high (alpha endosulphan to $9.6 \mathrm{ng} / \mathrm{g}$ w.w.) in peamouth chub liver from the lower Fraser River reaches. There appears to be a clear relationship between elevated alpha- endosulphan levels in muscle and liver tissues from the lower Fraser, suggesting exposure of the fish to some source of the insecticide. Declining levels in the lower Fraser reaches in 1995 compared to 1994 suggests that the exposure is not chronic. In the Thompson basin, levels were low ( $<0.5 \mathrm{ng} / \mathrm{g}$ w.w.) and in both 1994 and 1995 were highest in the North Thompson reach.

Endosulphan sulphate detection in both fish species collected in the lower estuary sites doubtless reflects the continued use of the pesticide.

## DDT

Commercial DDT is composed of $70 \% ~ p, p$ '- and $30 \% ~ o, p$ - isomers (WHO 1979). These compounds are highly stable to photolysis and hydrolysis, and the $p, p^{\prime}$ - DDT is transformed in the environment to corresponding DDE and DDD isomers through biological activity (Barrie et al.
1992). In fish, the time-course of this transformation is rapid, probably on the order of days after exposure (Suedel et al. 1994). DDE, the principal environmental metabolite, is highly lipophilic and is stable in the environment, and as a result is the congener most commonly detected in environmental samples. DDT application was banned in North America in the 1970s, though use continued in India and parts of Europe as recently as 1992 (Barrie et al. 1992). Worldwide contamination continues through volatilization and long-range atmospheric transport (Wania and Mackay 1996).

All of the four DDT congeners measured (Figure 45) were detected in fish tissues in the basin. Both $p, p^{\prime}-$ DDD and $p, p^{\prime}$-DDE were detected in 80 to $100 \%$ of all analyses in both 1994 and 1995 (Figure 37). The parent $p, p^{\prime}$-DDT isomer was detected in about $35 \%$ of samples at low levels, at concentrations generally less than $0.3 \mathrm{ng} / \mathrm{g}$ w.w. (Figure 46). The highest $p, p$ '-DDT levels in muscle were from whitefish in the Agassiz reach on the Fraser and from the two reaches in the Thompson basin, where levels in both 1994 and 1995 ranged from 1.1 to $2.0 \mathrm{ng} / \mathrm{g}$ w.w. (Table 49, Table 50). DDT concentrations have been recorded in other studies in these reaches; in soils of the lower Fraser (Szeto and Price 1991) and in bed sediments in the Thompson (Brewer et al. 1998). In the Thompson River, $p, p^{\prime}$-DDT in particular constituted about $99 \%$ of the total DDT in sediments, the source of which is unknown (Brewer et al. 1998). DDT detections in liver were sporadic through the upper Fraser and Thompson in 1994 and 1995 but were strong and elevated through the lower Fraser in both peamouth chub and flounder in both sampling years (Figure 46).

The pattern of DDD and DDE concentrations mirror those of the parent DDT (Figure 47, Figure 48). Trace levels of DDD, generally less than $0.25 \mathrm{ng} / \mathrm{g}$ w.w., were measured in muscle of all three fish species. Prominent spikes in concentration occur in whitefish in those sampling reaches where DDT levels were similarly elevated - Agassiz in the lower Fraser and in the Thompson basin. DDD levels in peamouth liver probably reflect the influence of tissue lipid levels, with DDD being low to nondetectable through the upper Fraser and consistently $4-6 \mathrm{ng} / \mathrm{g}$ w.w. throughout the estuary.

Levels of $p, p^{\prime}$-DDE were the highest of all organochlorine pesticides measured in the study, up to 75 ng/g w.w. in peamouth chub liver from the lower Fraser River. Average concentrations of DDE in muscle tissue from both peamouth chub and whitefish in the upper basin were consistently less than 3 $\mathrm{ng} / \mathrm{g}$ w.w., with a clear increasing trend in concentration from the headwaters to the estuary (Figure 48). In peamouth chub, the pattern in DDE is probably closely related to changes in lipid content in reaches throughout the basin. As was observed in both DDD and DDT, the highest average DDE levels in muscle of both species ( 5.3 to $17.4 \mathrm{ng} / \mathrm{g}$ w.w.) in both sampling years were from the Agassiz reach (Figure 48). Concentrations in liver were again consistently lowest in the upper basin, increasing to levels of 6.5 (starry flounder) to 75 (peamouth chub) ng/g w.w. in the estuary reaches.

The presence of parent DDT isomers in some sampling reaches suggests local contamination sources. As a general guideline, ratios of DDE/ $\Sigma$ DDT in fish tissues of less than 0.70 are indicative of recent DDT contamination, with higher values of this ratio being "consistent with chronic uptake from very weathered sources" (Sanchez et al. 1993). The mean ratios for fish tissues from the Fraser collected during both 1994 and 1995 typically ranged from 0.59 to1.0. Levels of this ratio in the Thompson River tended to be somewhat lower (1995 mean $0.64,1994$ mean 0.68 ) than in the rest of the basin.

Average values near 0.45 in starry flounder muscle from the Fraser River Main and North arms in 1995 also suggest recent local contamination. Typical ranges of the DDE/LDDT ratio measured in this work are well within other published values for fish tissues in Canada (Metcalfe-Smith et al. 1995, Muir et al. 1990, Niimi and Oliver 1989).

Maximal total DDT levels measured in fish tissue in the basin were far below current Health and Welfare Canada guidelines for consumption by humans ( $5000 \mathrm{ng} / \mathrm{g}$ w.w.) and other relevant guidelines such as the International Joint Commission DDE guideline for protection of aquatic life in the Great Lakes (1000 ng/g w.w., Brazner and De Vita 1998). However, DDE concentrations in fish muscle in some reaches and in peamouth chub liver in many sites in the basin do exceed the $14.0 \mathrm{ng} / \mathrm{g}$ w.w. CCME DDE tissue residue guideline (CCME 1999) for protection of wildlife (Figure 48), and as such may be of concern. Wildlife studies conducted under the Fraser River Action Plan (Wilson et al. 1999) did find evidence of egg-thinning in bald eagle populations from the lower Fraser Valley in 1990-1991, and concentration of DDE in eggs above the threshold the $4.3 \mathrm{ng} / \mathrm{g}$ w.w. threshold level at which toxicological effects would be expected.

## Hexachlorocyclohexanes (including Lindane)

Hexachlorocyclohexane ( HCH ; Figure 49) has, in the past, been marketed in both a technical formulation containing $55-80 \% \alpha-, 5-14 \% \beta-, 8-15 \% \delta$ - and $2-16 \% \gamma$ - isomers, and as a purely insecticidal formulation containing $99 \% \gamma-\mathrm{HCH}$ (Hoff et al. 1992). The technical formulation was banned in North America in the 1970s, but $\gamma$-HCH (lindane) remains one of the highest use insecticides in Canada (Hoff et al. 1992). Lindane is used widely in both agriculture and forestry and is nearly ubiquitous in environmental samples. Other congeners may be of more concern in an environmental context. One component of technical $\mathrm{HCH}, \alpha-\mathrm{HCH}$, is the most frequently encountered isomer detected in environmental samples. The $\beta-\mathrm{HCH}$ isomer, in particular, is resistant to degradation, highly lipophilic and bioaccumulative and is a common contaminant of mammalian fatty tissues (WHO 1991).

Lindane in fish muscle through the upper basin averaged less than $0.5 \mathrm{ng} / \mathrm{g}$ w.w.(Figure 50, Table 49 and Table 50). Detections tended to be sporadic in both 1994 and 1995 analyses, with detection frequencies in muscle averaging about $30 \%$. Levels in the upper Lower Fraser Valley and in the Thompson reach were highest, with average values from 1.0 to $1.4 \mathrm{ng} / \mathrm{g}$ w.w. in 1994. In the Thompson reach, in particular, levels in 1994 and 1995 were almost identical, suggesting low-level chronic contamination in this area. Sporadic peaks are apparent, particularly in whitefish from Agassiz in 1995, where an average level in excess of $2 \mathrm{ng} / \mathrm{g}$ w.w. was measured.

Alpha-HCH was detected in 57 and $76 \%$ of tissue analyses in 1994 and 1995, respectively. Levels in fish muscle were similar throughout the basin, with mean levels typically less $0.3 \mathrm{ng} / \mathrm{g}$ w.w. (Figure 51). Concentrations in liver in the upper basin show a uniform decline in whitefish and peamouth chub tissues from the upstream reaches, which seems to maintained between sampling years (Figure 51). Detection frequencies were highest in liver tissues (Figure 37), and similar between whitefish and peamouth chub (Figure 38). Tissue $\alpha-\mathrm{HCH}$ concentrations were highest in peamouth chub liver both
in the lower Fraser (to $2.6 \mathrm{ng} / \mathrm{g}$ w.w.) and in the Thompson (to $2.05 \mathrm{ng} / \mathrm{g}$ w.w.) reaches (Table 51). In fish muscle tissues, concentrations were generally less that $1 \mathrm{ng} / \mathrm{g}$ w.w.

In muscle tissues of both peamouth chub and whitefish in several reaches in the basin, other HCH isomers, particularly $\beta-\mathrm{HCH}$, tended to predominate (Figure 52). This was particularly evident in samples of peamouth chub from the lower Fraser and in analyses of both species from the Thompson River. This isomer is not a major product in the transformation series of lindane, and the only common source is in the technical HCH formulation. The measured contamination could perhaps be resulting from residual contamination or from atmospheric transport and deposition.

## Hexachlorobenzene

Hexachlorobenzene (HCB, Figure 53) has had wide application in both agriculture as a fungicide, and in a number of industrial processes (CCME 1988). In addition to its primary uses, HCB may also result from incineration, and is a byproduct of pesticide and chlorinated solvent manufacturing (CEPA 1993). In British Columbia, HCB was released to the environment through seed treatments, in HCBcontaminated herbicides, from chlor-alkali plants and from wood treatments (Wilson and Wan 1982). In the environment, the primary transformation route is though dechlorination and hydrolysis to pentachlorophenol, with subsequent modification to lower chlorinated chlorophenols.

HCB was detected in $95 \%$ of all tissue analyses. Levels in muscle throughout the basin were typically less than $0.4 \mathrm{ng} / \mathrm{g}$ w.w.(Figure 54), with only slight differences between species or between years. The highest average concentrations ( $0.65 \mathrm{ng} / \mathrm{g}$ w.w., $1994 ; 0.75 \mathrm{ng} / \mathrm{g}$ w.w., 1995) were found whitefish muscle in the North Thompson reach. Measured concentrations in the North Thompson were higher than in the Thompson reach in both 1994 and 1995.

HCB in liver was lowest in the upper Fraser reaches (max 0.7 to $1.5 \mathrm{ng} / \mathrm{g}$ w.w.), and increased into the estuary area ( $\max 3.2 \mathrm{ng} / \mathrm{g}$ w.w.) (Figure 54). The highest levels were in peamouth chub liver, probably a reflection of the higher lipid levels, relative to either whitefish or starry flounder.
Concentrations between years were very similar.

Based on a small amount of available data, levels have declined over time. Garrett (1980, cited in (Wilson and Wan 1982)) reported concentrations of 3.9 to $17.0 \mathrm{ng} / \mathrm{g}$ w.w. in peamouth chub from the Fraser River downstream of Hope. The present (1994/5) maximum levels averaged less than $0.40 \mathrm{ng} / \mathrm{g}$ w.w. in peamouth muscle downstream of Hope.

## Chlordane and Heptachlor

Commercial chlordane is a mixture of over 140 chlorinated compounds (Dearth and Hites 1991). The primary components (by weight) are 19\% cis-chlordane, 24\% trans-chlordane, 10\% heptachlor, $7 \%$ cis- and trans-nonachlor and $22 \%$ miscellaneous chlordane isomers (Figure 55).

Chlordane registration in Canada was restricted to termite control in 1985, and use was suspended in 1990, with all stocks to be depleted by 1995. Heptachlor was itself registered for control of insect pests from the mid-1950s to the late 1970s (Fendick et al. 1990). In the environment, both cis- and trans-chlordane are readily transformed to oxychlordane, while heptachlor is transformed principally to its corresponding epoxide (Fendick et al. 1990). In both cases, the transformation product is more persistent and is more toxic than the parent compound.

Both trans-nonachlor and cis-chlordane were measurable in 70-80\% of samples (Figure 37). Transnonachlor levels were low through the basin, with average within-reach concentrations in muscle generally less than $0.4 \mathrm{ng} / \mathrm{g}$ w.w. (Figure 56). Likewise, heptachlor epoxide was detected in greater than $90 \%$ of samples at trace levels throughout the basin, with average within-reach concentrations less than $0.1 \mathrm{ng} / \mathrm{g}$ w.w. (Table 49 to Table 52). In contrast, oxychlordane and trans-chlordane were detected infrequently.

Some between-species differences in detections are apparent. Several chlordane compounds, such as cis-chlordane and trans-nonachlor which were detected at either a low frequency or absent from whitefish were common in peamouth (Figure 56, Figure 57). As has been the pattern with many of the organochlorine contaminants, this may be due largely to differences in tissue-lipid content between the pairs of species.

## Toxaphene

Toxaphene (Figure 58) is a complex mixture of many hundreds of chlorobornanes and derivatives (Saleh 1991). Although never formally registered in Canada, toxaphene was used in insect control and saw a short application as a piscicide for killing coarse fish in lakes prior to trout stocking. In the southern United States, toxaphene was used extensively for protection of cotton crops. While toxaphene was banned in the US in 1986 and was de-registered in Canada in 1983, use of the pesticide continues for some applications in Russia and Mexico (Voldner and Li 1993). Of particular concern is the persistence of the compounds (Miskimmin et al. 1995) and the relatively high levels recorded at locations remote from sources of contamination (Donald et al. 1993, Voldner and Li 1993). Bioconcentration of toxaphene through food chain transfer is well-known (Kidd et al. 1995)

Toxaphene was detected in about $80 \%$ of all tissue samples, and in 92-100\% of peamouth chub liver and whitefish muscle samples (Figure 37) collected in 1994 and 1995. In both peamouth chub and whitefish, toxaphene tended to be the dominant organochlorine pesticide, occasionally exceeding even the relatively high levels of $p, p$-DDE. Tissue burden of toxaphene was clearly related to tissue lipid content, and patterns throughout the basin, particularly in peamouth chub liver, are probably attributable to this factor.

The highest levels in both peamouth chub and whitefish muscle were from Agassiz and the Thompson basin, with only slight differences between the two sampling years (Figure 59). The implication is that these reaches either receive a chronic source of toxaphene contamination, whether it be local or arising from atmospheric transport, or perhaps sustain a particular food web which encourages bioconcentration. Toxaphene levels in whitefish muscle tended to be higher than peamouth chub,
reflecting the somewhat fattier ( $2-5 \%$ lipid) muscle of whitefish compared to peamouth chub (1-2\%). Concentrations in peamouth chub liver tissue were, with the exception of elevated levels in particular estuary sites, quite uniform throughout the basin (Figure 59). Particularly high levels (>30 ng/g w.w.) were measured in peamouth chub liver from the Mission and Barnston reaches in the lower Fraser and in the Thompson River reaches in 1994 and Agassiz in 1995.

All measured values were well below the Health Canada human alert level for toxaphene of $100 \mathrm{ng} / \mathrm{g}$ w.w. (Haines et al. 1994), and far below the maximum allowable limit of $5000 \mathrm{ng} / \mathrm{g}$ w.w. for consumption by humans in the US and the IJC guideline for wildlife in the Great Lakes of $1000 \mathrm{ng} / \mathrm{g}$ w.w. (Brazner and De Vita 1998). However, concentrations in muscle within several reaches, and in peamouth chub liver throughout the basin were in excess of the CCME tissue residue guideline for protection of piscivorous wildlife of $6.3 \mathrm{ng} / \mathrm{g}$ w.w. (CCME 1999).

## Pesticide Summary

Nearly all of the target pesticides analyzed in this work are presently banned or have very limited application in North America. The levels found here probably represent the effects of both atmospheric transport and deposition, with subsequent accumulation in the food chain. Those pesticides dominating the total organochlorine pesticide burden, toxaphene and DDE, are also found in highest concentrations in remote locations elsewhere in Canada (Barrie et al. 1992, Donald et al. 1993, Muir et al. 1990). As such, the concentrations in tissues measured in 1994 and 1995 probably represent expected "background" levels which will persist until a world-wide ban of these chemicals is implemented. This conclusion is further supported by the consistency between years in tissue concentrations.

Concentrations in the three species of fish in both liver and muscle of all measured pesticides were very low relative to measured historical levels. Present levels in fish tissues in the estuary are orders of magnitude less than measured by Albright et al. (1975) in the early 1970s when many of these organochlorines pesticides were in current use.

Hazards arising from the current levels of the measured organochlorine pesticides are probably low, based on comparisons with existing consumption guidelines for human health. Levels may be of some concern to piscivorous wildlife, based on newly derived residue limits for DDE and toxaphene which are exceeded in a number of reaches in the basin.

## PAHs and PAH Metabolites

## Background

The polycyclic aromatic hydrocarbons (PAHs) include a wide range of natural and anthropogenic organic compounds which possess at least two cyclic carbon rings. Only a few PAHs are manufactured intentionally, the vast majority being released from petroleum seeps and burning. PAH contamination is characteristic of densely-populated municipal areas, where releases from vehicular
emissions, incinerators, sewage treatment plants, and road runoff introduce relatively high levels of a wide spectrum of PAHs. Eisler (1987) estimated that, of the total anthropogenic PAH contamination to the aquatic environment of about 230 kilotonnes, about $74 \%$ was due to petroleum spillage and $22 \%$ due to atmospheric deposition.

Characteristic congener suites and profiles result from each particular production mechanism, and in some cases it has been possible to attribute specific sources when full spectrum analyses of PAHs have been conducted (Yunker and McDonald 1995). In an environmental context, the most significant PAHs are those with 3-7 benzene rings (Nagpal 1993). To simplify interpretation of environmental data, PAHs are often divided into two groups -- high molecular weight (HPAHs) having 4 to 6 benzene rings and low molecular weight (LPAHs) having 2 to3 rings. These groupings provide a convenient contrast in solubility, bioaccumulation potential and phase partitioning behavior of the array of PAHs in the environment.

Biological effects of PAHs vary considerably. LPAHs tend to have relatively high acute toxicity, while several of the HPAHs, for example, benzo(a)pyrene ( $\mathrm{B}[a] \mathrm{P}$ ), are proven and potent carcinogens (Santodonato 1997, Varanasi 1989). Concentrations in tissues exposed to PAH contamination are a complex interaction of both the physicochemical properties of the compound and biological activities within the organism (Meador et al. 1995). The HPAHs tend to be strongly hydrophobic ( $\mathrm{K}_{\mathrm{ow}}$ 5-7.4) and might therefore be expected to accumulate in aquatic organisms. However, most PAHs are readily metabolized and excreted by fish, either in urine or in bile. In some cases these metabolic transformation products have higher carcinogenic activity than does the parent PAH (Meador et al. 1995).

Determining exposure to PAHs can be important in evaluating and interpreting the environmental health of populations. This can, however, be difficult because the relatively short half-life of these compounds in tissues creates transient elevated levels and the variety of metabolic transformations that can occur can make for challenging analytical determinations. As a screen for exposure, a broad analysis of PAH classes in bile has been used effectively in a number of studies (Escartín and Porte 1999, Krahn et al. 1992, Krahn et al. 1984, Stehr et al. 1997). This analysis determines concentrations in bile of major PAH classes based on fluorescence of aromatic rings, independent of lateral groups. This analysis provides some indication of exposure to naphthalene-like (2-rings), phenanthrene-like (3-rings) or $\mathrm{B}[a] \mathrm{P}$-like (5-ring) compounds (Figure 60).

Anthropogenic sources of PAHs in the Fraser basin are vast and varied. Many of the industrial and municipal effluents, such as pulp and paper mills and wastewater treatment plants, release a range of PAHs (Derksen 1997 (Draft)). Runoff from road surfaces, which carry accumulated hydrocarbons and deposited vehicle emissions, are also important sources of PAHs (CEPA 1994).

## PAH in Tissue Results

In 1994, muscle and liver from starry flounder and peamouth chub in the estuary areas were analyzed for a total of 23 individual PAH congeners -- six LPAHs and 17 HPAHs (Table 53). Results are summarized in Table 54 to Table 56.

In addition to parent PAH analyses, bile from peamouth chub, starry flounder, and whitefish was analyzed for PAH metabolite classes in 1994. As noted in the discussion of data quality, the replication of bile metabolite measurements was poor. Within batches of analyses, duplication tended to be good but between batches, and particularly with blind re-submissions, replication was quite poor. Several factors, including high levels of particulate matter, very high viscosity in the sample bile and problems with an autoinjector were likely contributors to the poor results (Axys Analytical Ltd., pers. comm.). Respecting these concerns, the results are included here both for completeness and because these are the first such data from fish in the Fraser basin. The relatively suspect nature of the data must be borne in mind when considering the discussion to follow.

## Parent PAHs in Tissues

Nineteen of the 23 PAHs were detected in at least one tissue sample. Four compounds were not detected -- the three isomers of dibenzopyrene and 3-methylcholanthrene. A total of 11 PAHs were detected in liver and 19 in muscle (Table 54). Of these, five were found only in peamouth muscle from the North Arm reach. Summary concentrations of the measured PAHs in muscle and liver in the two species are presented in Table 55 and Table 56, respectively.

The suite of detected PAHs in both liver and muscle was very similar, with the six LPAHs and two HPAHs (fluoranthene and pyrene) being detected in greater than $70 \%$ of analyses. The exception was 7,12-methylcholanthrene, which was not found in liver tissues, but was detected at relatively high levels (to $4.5 \mathrm{ng} / \mathrm{g}$ w.w.; Table 55) in nearly all muscle samples.

With regard to possible sources of these compounds, Derksen (1997 (Draft)) measured effluent PAH levels from wastewater treatment plants in the lower Fraser, and found phenanthrene (43-85 $\mathrm{g} / \mathrm{day}$ ), fluoranthene ( $23-63 \mathrm{~g} / \mathrm{d}$ ) and pyrene (18-74 $\mathrm{g} / \mathrm{day}$ ) in the highest concentrations of the measured compounds. Naphthalene, and several of the other LPAHs are characteristic of petroleum contamination, and their detection is consistent with releases from stormwater runoff and shipping traffic (Yunker and McDonald 1995). Others, such as acenaphthylene and acenaphthene are more typical of a combustion source (Yunker and McDonald 1995), and were detected in wastewater effluents (Derksen 1997 (Draft)).

Total low and high molecular weight PAHs in peamouth chub muscle and liver both show a clear seaward increase in concentration from the upstream Lower Fraser Valley sites at Mission and Barnston (Figure 61). The highest levels were found in the North Arm, an industrialized area with a high density of stormwater outfalls (Nener and Wernick 1998) and effluent discharges (Swain et al. 1995 (draft)). Total concentrations of individual LPAHs and HPAHs in peamouth chub liver were, in
general, roughly 10x greater than in muscle (Table 55, Table 56), probably reflecting the difference in lipid content between the two tissues and the lipophilic nature of PAHs in general.

In starry flounder liver and muscle tissues, concentrations of LPAHs and HPAHs were similar between the North and Main Arms (Figure 62). Here again, concentrations in liver were roughly 10x greater than in muscle tissues.

## PAH Metabolites in Bile

Results of PAH metabolite measurements in bile are presented in Figure 63. Because of the relatively suspect nature of the data (see previous discussion), results are included only in figures and have not been tabulated.

In the headwater areas of the Fraser and in the Thompson River reaches, levels of PAH metabolites in bile in peamouth chub tended to be relatively low, probably representing either low-level contamination or natural background levels from combustion sources (Figure 63). Downstream to Marguerite, concentrations of each of the major metabolite classes increase, consistent with increases in industrial activity and effluent volumes. The highest levels in peamouth chub were from the Main and North Arms in the estuary and from Marguerite (Figure 63).

Phenanthrene and naphthalene equivalents in peamouth chub in the estuary reaches were roughly similar to levels measured downstream of Quesnel (Figure 63), with the notable absence of $\mathrm{B}[a] \mathrm{P}$ equivalents in the estuary. The seaward increase in the tissue concentrations is mirrored in the bile, with levels of both phenanthrene and naphthalene equivalents $3-8$ fold higher in the Main and North Arms compared to the upstream Barnston and Mission sites. A trace of $\mathrm{B}[a] \mathrm{P}$ was measured in samples only from the North Arm. This is consistent with characteristics of the contamination sources through most of the lower Fraser area; Combined Sewer Outfalls (CSOs) discharging LPAHs in runoff (Derksen 1997 (Draft)) and releases of light fuel oils rich in naphthalenes and phenanthrenes (Nagpal 1993). The North Arm has, as has been noted, a high CSO density which probably contributes to the presence of HPAHs in local fish.

Analyses of whitefish bile were particularly complicated by crystallization and high viscosity, and, as such, the pattern in bile metabolites is less clear (Figure 63). Naphthalene and phenanthrene equivalents showed little relationship to potential sources, with levels either remaining constant or declining downstream. In the Thompson basin, for example, concentrations of all three classes were lowest downstream of Kamloops.

In the Fraser Estuary reaches, general patterns in PAH contamination are similar among the three different media, when parent PAH concentrations are presented as "equivalents" (Figure 64). The possible exception is in results for $\mathrm{B}[a] \mathrm{P}$ equivalents, which shows an increase far in excess of that seen in tissues. Further study will be required to evaluate the significance of this observation. However, it is consistent that $\mathrm{B}[a] \mathrm{P}$ was detected only in tissue (one peamouth muscle sample) and
bile from the North Arm. Even with the relatively poor data quality, the utility of metabolite measurement in reflecting general PAH contamination is evident in these data.

Effluent characterization of pulp and paper mill discharges in 1994 and 1995 (Derksen 1997 (Draft)) showed some of these effluents to be minor sources of a range of HPAHs, particularly fluoranthene, pyrene and chrysene. Relatively high loadings of HPAHs in the estuary was associated with discharges from sewage treatment plants. Metabolites of these tetra and pentacyclic compounds could fluoresce in the same range as $\mathrm{B}[a] \mathrm{P}$ and may be included in determinations of $\mathrm{B}[a] \mathrm{P}$ equivalents in bile.

## Historical Levels and Comparison to Guidelines

PAHs in peamouth chub and starry flounder tissues from the lower Fraser River were measured during a survey conducted in 1988 (Swain and Walton 1989). For most of the target compounds, levels in 1994 were much lower than were measured in 1988 (Table 57). Exceptions were acenapthene and phenanthrene, for which the maximum levels measured here were as high, or several times higher than in 1988. Phenanthrene is commonly detected at relatively high levels in storm-water runoff and municipal treatment plant effluents (Derksen 1997 (Draft)).

A recent study by Stehr et al. (1997) provides some data from starry flounder in the San Francisco Bay area with which to compare the present results. They measured average naphthalene equivalents of $15,000 \mathrm{ng} / \mathrm{mL}$ in the "reference" area, and $90,000 \mathrm{ng} / \mathrm{mL}$ in affected areas. Similarly, B[a]P levels in the reference area averaged $40 \mathrm{ng} / \mathrm{mL}$, while in affected areas levels were approximately 250 $\mathrm{ng} / \mathrm{mL}$. For comparison, average naphthalene equivalents in peamouth bile in estuary reaches ranged from 63,000 to $80,000 \mathrm{ng} / \mathrm{mL}$ and $\mathrm{B}[a]$ P equivalents from $<1$ to $37 \mathrm{ng} / \mathrm{mL}$ (Figure 64). For starry flounder collected in the Fraser estuary, average naphthalene equivalents ranged from 5,000 to $26,000 \mathrm{ng} / \mathrm{mL}$, and $\mathrm{B}[a] \mathrm{P}$ equivalents were all below detection ( $<1 \mathrm{ng} / \mathrm{mL}$ ).

Some comparative data are available from analyses of bile from longnose sucker sampled in the Athabasca River in 1992 during the Northern Rivers Basin Study. B[a]P equivalent concentrations ranged from 54 to $480 \mathrm{ng} / \mathrm{mL}$ and phenanthrene levels ranged from 5300 to $15,000 \mathrm{ng} / \mathrm{mL}$. While the $\mathrm{B}[a] \mathrm{P}$ levels measured in this study were much lower, phenanthrene concentrations in peamouth chub from the estuary areas were almost 10 times greater than in the Athabasca samples, as might be expected in this more industrialized setting.

## Comparison to Guidelines

The only established guideline for PAH in tissue is for $\mathrm{B}[a] \mathrm{P}$ for protection of human health by BC Ministry of Environment. Levels relate to consumption rates, with $4 \mathrm{ng} / \mathrm{g}$ w.w. being a maximum residue for those consuming $50 \mathrm{~g} /$ week to $1 \mathrm{ng} / \mathrm{g}$ for consumers of 200 g fish tissue per week (BCMELP 1998). B[a]P was detected in only one peamouth muscle sample - estimated at $2.0 \mathrm{ng} / \mathrm{g}$ w.w., therefore levels measured in fish tissues from the lower Fraser River are not likely a hazard to human health (detection limits 0.1 to $0.7 \mathrm{ng} / \mathrm{g}$ w.w., except one sample at $2.0 \mathrm{ng} / \mathrm{g}$ w.w.).

## Trace Metals

## Background

Trace metals are of particular environmental concern because of both toxicity and potential for bioaccumulation and biomagnification in biological tissues (Suedel et al. 1994). Many metals are of concern due to their high acute toxicity, which may be either mitigated or attenuated by ambient pH and/or water hardness (Enserink et al. 1991, Foulkes 1989). Others, particularly the heavy metals such as lead, mercury, cadmium, arsenic and selenium may accumulate in tissues. In many instances, trace metals in the ambient environment may affect fish health through reductions in immune response (Anderson and Zeeman 1995).

Accumulation of metals in tissues is of concern for the health of fish and both human and wildlife consumers of fish tissue. The mechanisms for metal accumulation vary, but a particularly effective route for metals such as mercury and selenium (Maier and Knight 1994) is through binding of the metal by microbial activity to an organic ligand, increasing both the lipid-solubility and the bioavailability (Suedel et al. 1994). Tissue concentrations of trace metals are affected by a number of factors, such as the metabolic requirements of the organism for the element, the toxic threshold of the metal and whether the tissue has a role in sequestering detoxified metals, such as by binding to protective proteins such as metallothioneins (McCarthy et al. 1997).

Erosion of natural ore deposits can be a significant source of metals to the environment, particularly where particular environmental conditions encourage dissolution, with subsequent biological uptake either through food items or direct absorption of dissolved metal in ambient water. Sediment-borne metals often remain in particulate form and therefore relatively benign in the environment.

Potential anthropogenic sources of metals to the environment are vast and varied, ranging from industrial effluents to mine leachates, to high dissolved copper and lead from delivery pipe dissolution in municipal wastewater, run-off from urban areas, and atmospheric deposition from combustion. A particular environmental concern in these instances is often not the absolute concentrations, but the high proportion of the total metal content in the dissolved phase which is highly available to organisms.

## Trace Metal Results

Twelve trace metals were measured in whitefish, peamouth, and flounder from 11 reaches in 1994 and from 9 reaches in 1995. A list of the metals and their associated detection limits are presented in Table 58. Results of these analyses are present in Table 59 to Table 62.

Concentrations of many metals in both liver and muscle tissues were uniform throughout the basin, and showed only slight differences in mean within-reach concentrations between years.

Several metals were detected only rarely at or near their detection limits in either muscle or liver. Lead was not detected in muscle tissue in either 1994 or 1995, and measured only occasionally at low level in liver (Table 61- Table 62). The highest lead concentration measured ( $0.32 \mu \mathrm{~g} / \mathrm{g}$ w.w.) was in starry flounder liver from the North Arm in 1994 (Table 62). Basin-wide, the highest measured levels in peamouth chub liver were in the Nechako reach (Table 60).

Metals at very low levels or not detected in muscle but present at elevated levels in liver were chromium, cobalt and cadmium. In both peamouth and whitefish, the concentrations of these three metals show no clear pattern through the basin. Cadmium in liver is of particular interest because of the elevated concentration in starry founder compared to the other two species (Figure 65). In terms of risk to fish, Eisler (1985) suggested that tissue concentrations greater than $2 \mu \mathrm{~g} / \mathrm{g}$ w.w. were indicative of cadmium contamination, and levels greater than $5 \mu \mathrm{~g} / \mathrm{g}$ w.w. probably having a detrimental effect on the organism. Liver cadmium levels in starry flounder in 1994 and 1995 were near or greater than the $2 \mu \mathrm{~g} / \mathrm{g}$ w.w. level.

In view of the geographic extent of the sampling program and the number of species involved, factors related to water quality, salinity effects and habitat preferences will be of importance in determining tissue metal burden. This is particularly evident in patterns of total arsenic in whitefish and peamouth chub muscle. Arsenic has an array of possible uses and sources in the Fraser basin, including arsenical pesticides, wastewater treatment plant effluents and natural mineral sources (Norecol 1993). Arsenic levels in both peamouth chub and whitefish tended to be low (average $<1 \mu \mathrm{~g} / \mathrm{g}$ ) throughout the basin (Figure 66), being elevated only in the estuary reaches. Tissue arsenic concentrations in starry flounder were typically 3-5 times greater than peamouth chub collected in the same reach. Arsenic has high solubility in seawater compared to freshwater, and accumulation in starry flounder probably reflects its marine/estuarine habitat. Levels measured in starry flounder from estuary sites, while elevated relative to other reaches in the basin, were lower than levels measured in mature starry flounder from marine waters in Boundary Bay and Roberts Bank ( $0.41-1.01 \mu \mathrm{~g} / \mathrm{g}$, Swain and Walton 1994). Arsenic levels in peamouth chub muscle in 1995 were considerably higher than in 1994 (Figure 66), indicating sampling of a different habitat or perhaps a refection of the relatively higher 1994 flows diluting marine waters in the estuary. Unfortunately, enough liver tissue was not available for arsenic analysis at all sites but where the data are available, levels in both muscle and liver were quite similar (Figure 66).

Mercury has long been of concern in human and environmental health because of its potential for methylation and subsequent rapid uptake and transfer through the food chain. Methylmercury typically comprises greater than $99 \%$ of total mercury in fish (Bloom 1992). Tissue mercury levels in fish from the Fraser basin were less than $0.3 \mu \mathrm{~g} / \mathrm{g}$ w.w.in all reaches, and generally less than $0.1 \mu \mathrm{~g} / \mathrm{g}$ w.w. (Table 59 to Table 62, Figure 67). Levels in peamouth chub muscle were consistently higher than in either whitefish or flounder and showed only minor differences between reaches.

Mercury in both whitefish and peamouth chub muscle and liver in the upper Fraser reaches tended to be highest near the natural mercury deposits of the Pinchi Fault in the Nechako watershed (Plouffe 1995), and concentrations in tissue declined downstream to the estuary (Figure 67). Present levels in
the Nechako area are far below those measured in the late 1970s, when levels as high as $1.5 \mu \mathrm{~g} / \mathrm{g}$ w.w. were measured in whitefish muscle (Garrett et al. 1980). Elevated mercury concentrations found in peamouth chub liver in the estuary are probably related to both industrial releases and accumulation of methylmercury due to high tissue lipid content. In terms of risk to human health, maximum measured concentrations were well below the $0.5 \mu \mathrm{~g} / \mathrm{g}$ w.w. maximum consumption guideline level set by Health and Welfare Canada (Haines et al. 1994).

Zinc in tissues in the basin showed little pattern from upstream to downstream (Figure 68). Zinc has wide industrial and residential application, but an expected downstream increase in tissue levels is not apparent in either whitefish or peamouth chub. Between-species differences in muscle burden are apparent, with concentrations in peamouth chub being generally higher than those of whitefish from the same reaches. In liver analyses, whitefish and peamouth chub zinc levels are very similar, while concentrations in starry flounder were about 2 x greater than those of peamouth (Figure 68).

Selenium, like arsenic and mercury, has potential for biomethylation and subsequent accumulation in tissues (Maier and Knight 1994). Environmental levels may result from natural weathering, but selenium may also be released from agricultural runoff, ash runoff, and metal mines and can reach high levels in municipal wastewater. Selenium concentrations in fish muscle were uniform and consistently low ( $<0.6 \mu \mathrm{~g} / \mathrm{g}$ w.w.) throughout the Fraser basin, with minor differences between species or reaches (Table 59 to Table 62, Figure 69). Selenium would appear to be accumulated preferentially in liver tissues, since levels in liver in both whitefish and peamouth chub were about 5-10x greater than those in muscle. Lemley and Smith (1987), cited in (Maier and Knight 1994) estimated "safe" levels of selenium in fish at approximately $3.5 \mu \mathrm{~g} / \mathrm{g}$ w.w. in muscle and $7.5 \mu \mathrm{~g} / \mathrm{g}$ w.w. in gonads and liver tissues. Levels in fish tissues from the Fraser Basin in both 1994 and 1995 were well below these respective "safe" levels. Some levels of selenium in mountain whitefish liver were slightly higher than the $3 \mu \mathrm{~g} / \mathrm{g}$ w.w. BC tissue residue guideline for the protection of human health, but are unlikely to be of concern to consumers.

Year to year concentrations of most trace metals were quite similar. Exceptions were in levels of manganese, copper, iron and nickel in muscle of both peamouth chub and whitefish, although in some cases this was inexplicably restricted to upper basin reaches (Figure 70, Figure 72). Nickel concentrations in muscle in 1995 averaged about 1/10th or less of those measured in 1994 (Table 59, Table 61). Accumulation of nickel in tissue is thought due principally to absorption from the water column (Suedel et al. 1994), the difference might, perhaps, be due to changes in ambient concentration although such variation in dissolved levels in the Fraser River are unlikely. Interestingly, in contrast to the results found for whitefish and peamouth chub, nickel levels in starry flounder were similar between years.

## Comparisons with Historical Levels

Wide-scale geographic assessments of trace metals in fish tissues in the Fraser basin have been conducted on at least two occasions, a longitudinal sampling in 1980 of fish on the Fraser River (Singleton 1983) and a large-scale assessment of levels in "uncontaminated" lakes (Rieberger 1992).

Other sampling has focussed on habitats and species of the lower Fraser area and estuary (Swain and Walton 1989).

Singleton (1983) measured a suite of 12 metals in a wide variety of both resident and anadromous species in the Fraser River. Industrial areas were, surprisingly, not always the areas associated with the highest levels of many of the metals. Of the measured metals, chromium, copper, molybdenum, and nickel were elevated in the upstream reaches, and lead, magnesium and mercury were highest in the estuary area.

Reiberger's (1992) results from analyses of mountain whitefish in lakes collected from 1982-1987 are either well within or slightly higher than the ranges of the Fraser Basin data from either 1994 or 1995 (Table 64). A notable exception is that of nickel in both muscle and liver. Within reach averages for whitefish muscle in 1994 were about 0.20 to $0.48 \mu \mathrm{~g} / \mathrm{g}$, and for liver less than $0.06 \mu \mathrm{~g} / \mathrm{g}$. Average levels in lake samples were 1.21 and 1.24 for muscle and liver, respectively. Factors causing this discrepancy warrant some further investigation.

Results from the present study are within or near historical ranges measured for species in the Fraser. Even in the estuary, where development has proceeded with great vigor over the decade since the survey of Swain and Walton (1989), tissue concentrations of most trace metals have remained surprisingly stable (Table 63).

## Mixed Function Oxygenase Activity

Mixed function oxygenase (MFO) induction was used as an indicator of organic contaminant exposure in the Fraser basin. Fish metabolize foreign chemicals by oxidation, reduction, hydrolysis, and conjugation reactions catalyzed by various enzymes, such as MFO, which are found mainly in the liver (Addison et al. 1994). Accordingly, hepatic MFO induction has been linked to exposure to contaminants such as dioxins and furans (Hodson et al. 1991), $\mathrm{B}[a] \mathrm{P}$ (Focardi et al. 1995), certain PCB congeners and pesticides (Jimenez and Burtis 1989), and retene (alkyl-substituted phenanthrene) (Fragoso et al. 1997, Parrott et al. 1994). In Canada, MFO induction in fish exposed to BKME has been well documented (Hodson 1996, Munkittrick et al. 1991, Nener et al. 1995).

Current literature seems to support two types of toxic compounds that induce MFO. Type I inducers that accumulate in tissues (i.e., dioxins, furans, PCBs, and chlorinated pesticides), and Type II inducers that are rapidly metabolized by MFO enzymes and excreted (i.e., PAHs and retene) (Hodson 1996).

Maturity, age, sex, species, season, and temperature may influence the MFO system and its response (Andersson and Forlin 1992, Forlin and Andersson 1984, Hansson et al. 1982, Jimenez and Burtis 1989, Parrott et al. 1996, Stegeman 1993). In addition, the current literature indicates that MFO induction is biphasic with strong induction at lower concentrations and an attenuated response at a higher concentrations. The attenuated response is attributed to inactivation of catalytic function at
higher concentration of an inducer (Hahn et al. 1996). Consequently, MFO induction results should be interpreted with caution.

Summary statistics for EROD and ECOD activity in mountain whitefish, peamouth chub, and starry flounder are shown in Table 66 and Figure 73.

## Quality Assurance/Quality Control

Initial EROD activity in the positive controls was very low in 1996, indicating that resorufin standards were not producing an adequate curve. However, after some samples were run, the positive controls returned to typical levels. MFO blind duplicate results are presented in Table 67. Variability between blind duplicates was high, as indicated by the yearly mean relative standard deviations (RSD) of $27 \%$ and $52 \%$, respectively, for 1995 and 1996. Individual RSDs ranged from $0.1 \%$ to $109 \%$ in 1995, and $0.5 \%$ to $141 \%$ in 1996. Other studies also indicated high variability in EROD data (Vandermeulen and Mossman 1994).

MFO activity levels in 1994 were much higher than in 1995 and 1996, in both peamouth chub and mountain whitefish. Unfortunately, the assay and laboratory were changed between 1994 and 1995, therefore the year-to-year variation in MFO activity may be attributed to the change in methods. To estimate the difference between the two methods, archived whole liver samples collected in 1994 at the Mission reach were analyzed in 1996 by the second method. The results indicate that the 1994 method produced results almost ten times higher than the 1995/6 method (Table 68). Therefore, the change in method appears to be the primary factor causing year-to-year variability in EROD activity, and the activity cannot be directly compared among years.

Another factor potentially contributing to higher EROD activity in 1994 was the extreme low flow condition of the Fraser River during the 1994 sampling (Figure 74). As a result, fish could be exposed to higher levels of contaminants resulting in higher EROD activity.

## EROD versus ECOD Activity

EROD and ECOD activities were correlated (mountain whitefish $\mathrm{r}^{2}=0.71$, peamouth chub $\mathrm{r}^{2}=0.59$, starry flounder $r^{2}=0.99$; Figure 75). Since both EROD and ECOD are induced by $\vartheta$-naphthoflavone, and EROD and ECOD activities in winter flounder have both been correlated with PAH concentrations in sediments (Addison et al. 1994), either of these assays can be used to measure MFO induction.

## Geographic Variability

EROD activity was significantly elevated in both mountain whitefish and peamouth chub downstream of urban centres and pulp mills (Figure 73, Appendix 1) on both the Fraser and Thompson rivers. Up to 78 -fold and 18 -fold increases in EROD activity were seen in mountain whitefish (Woodpecker) and peamouth chub (Main Arm), respectively, when compared to the Nechako reference reach in
1994. Specifically, mean EROD activities were 125 pmoles/mg protein/min for mountain whitefish from Woodpecker, 34.5 pmoles $/ \mathrm{mg}$ protein $/ \mathrm{min}$ for peamouth chub from the Main Arm, and 1.59 to 1.93 pmoles/mg protein/min at Nechako (Table 66).

EROD activity in peamouth chub was highest at Mission, Barnston, and Main Arm in 1994, and in the Main and North arms in 1995 (Figure 73). Mountain whitefish EROD activity was highest in 1994 at Woodpecker, Marguerite, and then Agassiz, and highest in 1995 at Marguerite and Agassiz. Elevated EROD activity at Woodpecker, Marguerite, and Agassiz is most likely due to exposure to pulp mill effluents. In contrast, EROD activity in the lower Fraser River may be due to many factors including discharges from sewage treatment plants, industrial activities, combined sewer outflows, and urban and agricultural runoff, as well as residual pulp mill contaminants from the upper reaches.

Within-reach variability was high in some cases. In 1995, two peamouth chub from the Hansard reach had EROD activities of 34.8 (8 year old female) and 135 (13 year old male) pmoles/mg protein $/ \mathrm{min}$ indicating possible movement from downstream reaches or recent exposure to MFO inducing compounds in the reference reach. Three peamouth chub from the North Arm in 1995 also had high EROD activity compared to the reach mean ( 9.52 pmoles $/ \mathrm{mg}$ protein $/ \mathrm{min}$ ) - 37.5, 23.0 and 22.8 pmoles $/ \mathrm{mg}$ protein $/ \mathrm{min}$, indicating possible hot spots with respect to MFO-inducing contaminants within the North Arm. These data points were treated as outliers and removed from analyses.

EROD was measured in starry flounder from only one reach in 1994 and two reaches in 1995. Therefore, only starry flounder collected from the Main and North arms in 1995 can be compared. EROD activity was significantly higher in the North Arm than Main Arm (Figure 73, Appendix 1). The North Arm has lower flows and more contaminant sources than the Main Arm, so this higher induction was expected.

## Sex and Maturity Effect

Male peamouth chub and mountain whitefish exhibited higher EROD activity than females, particularly in 1994 (Table 66, Figure 73). This finding is consistent with other studies (Flammarion and Garric 1997, Hansson et al. 1982, McMaster et al. 1991, Nener et al. 1995). EROD activity may be reduced or eliminated during the spawning period (McMaster et al. 1991, Vandermeulen and Mossman 1994) when levels of hormones, such as estradiol, vary dramatically in fish serum. Estradiol is known to suppress EROD activity (Stegeman 1993), while testosterone has no effect (Forlin and Andersson 1984). In addition, due to spawning migrations, the fish may not represent the areas in which they are caught (Hodson et al. 1991).

The gonadosomatic index (GSI) is an indicator of state of maturity in fish- the higher the GSI, the closer to maturity. In this study, there were few correlations between GSI and EROD activity in peamouth chub (Table 69). Peamouth chub are spring spawners, therefore the spawning period was well before the summer and fall sampling. As a result, maturity would not be expected to affect EROD induction. In contrast, there were strong negative relationships between the gonadosomatic
index (GSI) and EROD activity in female mountain whitefish collected from most reaches (Table 69). The graphical presentation of female mountain whitefish $\log$ EROD activity versus GSI indicated two distinct populations of mountain whitefish in 1994 (Figure 76). A population consisting of prespawning female mountain whitefish with high GSIs and zero EROD activities, and the other, consisting of immature mountain whitefish (GSI near zero) with a range of EROD activities.

## Seasonal Effect

Peamouth chub EROD activity was slightly higher in summer than in the fall in the Main Arm in 1996 (Figure 73), however the difference was not significant (Appendix 1, $\mathrm{p}=0.83$ ). Seasonal variability in EROD activity has been documented by many authors (McMaster et al. 1991, Munkittrick et al. 1991, Vandermeulen and Mossman 1994). Seasonal variations have been attributed to a variety of factors, including water temperature, sexual maturity, endocrine and metabolic functions, food consumption, or exposure to pollutants (Sloof et al. 1983).

## Species Effect

Large differences in MFO activity among species have been measured during field studies (Hahn et al. 1993, Kloepper-Sams and Swanson 1992, Munkittrick et al. 1992b). Such differences may be due to variations in feeding pattern, tissue lipid content, metabolic rates, contaminant exposure route (dietary exposure in addition to water borne exposure), contaminant accumulation, and sensitivity to inducers (Hodson et al. 1991).

Mountain whitefish and starry flounder seemed to be more sensitive to MFO-inducing contaminants than peamouth chub (Figure 73). In 1994, peamouth chub had higher activity ( 34.52 pmoles $/ \mathrm{mg}$ protein $/ \mathrm{min}$ ) than the starry flounder ( 9.88 pmoles $/ \mathrm{mg}$ protein $/ \mathrm{min}$ ). However, in 1994 the first supernatant fraction of starry flounder livers was analyzed, and this fraction is expected to yield lower activity than the microsome fraction. Activity in starry flounder was higher than peamouth chub in 1995, when the same method was used for both - 9.52 versus 13.77 pmoles $/ \mathrm{mg}$ protein $/ \mathrm{min}$ in the Main Arm and 8.63 versus 33.86 pmoles $/ \mathrm{mg}$ protein $/ \mathrm{min}$ in the North Arm, for peamouth and starry flounder, respectively.

## Fish Health and EROD Activity

Studies have shown that MFO induction in fish may or may not coincide with pathology and physiological changes (McMaster et al. 1991, Servos et al. 1994, Sloof et al. 1983, Swanson et al. 1992, Van Der Oost et al. 1991, Williams et al. 1996). And the significance of MFO induction in terms of growth, reproduction and survival is unknown (Hodson 1996). For example, 30-fold MFO induction was seen in mountain whitefish collected downstream of pulp mills from the Wapiti River but no health effects were detected (Swanson et al. 1992). However, low levels of induction in mountain whitefish from the Columbia River were associated with health effects (Nener et al. 1995).

Over all reaches in the Fraser basin, there were weak relationships between the HAI, condition index, and hepatosomatic index (HSI) and male EROD activity in mountain whitefish, peamouth chub, and starry flounder (Table 70). Since some relationships were positive and some negative, factors other than those inducing MFOs were likely responsible for variability in fish health. For example, fish captured at reference sites, where MFO levels were low, were significantly older and had higher parasite loads than those from exposed sites, resulting in higher HAIs. Consequently, there were negative correlations between HAI and EROD in 1994, for peamouth chub and mountain whitefish ( $\mathrm{r}=-0.32, \mathrm{p}=0.00 ; \mathrm{r}=-0.45, \mathrm{p}=0.00$ ).

## Contaminants and EROD Activity

Since hepatic MFO induction has been linked to exposure to contaminants and a significant induction of EROD activity was seen at reaches downstream of contaminant sources, the relationships between EROD activity and contaminant levels were examined. MFO-inducing compounds measured in this study and a synchronous FRAP sediment study (Brewer et al. 1998) were dioxins, furans, PCBs, pesticides, PAHs, and retene. For compounds that accumulate in fish tissue, i.e., dioxins, furans, PCBs, and pesticides, the relationships between fish tissue residues and EROD activity were explored. For compounds that do not accumulate in tissues, i.e., PAHs and retene, the relationships between sediment residues (Brewer et al. 1998) and EROD activity were explored. Results are presented in Table 71. Results should be interpreted with caution since contaminants tended to cooccur at sites downstream of the major urban and industrial sources, making it difficult to isolate the contaminant causing the induction.

Dioxins and furans TEQs in liver tissues of mountain whitefish ( $\mathrm{r}=0.89, \mathrm{p}=0.04, \mathrm{n}=5$ ) correlated positively with EROD activity, but peamouth chub TEQs did not $(\mathrm{r}=0.42, \mathrm{p}=0.30, \mathrm{n}=8)$ (Figure 77). PCB TEQs in liver tissues of peamouth chub correlated with EROD activity in both 1994 and $1995(\mathrm{r}=0.66$ and $0.68, \mathrm{p}=0.04, \mathrm{n}=10$ and 9$)$. PCB TEQs in mountain whitefish liver also correlated with EROD in $1995(\mathrm{r}=0.92$, $\mathrm{p}=0.00, \mathrm{n}=7)$ but not in 1994.

Total DDTs (DDE, DDD, and DDT) and toxaphene were selected to represent the chlorinated pesticides because they occurred at the highest concentrations throughout the basin. DDTs and toxaphene levels in peamouth liver tissue correlated with EROD activity ( $\mathrm{r}=0.53$ to $0.76, \mathrm{p}=0.02$ to $0.15, \mathrm{n}=9$ to 10 ). No correlations were seen for mountain whitefish tissue levels and EROD activity.

PAHs were measured in bed sediment in both 1994 and 1995, but samples were not collected from the estuary in 1994. Bed sediment PAHs were weakly correlated with EROD activity in peamouth chub only ( $\mathrm{r}=0.72$ and $0.70, \mathrm{p}=0.07$ and $0.11, \mathrm{n}=7$ and 9 ; in 1994 and 1995).

Retene is an alkyl-substituted phenanthrene commonly found in BKME. Retene is one of many naturally occurring wood extractives which is released when the wood is processed. It induces EROD activity in fish and is thought to be formed by the bacterial reduction of dehydroabeitic acid (DHAA) under anaerobic conditions (Fragoso et al. 1997, Parrott et al. 1994). Retene was measured in bed sediment samples collected in 1995, and correlated with peamouth chub EROD
activity $(\mathrm{r}=0.70, \mathrm{p}=0.03, \mathrm{n}=9)$. But no relationship was observed for mountain whitefish ( $\mathrm{r}=0.30$, $\mathrm{p}=0.52, \mathrm{n}=7$ ).

In summary, analyses suggest that EROD induction in peamouth chub may be caused by several inducers - PCBs, DDT, toxaphene, PAHs, and retene, while induction in mountain whitefish may be attributed almost exclusively to dioxins and furans. However, mountain whitefish were not sampled from the estuary where levels of PAHs, retene, and PCBs were highest. The contribution of other inducers to peamouth chub EROD activity is suggested at Woodpecker, Marguerite, and the estuary in the plot of EROD activity versus dioxin and furan TEQs (Figure 77).

## Summary

EROD activity in mountain whitefish and peamouth chub indicated exposure to organic contaminants downstream of pulp mills and urban centres. Dioxins and furans appear to be responsible for EROD induction in mountain whitefish, but a variety of contaminants appear to be causing the induction peamouth chub, particularly in the estuary. Sex and maturity strongly influenced EROD activity in mountain whitefish; activity was inhibited in mature, pre-spawning females. No clear relationship was found between EROD activity and fish health. Based on these results, induction of EROD activity is a good screening tool for organic contaminant exposure but not an indicator of biological effects.

## Age, Length, and Weight

Variability in size and age of fish is important because of their relationships with somatic indices, reproduction, and health. During sampling we attempted to standardize fish sizes in the samples, but differences in rates of growth and availability of targeted sizes resulted in variability in size and age among reaches and years. Size and age distributions are also important because they integrate the interaction of rates of reproduction, growth, and mortality of age groups present in fish populations. These distributions help to identify problems such as year-class failures, low recruitment, slow growth, or excessive annual mortality (Anderson and Gutreuter 1983).

Age assessment from calcified structures, such as otoliths, is strongly subjective and tends to overestimate age (Weatherley and Gill 1987). However, age-validation studies indicate that there is usually good agreement between age assessed from calcified structures and actual age for young and fast-growing fish, usually up to 6 to 10 years of age, depending on growth rate. Since age-validation studies have not been done on the species sampled in this study, reported ages, particularly for older fish, should be considered only an estimate.

Ages, lengths, and weights are summarized by reach, year, and species in Table 72 to Table 74. Box and whisker plots, by reach, year, and species, of ages, lengths, and weights are illustrated in Figure 78 to Figure 85. Age- and length-frequency plots by reach, year, and species are presented in Figure 86 to Figure 101.

Two-way ANOVAs were used to test for differences among reaches and between years for ages (log), lengths (log), and weights (log), for each species (Appendix 2 to Appendix 4). The GT2method (Hochberg 1974, cited in Sokal and Rohlf 1995) was used to compare means (Appendix 2 to Appendix 4). Data from 1996 were not included in the ANOVA because the large number of empty cells would result in a less meaningful analysis. The year-by-reach interaction effects were significant for all variables ( $\mathrm{p}<0.001$ ), except starry flounder ages, i.e., the effects of reaches on the variables were not consistent between years or vice versa (Appendix 2 to Appendix 4). Therefore, the multiple comparison tests compared samples which were defined by years and reaches.

## Peamouth Chub

Peamouth chub in samples decreased in age going downstream, i.e. from north to south (Figure 78 to Figure 80, Appendix 2). Mean ages ranged from about 10 years at Hansard down to a range of 2 to 5 years in the Main Arm (Table 72). Peamouth chub from northern reaches (mean lengths up to 239 mm ) were larger than those from southern reaches ( 197 mm at Marguerite in 1995 to 214 mm at the North Arm in 1995) (Table 72), even though the same size range was targeted at all reaches. Mean age and size did not vary significantly between the two Thompson basin reaches ( 6.3 to 8.9 years; 219.1 to 231.5 mm ) (Table 72 and Appendix 2).

Peamouth chub age and length distributions were generally skewed to the left (Figure 86 to Figure 91). Distributions were wider in northern reaches - Nechako, Hansard, and both Thompson reaches, than in the middle and lower Fraser reaches. In the middle and lower Fraser, peamouth chub over 5 years old and 230 mm were rare, whereas in the upper Fraser (Hansard), the Nechako, and the Thompson basin large, old chub were common, particularly in 1994. The lack of older, larger fish indicates mortality or out-migration of larger fish in the middle and lower Fraser that is not seen in the upper Fraser and Thompson basin.

Weak and strong age classes were apparent in the age distributions. At Woodpecker and Marguerite in the central mainstem, peamouth chub of age 5 years (and under 200 mm ) dominated the samples in 1994. Interestingly, this 1989 year-class dominated samples from the middle Fraser reaches in all years sampled, including Woodpecker in 1996 (7-year-olds). Other large and small year classes could be followed through the sampling years - large populations in year-classes 1982, 1983, and 1986 at Hansard and small populations in year-class 1984 at Hansard and year-class 1988 in the Nechako. This suggests that peamouth chub reproduction or survival varies from year to year.

## Mountain Whitefish

As with peamouth chub, mean ages of mountain whitefish sampled from the Fraser River decreased going downstream ( 5.4 and 8.9 years at Hansard versus 3.2 and 4.6 years at Agassiz, in 1994 and 1995) (Figure 81 to Figure 83; Appendix 3). In the Thompson basin, mountain whitefish were younger in the Thompson ( 3.3 and 3.2) than in the North Thompson ( 4.6 and 6.0) in both years. In 1994, Thompson whitefish (mean length 248 mm ) were also smaller than those from the North Thompson ( 282 mm ). No downstream trends were seen in size - in 1994 the largest fish were from
the North Thompson (mean length 282 mm ) and smallest from Hansard ( 252 mm ), two northern reaches, and in 1995 the largest fish were from Agassiz ( 287 mm ) and smallest from Hansard (257 mm ).

Mountain whitefish age distributions were skewed to the left; length distributions were less skewed (Figure 92 to Figure 97). Age distributions were wider in northern reaches - Nechako and Hansard, than in the middle and lower Fraser reaches and in the Thompson basin. At Woodpecker and Marguerite in the central mainstem, 1990 year-class mountain whitefish were rare or missing in both 1994 and 1995. There were also missing or relatively rare age-classes at Hansard and Woodpecker reaches. This suggests that mountain whitefish reproduction or survival also varies from year to year.

## Starry Flounder

Starry flounder were sampled only from two reaches in 1994 and 1995, and only immature fish were selected. Starry flounder were slightly older and larger in the North Arm in 1994 and younger and slightly younger and smaller in 1995, than in the Main Arm (Figure 84, Figure 85). In 1994, most starry flounder sampled were 2 years old (Figure 98). In 1995, wider distributions of sizes and ages were sampled, particularly in the Main Arm, but 2-year-olds were still the largest age class (Figure 100 and Figure 101).

## Size-At-Age

The variability of peamouth chub and mountain whitefish size-at-age, as an indicator of growth rate, was tested by ANCOVAs of log length using log age as the covariate. The GT2-method was employed for multiple comparisons of age-adjusted mean lengths. Only differences among samples (defined by reach and year) were tested so that all the data, including 1996, could be used in the analysis without having empty cells.

The selection of specific size categories during sampling introduced error in the calculation of growth rates. This sampling design selects larger fish in the younger age-classes and smaller fish in the older age-classes with the result that calculated growth rates are always lower than actual. An attempt was made to eliminate size-selection bias by selecting age-classes as follows:

1. Eliminating young age-classes based on the age frequency plots for each reach and year. The increasing left side of the distributions represent the young age-classes that were incompletely sampled (Ricker 1975).
2. Eliminating old age-classes based on length versus age scatter plots for each year and reach. The distributions that exceed the upper limit of the selected size classes represent old ageclasses that were incompletely sampled. Note that peamouth chub were rarely size censored on the upper end of the selected size range.
Lists of the age-classes included in the analyses, by species, year, and reach, are presented in Appendix 5 and Appendix 6. An ANOVA was used to compare size-at-age for starry flounder because the age-class 2 fish are least biased by the size-selected sampling. Even these are probably biased because the distributions extend beyond the selected class of 15 to 20 cm .

To attain homogeneity of slopes for the peamouth chub ANCOVA, data collected in 1995 for the Nechako reach were eliminated. The slope of the regression calculated for this data was steep compared to other reaches and years. A second multiple comparison test was run including the 1995 Nechako data just to get an estimate of size-at-age for this sample.

Results of the ANCOVA and multiple comparison for peamouth chub are presented in Appendix 7, Appendix 8, and Appendix 9. Differences in age-adjusted mean lengths among samples were significant ( $\mathrm{p}=0.02$ ). Adjusted lengths generally showed an increasing trend moving downstream in the basin (Figure 102). However the multiple comparison test detected two main groupings:

1. significantly larger size-at age in the Nechako ( 224.0 mm ), Thompson basin
( 215.7 to 219.9 mm ), and lower Fraser River ( 214.9 to 223.5 mm )
2. than in the Hansard (197.1 to 201.0 mm ), Woodpecker (means of 200.5 to 204.2 mm ), and Marguerite ( 190.8 to 193.6 mm ) reaches.

Results of the ANCOVA and multiple comparison for mountain whitefish are presented in Appendix 10, Appendix 11, and Appendix 12. Differences in age-adjusted mean lengths among samples were significant ( $\mathrm{p}=0.0001$, Appendix 10). The multiple comparison test for whitefish adjusted mean lengths did not detect distinct groups, but many overlapping groups increasing in size downstream. The smallest fish were found at Hansard (adjusted mean lengths of 223.9 to 248.2 mm ) and the largest at Agassiz (adjusted mean lengths of 285.4 to 290.3 mm ). The increase in growth rates from the north to south (going downstream) indicates that latitude and altitude were the primary factors controlling growth. Results for the Thompson basin were interesting -- North Thompson mountain whitefish were larger than Thompson fish in 1994 and vice versa in 1995 (Figure 103). These results suggest that different whitefish populations were sampled within these reaches in 1994 and 1995, possibly due to sampling earlier in 1995 and sampling more northerly sites on the North Thompson in 1995.

The Hansard reach covered a long stretch of river compared to other reaches. Sites were spread over a distance greater than 200 km , from the mouth of the Dore River (near McBride) downstream to Averil Creek just upstream of Prince George (Figure 1). The distribution of sampling over these sites varied from year to year due to variability in fishing success. Because mountain whitefish collected from the Hansard reach in 1995 were much smaller for their age than those collected in 1994 and 1996 (Figure 103), a size-at-age ANCOVA was conducted on just the Hansard data set to assess the effect of sampling site (Appendix 13). The ANCOVA indicates that sites are responsible for the inter-annual variability ( $p=0.01$ ), and the adjusted mean lengths are larger at downstream sites - 241.9 mm to 245.7 mm at McBride to 285.0 mm at Hansard, adjusted for a mean age of 6.92 Appendix 13). A similar analysis for peamouth chub indicated that size-at-age did not vary among sites $(\mathrm{p}=0.52)$ (Appendix 14).

To verify results of the size-at-age ANCOVA, mean lengths were plotted versus ages for peamouth chub and mountain whitefish (Figure 104). These plots show the same trend in growth rates as the ANCOVA, confirming the above conclusions. Included in the plot for mountain whitefish are data
from other studies on the McGregor and lower Columbia rivers. Zimmerman (1994) compared mountain whitefish size-at-age data collected from rivers throughout BC and found that growth rates were lowest in the McGregor River (a tributary to the Fraser in the Hansard reach) and highest in the Columbia River. The range of growth rates in the Fraser basin was almost as broad as that previously reported for BC.

Results of the ANOVA and multiple comparison for starry flounder log lengths are presented in Appendix 15 and Appendix 16. Starry flounder mean lengths of two-year-old fish were significantly larger in the North Arm (183.4 mm) than in the Main Arm (176.6 mm) $(\mathrm{p}=0.001)$.

## Fat Reserves, Condition Index, and Hepatosomatic Index

Fat reserves and condition indices, in addition to growth rates (size-at-age), are generally used as indicators of the well-being of fish. Higher fat reserves, condition indices, and growth rates infer better condition. Low fat reserves and condition indices may be caused by stress such as contaminant exposure but they also fluctuate seasonally with feeding activity, migrations, and sexual maturation (Goede and Barton 1990).

Fat reserves in fish are critical for over-wintering survival and for synthesis of reproductive tissue (Goede and Barton 1990). Fat storage enables fish to store energy over the summer for winter metabolism and gonad maturation (MacKinnon 1972). Generally, fat reserves are low during spawning, maximum at the end of summer foraging, and decline during wintering and migrations. Shul'man (1974) proposed that fish exhibit six types of lipid dynamics; two types apply to peamouth chub and mountain whitefish:

1. In spring-spawning and non-migrating temperate fish, such as peamouth chub, lipid levels are minimum in June from over-wintering and gonad maturation and maximum in December from intensive post-spawning feeding.
2. In winter-spawning and moderately-migratory fish exposed to marked fluctuations in environmental conditions, such as mountain whitefish, lipid levels are maximum in June and minimum in December.
However, the variability of fat dynamics and storage depots can be extreme in different fish species and races due to condition of habitat, time and nature of spawning, length of migration, duration of wintering, food supply during gonad ripening and foraging, and temperature, salinity, and gas regime of the water (Shul'man 1974).

In this study, fat reserves were assessed using lipid levels in muscle and liver and a mesenteric fat index. Muscle and liver lipid levels and mesenteric fat indices for peamouth chub, mountain whitefish, and starry flounder are summarized by reach, year, and species in Table 75 and Figure 105 to Figure 107.

Condition and hepatosomatic indices (HSI) were used to describe relationships between somatic variables. Fulton's condition index ( K ) was used to indicate the fatness of the fish: $\mathrm{K}=\mathrm{c}\left(\mathrm{W} / \mathrm{L}^{3}\right)$, where c is a scaling constant, W is the gutted weight, and L is the fork length. "Normal" or "average"
condition indices vary depending on the morphology of the species (Anderson and Gutreuter 1983). The HSI is the percentage of body weight attributable to the liver and can be an indicator of fat reserves and nutritional state when the liver is an energy storage depot. Condition indices and HSI are summarized by reach, year, and species in Table 72 to Table 74. Box and whisker plots, by reach, year, and species, of condition indices and HSI are illustrated in Figure 108 to Figure 110.

## Storage Depot Variability

Mesenteric fat, liver, and muscle are the major fat storage depots in fish. In cyprinids, such as peamouth chub, the liver is the major fat storage organ. However, salmonids, such as mountain whitefish, store fat primarily in muscle and the mesentery (Roberts 1978, Sheridan 1988). Consequently, in the Fraser basin, peamouth chub liver lipid levels were an order of magnitude higher than muscle lipids (muscle:liver lipid ratios of 0.06 at Mission in 1994 to 0.45 at Marguerite in 1994) (Figure 105); but in mountain whitefish, lipid levels in muscle and liver were similar (muscle:liver lipid ratios of 0.45 at Agassiz to 1.85 in the North Thompson River in 1994) (Figure 106). Starry flounder collected from the estuary also had more than an order of magnitude higher lipid levels in the liver than muscle (muscle:liver lipid ratios of 0.04 at the Main Arm in 1994 to 0.07 for all other samples) (Figure 107).

## Geographic and Annual Variability

Two-way ANOVAs were used to test for differences in lipid levels, mesenteric fat, condition, and HSI among reaches and between years (Appendix 17 to Appendix 19). At Agassiz, large differences were seen between years in lipid reserves, condition, and HSI (Figure 105, Figure 106, Figure 108, and Figure 109). Removal of Agassiz data eliminated year by reach interactions for ANOVAs on peamouth chub liver lipids and mountain whitefish muscle lipids (Appendix 17, Appendix 18) and allowed among-reach comparisons.

Because size and age varied among samples, analyses of variability in indices and lipid among reaches should include the effects of these factors as covariates. However, the effects of the size and age, although usually significant, were not consistent among reaches and between years. Therefore, these factors could not be used to adjust somatic indices in a basin-wide ANCOVA, and only the ANOVA results are presented here.

The liver is the major lipid storage site for peamouth chub, consequently liver lipid levels were highly variable in this species. Peamouth chub mean liver lipid levels ranged from $3.75 \%$ to $28.0 \%$ (Table 75 , Figure 105), and differences among reaches ( $p<0.001$ ) and between years ( $p=0.04$ ) were significant (Appendix 17). Mean liver lipid levels were highest at Agassiz (1995 only; 24.33\%), estuary ( 22.3 to $25.2 \%$ ), and Thompson (20.7\%) reaches; intermediate at Nechako (14.3\%), Hansard (13.8\%), and Woodpecker (11.6\%) reaches; and lowest at North Thompson (7.6\%) and Marguerite (5.3\%) reaches (Appendix 17). Liver lipid levels were higher in 1995 (least squares mean 16.0\%) than 1994 (14.1\%) (Appendix 17).

Peamouth chub muscle lipid levels varied little among reaches compared to liver -- averaging between $1.36 \%$ and $2.16 \%$, except in the Thompson River (3.03\%) in 1994, and at Agassiz ( $2.48 \%$ ) and Marguerite ( $0.49 \%$ ) in 1995 (Table 75, Figure 105). The year by reach interaction was significant when testing for differences among reaches and between years ( $\mathrm{p}<0.001$ ) (Appendix 17), i.e. there was not the same geographic pattern in both years.

Peamouth chub mean mesenteric fat indices followed the same geographic pattern as liver lipid levels - highest in the estuary and the Thompson River (3.4 to 3.7) and lowest in the North Thompson (1.7 to 3.0) and at Marguerite (1.1 to 1.9) (Figure 105). The sample means overall ranged from 1.5 to 3.9. Inter-annual differences were also similar to liver lipids -- fat indices were lower in 1994 than in 1995, particularly at Woodpecker (2.6 to 3.6), Agassiz (1.5 to 3.9), and North Thompson (1.7 to 3.0), and the ANOVA year by reach interaction was significant ( $\mathrm{p}<0.001$ ) indicating different reach effects between years (Appendix 17). Furthermore, indices were lower at Hansard in 1994 and 1995 compared to 1996 ( 2.4 and 2.7 to 3.7).

Peamouth chub mean condition indices ranged from 0.89 to 1.05 , and were highest in the Nechako (1.05 and 1.01, in 1994 and 1995) and Thompson (1.02 and 1.05) rivers and lowest at Marguerite ( 0.91 and 0.95 ) (Table 72, Figure 108). Mean condition indices decreased going downstream from Hansard ( 0.95 and 0.98 ) to Woodpecker ( 0.92 and 0.97) and Marguerite then increased in the lower Fraser ( 0.89 to 1.01 ). Condition was higher in the Nechako River (significant in 1994) than at Hansard (Appendix 17). In the Thompson basin, condition indices were significantly higher in the Thompson than in the North Thompson ( 0.96 and 0.98 ) in both years. Large differences were seen between years in the lower Fraser at Agassiz -- 0.89 in 1994 versus 1.01 in 1995, and in the estuary -0.98 to 1.00 in 1994 and 0.93 to 0.94 in 1995.

Peamouth chub HSI ranged from 1.43 to $4.94 \%$ and varied little among reaches, despite variability in lipid levels. In 1994, HSI were significantly higher in the North Arm (1.90\%) than at Nechako, Thompson, Agassiz, and Mission (1.43 to 1.55\%) (Table 72, Figure 108, Appendix 17). In 1995, peamouth chub HSI were significantly higher at the Agassiz and Thompson reaches (4.94 and $2.68 \%$ ) than all other reaches ( 1.76 to $2.25 \%$ ). Mean HSI at these two reaches were much higher in 1995 (4.94 and 2.68\%) than 1994 (1.53 and 1.47\%).

Mountain whitefish mean liver lipid levels ranged from 2.24 to $4.35 \%$ (Table 75, Figure 106). Differences among reaches ( $\mathrm{p}=0.04$ ) were significant, but only between Agassiz (mean of 3.78\%) and the North Thompson ( $2.79 \%$ ) $(\mathrm{p}=0.04)$ (Appendix 18). Mean liver lipid levels were higher in 1995 (3.43\%) than in 1994 ( $2.96 \%$ ) ( $\mathrm{p}=0.01$ ) (Appendix 18).

Mountain whitefish were expected to store lipid in their muscle, rather than liver, however mountain whitefish muscle lipids were neither as variable nor as high as peamouth liver lipids. Mean muscle lipid levels ranged from 1.43 to $5.88 \%$ (Table 75, Figure 106), and differences among reaches ( p < 0.001 ), but not between years ( $p=0.75$ ), were significant. The geographic pattern was similar to lipid reserves of peamouth chub except that muscle lipid levels were highest in the North Thompson (4.16\%) -- significantly higher than all Fraser and Nechako reaches (Appendix 18). Intermediate
muscle lipid levels were found in the Thompson (3.24\%), Nechako (2.71\%), Woodpecker (2.65\%), and Hansard ( $2.47 \%$ ) reaches. The Marguerite mean lipid level ( $2.02 \%$ ) was significantly lower than both the Thompson and North Thompson reaches (Appendix 18). Over all reaches, both lowest and highest muscle lipid levels were seen at Agassiz -- 1.43\% in 1994 and 5.88\% in 1995.

Mountain whitefish mean mesenteric fat indices ranged from 0.8 to 3.2 , and the year by reach interaction was significant ( $\mathrm{p}<0.001$ ) (Figure 106, Appendix 18). The geographic pattern was similar to muscle lipids, particularly in 1994. In 1994, indices were higher in the Thompson (3.2) and North Thompson (2.1) reaches than in the Fraser mainstem (1.0 to 1.7) and Nechako (1.5). In 1995, indices were higher than in 1994 at Nechako (2.2), Hansard (2.3), and Agassiz (2.7). Marguerite (1.3) and Woodpecker (1.2) indices were the lowest, averaged over both years, and Agassiz (1.0) was lowest in 1994.

Mountain whitefish mean condition indices ranged from 0.88 to 1.05 , and were highest at Woodpecker ( 0.96 to 1.01 ) and in the Thompson basin ( 0.96 to 1.05 ) (Table 73, Figure 109, Appendix 18). Thompson (1.05 and 0.99) mountain whitefish had higher condition indices than North Thompson (0.96). Condition indices were higher in 1995 than in 1994 at all Fraser and Nechako reaches. In contrast, condition indices in the Thompson basin were higher in 1994 than in 1995. As with peamouth chub, condition at Agassiz was much higher in 1995 (1.03) than in 1994 (0.88).

A standard weight equation for mountain whitefish 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 $100 \%$. Mean relative weights for the Fraser basin ranged from $78.6 \%$ to $91.7 \%$, except at Agassiz in 1995 (101.2\%) (Figure 109). 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"), also may result in low condition indices and bias estimates of fish health (McPhail and Troffe 1998).

In 1994, mountain whitefish HSI were significantly higher at Agassiz (1.11\%) than in the Thompson $(0.73 \%)(p=0.01)$ (Table 73, Figure 109, Appendix 18). In 1995, HSI were significantly higher at Agassiz ( $2.11 \%$ ) than at all other reaches ( 0.68 to $1.06 \%$ ), and higher at the North Thompson (1.06\%) than at Woodpecker (0.68\%).

Starry flounder were sampled only from two reaches - the Main and North arms of the estuary. The lipid levels in muscle were not significantly different between the Main ( 0.20 to $0.37 \%$ ) and North ( 0.34 to $0.36 \%$ ) arms or between years ( $p=0.48, \mathrm{p}=0.31$, respectively) (Table 75, Figure 107, Appendix 19). Only one liver sample per reach was analyzed for lipid -- levels ranged from 4.90 to $5.55 \%$. No mesenteric fat was found in any starry flounder except one fish from the Main Arm in 1995 (rating of 1). Starry flounder condition indices and HSI were higher in 1995 (1.00) than 1994 (0.96) (p < 0.001) (Table 74, Figure 110, Appendix 19). Starry flounder HSI were higher in the

North Arm than the Main Arm in both years ( p < 0.001 ). This result is consistent with results for size-at-age - 2-year-old starry flounder were larger in the North Arm than the Main Arm. The larger marine influence in the North Arm than in the Main Arm possibly results in additional food resources in North Arm.

In summary, peamouth chub energy reserves and condition indices, as well as growth rates, were all highest in the Nechako, Thompson, and lower Fraser rivers, probably due to higher temperatures and lower sediment levels (Figure 3, Figure 4, Table 3) in these reaches resulting in higher productivity. Peamouth chub condition and growth rates were lowest at Marguerite in the central basin downstream of Prince George and Quesnel. While low energy reserves, condition, and growth rates at Marguerite may be due to contaminant inputs, these were more likely due to high suspended sediment levels and river gradient causing stress on fish and reducing productivity in the river. Mean suspended solids concentrations range from $18 \mathrm{mg} / \mathrm{L}$ (low flow) to $89 \mathrm{mg} / \mathrm{L}$ (high flow) at Hansard and $74 \mathrm{mg} / \mathrm{L}$ to $147 \mathrm{mg} / \mathrm{L}$ at Marguerite (Table 1). Tributaries contribute much of the suspended sediment load downstream of Prince George, but most of the increase is probably due to erosion along the mainstem (Carson 1988). Scrivener et al. (1994) suggested that sediment levels downstream of Quesnel in spring and summer were high enough to harm juvenile salmonids by disrupting feeding, growth, and social behaviour, and increasing susceptibility to disease, particularly during freshet. They believe that juvenile salmonids, including mountain whitefish, migrate into tributary streams during the spring and summer to clear their gills and reduce exposure to suspended sediment. In September, after sediment levels declined, salmonids disappeared from tributaries, presumably by migrating back into the mainstem. In the Thompson basin, lower temperatures in the North Thompson (Figure 3) may explain the lower fat reserves and condition in North Thompson versus Thompson peamouth chub.

In contrast, mountain whitefish energy reserves and condition indices were highest in the Thompson basin, and fairly similar among Nechako and Fraser reaches. In 1995, condition indices were more variable, and high at Woodpecker and Agassiz compared to other Fraser reaches. Mountain whitefish trends may be different from peamouth chub because whitefish can migrate long distances (Swanson et al. 1993) and may not be good indicators of environmental factors at their capture sites. McPhail and Troffe (1998) studied the movement of mountain whitefish in the Fraser River near Prince George. This research indicates that individual mountain whitefish return to the same summer foraging sites every year, and that there is very little movement during the summer feeding period. However, fall aggregations in the mainstem and large tributaries, which were sampled in this study, may contain mountain whitefish from many summer feeding locations, including smaller tributaries.

## Seasonal Variability

This study did not intend to assess the effect of season on fat reserves and indices, and most reaches were sampled in the fall (Table 3). However, some seasonal variability was captured because reaches could not be sampled simultaneously. The most southerly estuarine reaches were sampled earliest in the summer, the most northern reaches in late summer and early fall, then central and southern reaches through the fall. In addition, the Main Arm reach was sampled in both summer and fall in
1996. To detect a seasonal effect, condition indices were plotted versus Julian day for all three years (Figure 111). For peamouth chub, condition increased through the summer for the early-sampled estuarine reaches. Then condition declined when sampling the northern mainstem reaches of Hansard, Woodpecker, and Marguerite. Condition increased with the fall sampling of the Thompson tributaries and the southern mainstem, then condition declined through fall in the late-sampled reaches (North Thompson, Marguerite and Agassiz in 1994). A similar trend was seen for mountain whitefish.

The decline in energy reserves and indices in the fall was particularly evident at Agassiz. In 1994, when Agassiz was sampled in early November, lipid levels were lower (peamouth chub liver 7.20\%; mountain whitefish muscle $1.43 \%$ ) than most other reaches, but in 1995, when Agassiz was sampled in early October, lipid levels were higher (peamouth chub liver 24.3\%; mountain whitefish muscle $5.88 \%$ ) than any other reach except the North Arm (peamouth chub). Furthermore, mean condition was lower for peamouth chub and mountain whitefish sampled from Agassiz (both 0.89) in early November, 1994, than at any other reach, but was high (1.01 and 1.03) in early October, 1995. Mean HSI were also much lower in 1994 (1.53 and 1.11, chub and whitefish, respectively) than 1995 (4.94 and 2.11) at Agassiz.

In 1996, peamouth chub were sampled from the Main Arm in both July and late September to assess the effect of season on indices and health variables. The mean liver lipid level in July (23.17\%) was twice the level in September (11.19\%). Condition and HSI were also higher in July (0.97, 3.09\%) than in September (0.93, 2.19\%).

Fat reserves, condition indices, and HSI in spring-spawning peamouth chub were expected to be highest in the early fall, in preparation for over-wintering and gonad development, and lowest, postspawning, in the spring and early summer. The observed decline in these variables in the fall was therefore unexpected. Low energy reserves would leave peamouth chub in a precarious condition to survive the winter and mature in the spring. The low energy levels seen at Agassiz in the fall of 1994 may have been caused by abnormally high instream temperatures (near $20^{\circ} \mathrm{C}$ ) in the summer of 1994 (Figure 3). Water temperatures were high enough to stress fish and interrupt feeding and metabolism (McPhail 1997) so that fish may not have been able to store lipids for over-wintering. This may also explain the low condition of peamouth chub from the estuary in the summer of 1995 but not the comparatively low lipid levels in the estuary in fall versus summer of 1996.

Mountain whitefish spawn in the fall, therefore the observed declines in lipid levels and condition through the fall were expected as energy is expended for gonad maturation. Similar declines in fat reserves between May and October were observed in mountain whitefish captured from the Athabasca River and between summer and fall from the Wapiti, Smoky and North Saskatchewan rivers (Barton et al. 1993).

## Effect of Maturity

Regressions of mesenteric fat on GSI (arcsine-squareroot), length (log), reach, and sex for peamouth chub collected in 1995 over the entire basin (Appendix 20) were significant for length ( $p=0.03$ ) and
reach $(p<0.001)\left(r^{2}=0.26\right)$. The lack of relationship between mesenteric fat and GSI, indicates that fat reserves were not being used for gonad development. There was a positive relationship between fat and length (slope=2.98) indicating that larger fish have higher fat reserves, possibly related to storage for gonad development over winter or better feeding success by larger fish.

The same regressions for mountain whitefish (Appendix 21) were significant for GSI ( $\mathrm{p}<0.001$ ), year ( $\mathrm{p}<0.001$ ), and reach $(\mathrm{p}<0.001)\left(\mathrm{r}^{2}=0.31\right)$. There was a negative relationship between mesenteric fat and GSI (slope $=-1.28$ ), indicating that fat reserves had been used for gonad development at the time of sampling.

## Reproductive Indicators

The gonadosomatic index (GSI) and egg diameter are indicators of the state of gonad development; generally, the larger the GSI and egg diameters, the more advanced the state of development or maturity. The GSI is the percentage of body weight attributable to the gonads. Fecundity is an indicator of the reproductive capacity of mature fish and is the number of eggs per mature female fish. Age- or size-at-maturity are also important reproductive indicators because the age and size at which fish achieve reproductive maturity affects reproductive capacity of the population (Weatherley and Gill 1987).

Exposure to BKME has been shown to decrease fecundity (Munkittrick and Dixon 1988, Munkittrick et al. 1992a, Munkittrick et al. 1991) and delay maturity (McMaster et al. 1991). Changes in population sex ratios have been also observed in response to exposure to contaminants, such as pulp mill effluent, and in response to many environmental factors, such as temperature, pH , light intensity, and social conditions (Bortone and Davis 1994). Extreme female predominance has also resulted from commercial fishing and high population densities (Brown and Argyle 1987). Conversely, masculinization has been associated with pulp mill effluent exposure, old age, and parasitism (Bortone and Davis 1994). The presence of ova in testes tissue, or "intersex," has been observed in fish in British rivers downstream of sewage outfalls (Jobling et al. 1998).

## Gonadosomatic Index and Age/Size-at-Maturity

GSI are summarized by reach, year, and species in Table 72 to Table 74. Box and whisker plots, by reach, year, and species, of GSI are illustrated in Figure 112 to Figure 113. Two-way ANOVAs were used to test for differences among reaches and between years for GSI for each species (Appendix 22). The year-by-reach interaction effects were significant ( $\mathrm{p}<0.001$ ), therefore, the multiple comparison tests compared samples, which were defined by years and reaches. Scatter plots in Figure 114 to Figure 117 give a clearer picture of the relationship between gonad size and gutted weight by species, sex, reach, and year. Only fall-sampled fish are included in these plots.

Plots of \% mature fish on age and length (Figure 118 to Figure 123) were used to assess age- and size-at-maturity of peamouth chub and mountain whitefish at reaches in the Fraser basin. Age- and size-at-maturity were estimated to be the points where $50 \%$ of the fish were mature. GSI were used
to estimate which fish were maturing (peamouth chub) or mature (mountain whitefish). Age- and size-at-maturity could not be estimated for peamouth chub males, or for females collected post-spawning from the estuary in the summer, because there was no clear definition between immature and maturing gonads. All starry flounder collected in this study were immature, so age- and size-at-maturity could not be estimated.

Data for peamouth chub and starry flounder for 1994, and Main Arm in summer of 1996, should be viewed with caution because gender of immature gonads was not confirmed microscopically. As a result, the gender of many immature fish was unknown. In addition, many immature females may have been misidentified as males because eggs could not be seen in the ovaries. Both factors will bias the results for the reproductive indicators examined in this section, particularly GSI, ages-at-maturity, and sex ratios.

Mean GSI for peamouth chub ranged from 0.87 to $3.78 \%$. GSI were significantly higher than most other reaches at Hansard (3.78\%) in 1994 and at Agassiz (3.25\%) in 1995 (Table 72, Figure 112, Appendix 22). GSI were lower in the middle basin (Woodpecker and Marguerite, 1.40 to 2.07\%) and estuary ( 0.87 to $1.49 \%$ ) than at Hansard in both years. The lower GSI in the estuary compared to other reaches were due to sampling in the summer just after the chub spawned. Accordingly, in 1996 the GSI was higher in September (1.62\%) than July (1.06\%) in the Main Arm. GSI in the Thompson basin did not vary significantly between the two reaches in either year ( 1.37 to $2.62 \%$ ). GSI were generally higher in 1994 than in 1995, because sampling was conducted later in 1994.

Scatter plots of gonad size on gutted weight (Figure 114 and Figure 115) show that the low mean GSI at the Marguerite and Woodpecker reaches resulted from the predominance of immature female peamouth chub in the samples compared to other reaches sampled. While the predominance of immature fish appears to be partly due to sampling smaller fish in these reaches, the plots also show that fish were maturing at smaller sizes in other reaches.

Peamouth chub age-at-maturity ranged from 4 to 9 years for females throughout the Fraser basin (Figure 118). The literature reports that most females mature at age 4 (Hill 1962, Shanbhogue 1976), therefore most Fraser fish are maturing later than the norm. Size-at-maturity ranged from approximately 200 to 235 mm . Age- and size-at-maturity were highest in the central basin Woodpecker and Marguerite, at approximately 9 years and 225-235 mm. Over all, age-at-maturity was highest in the upper mainstem reaches ( 8 to 9 years) where growth rates were lowest, and lowest in the lower Fraser and estuary (4 years) where growth rates were high. Age-at-maturity in the Nechako River and Thompson basin were intermediate (5 to 6 years), despite growth rates as high as those in the estuary. Consequently, sizes-at-maturity were larger in these tributaries (approximately 215 mm ) than in the estuary (approximately 200 mm ) (Figure 119).

Mountain whitefish GSI tended to be higher at Woodpecker (11.69 and 9.38\%), Agassiz (9.20 and $12.10 \%$ ), and the North Thompson (14.38 and $11.08 \%$; significantly higher in both years) than at Hansard ( 7.11 and $6.37 \%$ ), Marguerite ( 0.69 and 3.69\%) , and the Thompson (3.05 and 6.16\%) (Table 73, Figure 113, Appendix 22). Overall, GSI were higher in 1994 than in 1995, because
sampling was conducted later in 1994, closer to spawning. The most striking results were the very low GSI at Marguerite and Thompson, in both years.

Scatter plots of gonad size on gutted weight (Figure 116 and Figure 117) show that gonads of most mountain whitefish sampled from the Marguerite and Thompson reaches were not mature. This was particularly the case for females from Marguerite in 1994. The field sampling crew noted that most of the fish captured at Marguerite in 1994 were spawned out females (Table 11); and field laboratory notes and photographs described small, and sometimes swollen and discoloured ovaries, despite large body size ( 4 to 6 years, 281 to 295 mm ) (Photo 49 to Photo 51). Spawning may explain the low GSI seen in 1994, but sampling was conducted earlier in 1995 to precede the spawning period, and the GSI was still low.

Mountain whitefish age- and size-at-maturity were estimated primarily from samples of spawning aggregations. Therefore estimates should be biased to a younger age and smaller size because we would be sampling only the mature members of the whole population. In addition, results varied substantially between years, indicating, as with other results such as growth rate, that different populations were sampled each year within the same reach. This was particularly true for age-atmaturity for the Hansard ( 3 to 7 years), North Thompson (<2 to 4 years), and Thompson (2 to 5 years) reaches (Figure 120 and Figure 122).

Over the whole basin, mountain whitefish age-at-maturity ranged from $<2$ to 6 years for females (Figure 120) and from <2 to 5 years for males (Figure 122). In most reaches, age-at-maturity was between <2 and 3 years for both females and males. Exceptions occurred at:

- Hansard where growth was slowest and, consequently, ages-at-maturity were high (3 to 7 years).
- Marguerite where female age-at-maturity was estimated at 4 to 8 years, in 1995 and 1994 respectively.
- the Thompson River in 1994 where age-at-maturity was estimated at 4 and 5 years, for females and males, respectively.

Mountain whitefish size-at-maturity ranged from approximately 215 to $>305 \mathrm{~mm}$ (Figure 1211 and Figure 123). Unusually high sizes-at-maturity were seen at the Marguerite ( $>305 \mathrm{~mm}$ ) and Thompson ( 275 mm ) reaches. The one "mature" female captured from Marguerite in 1994 was probably collected post-spawning since although large ( 347 mm fork length), the ovaries weighed just 8.90 g .

Only immature starry flounder were sampled, and mean GSI ranged from $0.10 \%$ to $0.25 \%$ (Table 74).

## Fecundity and Egg Diameter

Fecundity and egg diameter data collected in 1996 for peamouth chub and mountain whitefish at Hansard, Woodpecker, and the Main Arm are presented in Table 76 and Table 77. The variability of fecundity among reaches was tested by ANCOVA using the log of total weight as the covariate (Appendix 23). Northcote and Ennis (1994) reported that mountain whitefish fecundity increased
logarithmically with body size. In this study, the relationship of log fecundity on log weight and reach accounted for the most variability and met the assumptions of homogeneity of variances and normality of data for both mountain whitefish and peamouth chub ( $\mathrm{p}<0.001, \mathrm{r}^{2}=0.62 ; \mathrm{p}<0.001, \mathrm{r}^{2}=0.73$, respectively) (Figure 124). Differences in egg diameters among reaches were tested using ANOVA (Appendix 24).

Peamouth chub log-weight-adjusted mean fecundity estimates were higher in the Main Arm of the estuary $(10,752)$ than in the upper Fraser River at Hansard $(7,391)$ and Woodpecker $(7,226)$ ( $\mathrm{p}=0.08$, Appendix 23; mean weight of 132 g and length of 226 mm ). The unadjusted fecundity estimates at Hansard ranged from 4,640 (209 mm) to 12,536 (259 mm), at Woodpecker from $6,457(227 \mathrm{~mm})$ to $9,147(250 \mathrm{~mm})$, and at the Main Arm from 4,699 $(201 \mathrm{~mm})$ to 13,313 (234 mm , Table 76). Average fecundity estimates in the literature ranged from 5,657 eggs at 215 mm length to 34,841 at 330 mm length (Hill 1962, Shanbhogue 1976). Fecundity estimates ranged from 11,800 ( 300 mm ) to $18,900(325 \mathrm{~mm})$ for 2-month pre-spawning Seeley Lake peamouth chub (Hill 1962) and from $5,657(215 \mathrm{~mm})$ to $34,841(330 \mathrm{~mm})$ for Lake Washington peamouth chub (Shanbhogue 1976). Fecundity appears to be higher than average for the small peamouth chub sampled from the Fraser River, particularly in the estuary.

Mountain whitefish fecundity estimates were not significantly different between Hansard $(3,089)$ and Woodpecker $(2,716)$ when adjusted for the average weight of 225 g (Appendix 23). Fecundity estimates ranged from 1,784 (252 mm) to 6,407 (318 mm) at Hansard and from 1,789 (247 mm) to 3,954 (275 mm) at Woodpecker (Table 77). Fecundity ranges of $1,426(259 \mathrm{~mm})$ to $24,143(495$ mm ) were reported for Montana, and 5,500 (312 g) to 14,000 (680 g) for Utah (Northcote and Ennis 1994). Mountain whitefish from Hansard and Woodpecker are comparable in size and fecundity to the smallest fish reported in the literature.

Egg diameter is dependent primarily on time to spawning, and increases until spawning (Orcutt 1950). Peamouth chub spawn from May to June, therefore were 3 to 4 months post-spawning when captured in September. In a frequency plot of egg size categories, peamouth chub from Hansard have the widest distribution of size categories indicating more numerous stages of egg maturation (Figure 125). Mean egg diameters ranged from $920 \mu \mathrm{~m}$ to $1260 \mu \mathrm{~m}$ at Hansard, $1050 \mu \mathrm{~m}$ to $1150 \mu \mathrm{~m}$ at Woodpecker, and $800 \mu \mathrm{~m}$ to $1080 \mu \mathrm{~m}$ at the Main Arm. Mean egg diameters were significantly smaller in the Main Arm $(950 \mu \mathrm{~m})$ than in the northern reaches of Hansard $(1130 \mu \mathrm{~m})$ and Woodpecker ( $1091 \mu \mathrm{~m}$ ) (p < 0.001, Appendix 24). It is possible that egg development is more advanced in the fall in the northern reaches because less development occurs over the colder northern winters.

Mountain whitefish in BC spawn from mid-October to November (Northcote and Ennis 1994), therefore egg diameters should be near the maximum when sampled in mid-September. A frequency plot of egg size categories indicates that mountain whitefish from Hansard have the widest distribution and that those from Woodpecker contain only the upper size categories (Figure 125). Mean egg diameters ranged from $2050 \mu \mathrm{~m}$ to $2540 \mu \mathrm{~m}$ at Hansard and $2420 \mu \mathrm{~m}$ to $2650 \mu \mathrm{~m}$ at Woodpecker. Egg diameters were significantly larger at Woodpecker (2530 $\mu \mathrm{m}$ ) than at Hansard ( $2327 \mu \mathrm{~m}$ ) (p
<0.001, Appendix 24). Mountain whitefish were found at only one location at Woodpecker -- the mouth of Naver Creek, and may have been a spawning aggregation closer to spawning than those sampled from the Hansard reach.

## Sex Ratios

Ratios of female to male peamouth chub, mountain whitefish, and starry flounder are shown in Table 78; 1994 peamouth chub and starry flounder data are not included in this discussion due to gender misidentification. Sex ratios for peamouth chub and mountain whitefish were near one at most reaches ( 0.8 to 2.0 and 0.5 to 1.7, respectively). Exceptions are a high proportion of peamouth chub females at Woodpecker (2.2 to 4.3) and in the North Arm (5.8 in 1995), and mountain whitefish females at Marguerite ( 2.3 to 4.2) and Agassiz ( 2.2 to 3.2). In almost all samples, females were more numerous than males. Sex ratios varied between years within several reaches (e.g., 1.0 to 1.5 for Hansard and 0.5 to 1.2 for Woodpecker mountain whitefish; 2.2 to 4.3 for Woodpecker peamouth chub). The variability in sex ratios and predominance of females could be caused by habitat segregation of sexes at the time of sampling. However, in Lake Washington, females are favoured in the peamouth chub population, particularly over the age of three (Shanbhogue 1976).

## Intersex

Peamouth chub and starry flounder testes collected in 1995 and 1996 from the estuary (Main Arm and North Arm) were examined for intersex because these reaches are in the large urban Vancouver area and have the highest loading of sewage effluent. No evidence of intersex was seen in peamouth chub but approximately $15 \%$ of the male starry flounder, in both reaches, contained ova in testes tissue (Table 79). This preliminary investigation indicates that further work on endocrine effects in the Fraser River estuary is warranted.

## Summary

In summary, relative gonad sizes were low and age- and size- at-maturity were high for both fish species in the central basin at Marguerite and in the Thompson River compared to upstream reaches. Both of these observations indicate impaired reproductive capacity in these reaches. This apparent reproductive impairment could be caused by several factors, such as natural habitat characteristics, fish behaviour, or contaminant exposure. For example, at Marguerite the river gradient and sediment levels are the highest in the basin, therefore reproductive impairment could be caused by the harsh natural habitat. Regarding fish behaviour, mature mountain whitefish may move out of these reaches to spawn, i.e., into tributaries from the central mainstem Fraser or up the North Thompson from the Thompson River, therefore we could be sampling only from populations that were not spawning (Table 11). The suggestion that mountain whitefish had been residing downstream of the pulp mill in the Thompson River then moved to the North Thompson to spawn is supported by:

- the presence of pulp mill contaminants in tissue of North Thompson mountain whitefish, and
- the large aggregations of very ripe fish captured from the North Thompson; more than 500 fish were captured in one seine set in 1994.

However, the fact that both Marguerite and Thompson 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. The presence of intersex in starry flounder juveniles in the estuary also warrants further research into possible effects of endocrine disrupting chemicals.

## Gut Indices and Contents

## Gut Indices

Two indicators of feeding activity are summarized by species, reach, and year in Table 80 to Table 82 and Figure 126 to Figure 128 -- gut indices based on gut fullness and bile indices based on the colour and fullness rating from the necropsy. The gut index was calculated as:

## Gut Weight/Gutted Weight *100.

Higher gut indices and lower bile values indicate higher feeding activity. As a result, the two indices are negatively correlated ( $p<0.001$, all species; $r=-0.25$ to -0.43 ), and show the same trends in feeding activity in the plots.

Both mountain whitefish and peamouth chub gut and bile indices indicated generally increasing feeding at downstream sites (Figure 126, Figure 127). Feeding activity was higher at most reaches in 1995 compared to 1994, possibly due to sampling earlier in the season in 1995. Feeding activity would be expected to decline through the fall due to declining temperatures, particularly in the northern reaches.

Geographic patterns of peamouth chub gut indices differed somewhat between 1994 and 1995 (Figure 126). In 1994, gut indices were higher in the estuary ( 4.14 to $4.60 \%$ ) than at all upstream reaches ( 3.05 to $3.67 \%$ ). In 1995, gut indices increased in the mainstem Fraser from Hansard (3.92\%) downstream to Agassiz (5.76\%), and in the Thompson basin from the North Thompson ( $4.07 \%$ ) downstream to the Thompson ( $4.67 \%$ ). The gut indices at Agassiz were lower than all other reaches in 1994 (3.05\%) and highest in 1995 (5.76\%), comparable to results for condition and lipid reserves.

Mountain whitefish gut indices showed similar trends in both 1994 and 1995. Indices were highest at Agassiz ( 2.86 to $4.24 \%$ ), Nechako ( 2.30 to $2.59 \%$ ), and Marguerite ( 2.89 to $2.51 \%$ ) in both years, and lowest at the North Thompson (1.48 to 2.09\%), Hansard ( 1.56 to $2.16 \%$ ), Thompson ( 1.60 to $1.73 \%$ ), and Woodpecker ( 1.69 to $1.65 \%$ ) (Figure 127).

Starry flounder gut indices were slightly higher in the North Arm (4.04 to 5.48\%) than the Main Arm ( 3.95 to $4.75 \%$ ), and were higher in 1995 in both reaches (Figure 128). Despite gut indices near 5\%, only 3 starry flounder bile samples were not green, which should indicate that no starry flounder were actively feeding. Perhaps the rating scheme needs to be modified.

## Gut Contents

Gut contents of peamouth chub and starry flounder collected from the Fraser River estuary in 1994 are summarized in Table 83, Table 84, and Figure 129. Increased incidences of marine benthos were consumed by peamouth chub at downstream reaches, with the maximum in the North Arm.
Freshwater benthos dominated the diet in the Main Arm and at Barnston and terrestrial, freshwater, and marine sources were equally important at Mission, although most ( $66 \%$ ) guts were empty at this reach. Marine benthos dominated (>95\%of food items) the diets of starry flounder in both the Main and North arms. Low contaminant levels found in starry flounder may be due, at least in part, to the flounders feeding in salt water, outside the influence of river pollution.

## Necropsy and Health Assessment Index

The necropsy-based fish health assessment and the resultant health assessment index (HAI) were used in this study as another potential indicator of the response of fish to contaminants. Poorer health results in a higher health assessment rating or index. 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, Schlenk et al. 1996). Abnormalities assessed in the HAI have also been associated with environmental factors such as habitat degradation - low dissolved oxygen, high temperature, high microbial populations, or natural factors such as parasitism, diet, or disease (Table 85). Health assessment index (HAI) and necropsy results are plotted in Figure 130 to Figure 141 and summarized in Table 86 to Table 103.

## Assessor Variability

In 1996, health assessments were conducted by two assessors independently on the same fish to evaluate the quality of the data. Duplicate assessments are plotted side-by-side in Figure 130 to Figure 141. Generally, duplicate assessments were very similar, indicating good data quality. Variability between assessors is described in Table 89, with assessments contributing significant variability to the overall HAI bolded. A two-way ANOVA -- assessor by reach, indicated that differences between assessments were statistically significant only for gill condition (Appendix 25). The assessor by reach interaction was significant for mountain whitefish HAI, mesenteric fat, and hindgut tests, i.e. the effect of assessor varied between the two reaches (Figure 130, Figure 137, and Figure 139).

It is interesting that in previous years the incidences of frayed gills and nodular spleens were low compared to Assessor 2's evaluations in 1996, suggesting that assessments in each year was being done differently. The variability in assessments is greater than in previous years, particularly for internal assessments, probably because the protocol was changed to accommodate the quality assurance (QA) work. In previous years, the recorder would check the assessments of the dissectors, so that two people would need to agree on the assessment before it was recorded. During the QA test, the two recorders did independent assessments without dialogue between dissectors and recorders so that assessments could not be overheard. This increases the possibility of erroneous and
missed abnormalities. Based on similarities in duplicate assessments and the two-person confirmations done in previous years, the data for 1994 to 1995 are expected to be accurate and repeatable.

## Health Assessment Index

HAIs ranged from 31.5 to 76.0 for peamouth chub, 4.0 to 47.3 for mountain whitefish, and 26.3 to 44.5 for starry flounder. 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 (Table 70, Figure 130, Table 90). In fact, HAIs at reference reaches, where contaminant levels were low, were generally higher than or comparable to other reaches in the basin; and the highest HAIs occurred in the Nechako River, due primarily to heavy parasite infestations. However, the next highest HAI were found downstream of contaminant inputs from Prince George - at Woodpecker and Marguerite for peamouth chub and at Marguerite for mountain whitefish.

In addition, the geographic pattern in the HAI (Figure 130) differed from that of growth, lipid levels, and condition (Figure 102 to Figure 103, Figure 105 to Figure 107, Figure 108 to Figure 110). Therefore, in this study, elevated HAI cannot be linked to decreased growth and condition.

Other studies have found a significant relationship between mountain whitefish health and age (Antcliffe et al. 1997, Nener et al. 1995), therefore, high HAI at northern reference reaches, where older fish were sampled, could be due to age. However, for this study, regressions of HAI on the log of age were either weak or insignificant (Table 91).

Key HAI results were:

- high levels of parasitism in the Nechako River in both mountain whitefish and peamouth chub.
- prevalence of discoloured livers or kidneys downstream of Prince George in both mountain whitefish (1994 and 1996) and peamouth chub, particularly dark (brown, grey, or black) discolouration. 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); (Antcliffe et al. 1997) and in peamouth chub sampled for the pulp mill EEM program at Prince George, Woodpecker, and Quesnel (Hatfield Consultants Ltd. 1996a).
- similar starry flounder HAI between the North and Main arms in 1994, and higher HAI in the North Arm in 1995, the increase due primarily to discoloured and fatty livers.
- high incidences of urolithic kidneys in mountain whitefish from Hansard and Woodpecker in 1996.
- higher HAI in 1994 than 1995 for all species, possibly related to lower flows and higher water temperatures in 1994, or natural cycles in parasitism.

ANOVAs to test for differences among reaches and years had significant reach by year interactions for all species ( $\mathrm{p}<0.001$ ) (Appendix 26). HAIs and their component ratings varied between species,
among reaches, and even between years within the same reaches, confounding data interpretation. Examples of this variability are:

- peamouth chub HAIs were higher than mountain whitefish primarily because all peamouth 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.
- HAIs and parasite ratings were highest in 1994, as mentioned above.
- mountain whitefish from Marguerite had the second highest HAIs (after Nechako) in both 1994 and 1995, the result of different abnormalities -- discoloured livers in 1994, and inflamed hindguts in 1995.


## Fin Erosion

Incidences of fin erosion were below 5\% at most reaches and contributed minimally to the total HAI (Figure 131, Table 92). Highest incidences occurred in peamouth chub from the Nechako River ( $8.1 \%$ ) and the estuary ( $3.3 \%$ to 6.5\%) in 1994; in peamouth chub from the Main Arm in 1996 ( $7.7 \%$ in the summer and $5.7 \%$ in the fall, assessor 2 only); and in mountain whitefish from the Nechako (8.3\%) in 1995. Starry flounder had a low incidence of fin erosion ( $1.5 \%$ and $1.6 \%$ ) in the Main Arm only, in both years.

## Skin Lesions

As with fin erosion, skin lesions were a minor component of the total HAI and incidences of lesions were low, less than 10\%, at most reaches (Figure 132, Table 93, Photo 52 and Photo 53). Highest incidences occurred in peamouth chub from the Nechako ( $24.2 \%$ ) in 1994, in mountain whitefish from Hansard ( $32.2 \%$ ) in 1994, and in mountain whitefish from the Nechako (15.0\%) in 1995. Peamouth chub in the estuary had moderately high incidences of lesions in both arms and years ( $6.5 \%$ to $10.0 \%$ ). Incidences of lesions in starry flounder were high in the Main Arm (14.5\%) and North Arm (19.4\%) in 1994, but were very low in 1995 (1.5\% and 0\%).

High incidences of mountain whitefish skin lesions were associated with pulp mill effluent exposure in the Athabasca River in Alberta (Barton 1994), but lesions were seen only during the spring and not the fall sampling. In our study, incidences of skin lesions were highest in reference reaches, but sampling was conducted downstream of the pulp mills only in the fall.

## Gills

Very low incidences of gill abnormalities, except frayed gills, occurred at all reaches and for all species (Figure 133, Table 94). Frayed gills were the most common abnormality - seen in $20 \%$ of peamouth chub captured at Hansard in 1994, 13.3\% and 1.9 (26.9)\% (Assessor 1(2)) of mountain whitefish at Hansard in 1994 and 1996, and 7.9 (18.9)\% of mountain whitefish at Woodpecker in 1996. As noted in Table 89, the high incidences of frayed gills were due probably to misidentification of separated lamellae as "frayed" (Photo 54).

## Parasites

The effect of parasites on the health and fate of fish varies enormously depending on the type of parasite and the degree of infestation. Some parasites have little effect on the fish even with a heavy infestation, e.g., encysted nematode worms, while others result in severe effects, e.g., the tapeworm Ligula intestinalis (Northcote 1957). "Parasites" in Figure 130 includes the parasite assessment (Figure 134, Table 95), nodular livers (Figure 136), and parasitized kidneys (Figure 140 and Figure 141). Parasites found in this study are summarized in Table 96.

Parasitism varied greatly at each reach from year to year and was heaviest in all species in 1994. The largest parasite infestations in 1994 affected:

- peamouth chub from the Nechako, with black spot, internal cysts, and nodular livers.
- peamouth chub from the Thompson and North Thompson rivers, where larval tapeworms (L. intestinalis) infested $28.3 \%$ and $26.7 \%$, respectively, of the peamouth chub (Table 97). These worms filled the body cavities and entwined around organs (Photo 55, Photo 56).
- mountain whitefish from the Nechako River, with worms and cysts filling the body cavity and air bladder and covering internal organs (Photo 51), and flukes and copepods on the gills.
- mountain whitefish from the Thompson River, with internal cysts and worms, and gill flukes and copepods.
- mountain whitefish from Marguerite, with gill flukes, black spot, and encysted worms.
- starry flounder from the estuary, with cysts and gut parasites affecting $100 \%$ in the Main and $87.1 \%$ in the North arms, and nodules in the livers of $43.5 \%$ and $27.4 \%$, respectively. Most of the starry flounder HAI in both years was due to parasites and nodular livers. Parasite assessments in 1995 were similar to those in 1994, with slightly lower incidences but higher severity.

Incidence of larval tapeworms in peamouth chub decreased to $6.7 \%$ in 1995 in the Thompson but incidence increased to $46.7 \%$ in the North Thompson (Table 97). Worm weights were not measured in 1994, and were estimated by subtracting organ and gutted weights from the total weights. These tapeworms can have severe health effects, i.e., the fish become sluggish and more susceptible to predation and are eventually consumed by the parasite's adult host (Northcote 1957). Thompson fish had a larger estimated worm weight than North Thompson fish in 1994, and it is possible that infested Thompson fish were eliminated from the population by 1995. The tapeworms also appear to affect reproduction; peamouth chub with tapeworms had lower GSI for the same age than those without tapeworms (Figure 142).

It should be noted that parasite cysts occurring in organs strongly affect the HAI. For example, a peamouth chub from the Nechako with a cyst in each of the liver, kidney, and spleen would receive an HAI rating of 100 (nodular liver 'D'--> 30, nodular spleen 'D'-->30, kidney 'OT'-->30, and
parasite rating 1-->10), whereas a huge larval tapeworm in the peamouth chub of the Thompson would receive a rating of only 30 , provided the organs were not affected. Therefore, the Nechako HAI resulting primarily from encysted worms was always highest, but the consequences to the fish of this high HAI may not be as severe as a lower HAI, for example, on the Thompson for peamouth chub which are infested by larval tapeworms.

## Bile

Bile assessments are presented in Figure 135 and Table 98 but not included in the calculated HAI and were discussed previously as an indicator of feeding activity.

## Liver

Abnormal livers were common in peamouth chub (Figure 136, Table 99). There were high incidences of nodular livers in the Nechako River ( $58.1 \%$ and $11.7 \%$, in 1994 and 1995) corresponding to high parasite loads. Discoloured livers were prevalent downstream of Prince George at Woodpecker $(85.0 \%, 36.7 \%$, and $61.8 \%$ in 1994, 1995, and 1996) and Marguerite ( $70.0 \%$ and $36.7 \%$ in 1994 and 1995) (Photo 57, Photo 60).

Fatty livers were not considered abnormal in peamouth chub in this study because livers are a major fat storage depot in cyprinids, so a fatty liver was considered a sign of lipid storage rather than liver degeneration. Fatty livers were prevalent in peamouth chub, and were rare only in 1994 at the Woodpecker (6.7\%) and Marguerite (10.0\%) reaches downstream of Prince George (Photo 58). More fatty livers were seen in 1995 when fish were sampled earlier in the year. Incidences of fatty livers showed the same trend as mesenteric fat and liver lipid levels, which were discussed in the section on lipid reserves.

Abnormal livers were rare in mountain whitefish (Figure 136, Table 99). High incidences of abnormal livers were found only in 1994 at Marguerite ( $61.3 \%$ discoloured) and Agassiz ( $4 \%$ nodular, $16 \%$ fatty, and $20 \%$ discoloured). At Marguerite in 1994, two livers were coded "discoloured" due to grey discolouration, while most were due to thinning around the edges. Some "discoloured" livers at Agassiz also had lighter edge colouration, which may be a "normal" condition. In contrast, $100 \%$ of Marguerite mountain whitefish livers were described as normal in 1995.

Abnormal livers were common in starry flounder (Figure 136, Table 99), including:

- nodular livers in both arms (16.9\% Main Arm 1995 to 43.5\% Main Arm 1994 ).
- fatty livers in the North Arm ( $25.8 \%$ and $55.0 \%$ in 1994 and 1995).
- discoloured livers in the North Arm (12.9\% and $25.0 \%$ in 1994 and 1995).


## Mesenteric Fat

Mesenteric fat ratings are presented in Figure 137 and Table 100. The mesenteric fat index is not included in the calculated HAI and was discussed in the section on lipid reserves.

## Spleen

Almost all peamouth chub, mountain whitefish, and starry flounder spleens were normal (Figure 138, Table 101). High incidences of enlarged spleens were observed only in peamouth chub from the North Thompson in 1994 (25.0\%) and North Arm in 1995 (13.3\%).

The variability between assessments of mountain whitefish spleens at Hansard in 1996 was large $5.8 \%$ nodular by assessor 1 versus $17.3 \%$ by assessor 2 . Such a high incidence of nodular spleens was not seen in any other samples.

## Hindgut

Hindgut assessments for peamouth chub and mountain whitefish were highly variable between samples from each reach, species, and year (Figure 139, Table 102). No consistent trends were noted, with high mean assessments seen in 1995 at reaches where assessments were low in 1994, and vice versa.

Starry founder hindgut assessments were higher in the Main Arm than the North Arm in both years (Figure 139).

## Kidney

Peamouth chub kidneys were not dark red and lying relatively flat dorsally, i.e., the "normal," salmonid form used by the necropsy (Figure 140, Table 103). All kidneys were swollen compared to this standard, and most were pale and pink (Photo 61). As mentioned above, this condition may be normal for a cyprinid, or may be caused by a myxosporidean parasite that infested all the peamouth chub. The most notable results were the high incidences of brown to grey kidneys (Photo 62) downstream of Prince George at Woodpecker (23.3\%, 38.3\%, 36.4-43.4\% in 1994, 1995, and 1996) and Marguerite ( $16.7 \%$ and $50.0 \%$ in 1994 and 1995), corresponding to the darkly discoloured livers at these same reaches.

Mountain whitefish from the Nechako River in 1994 had the highest incidence of abnormal kidneys (60\%), including parasitized kidneys (23.3\%) (Figure 141, Table 103). High incidences of abnormal kidneys were also found in 1994 at Hansard (23.3\%), Marguerite (38.7\%), and Thompson (25.6\%), distributed over all abnormality types (Photo 63). In 1995 almost all kidneys were normal. In 1996, there were high incidences of urolithic kidneys (Photo 64) at Hansard (19.2\%) and Woodpecker (28.9\%), the only reaches sampled.

All starry flounder kidneys were normal, except two mottled (1994) and one granular (1995) in the North Arm (Figure 141, Table 103).

## Histology

## Blind Duplicates

Blind duplicate readings of the same slides by two histologists tested the repeatability of the results. In 1994, all slides were read by one histologist and a second histologist re-read a subset of the slides. The duplicate readings in 1994 were very different and resulted in an attempt to better standardize and define abnormalities. Subsequently, the first histologist read the 1995 and 1996 slides, and the second histologist re-read the 1994 slides, both using the standardized method. The results presented in Table 104 are for duplicate readings done in 1995, after the method was refined; no further duplicate readings were done on 1994 and 1996 slides. Because the results obtained by the two histologists were still somewhat different, 1994 data cannot be directly compared to that for 1995 and 1996.

The differences in the readings were most likely the result of the histologists having experience with different types of studies and effects. In addition, one histologist noted that the samples generally looked healthy; had the abnormalities been more severe, the assessment of what was abnormal versus normal tissue may have been clearer and therefore more consistent. These differences underscore the need to establish a standardized and detailed methodology before initiating histological analyses.

## Field Splits

Results from analysis of field-split samples are presented in Table 105. Only one tissue section per sample was analyzed, therefore, poor agreement between splits likely resulted from the inadequacy of one section to characterize a whole tissue or organ. Good agreement for gill hyperplasia and aneurysms, abnormalities that affect the entire gills versus poor agreement for parasites, which could be easily missed when taking only a single section, supports this assumption.

## Relationship Between Histology and Necropsy Results

Microscopic histological and gross necropsy results were not well correlated (Table 106). Increased incidences of histological abnormalities were sometimes seen in fish with gross abnormalities, however the necropsy and histological incidences of abnormalities were never the same, for example, only 4.5\% of peamouth chub "black/grey" kidneys have melanosis ( $0 \%$ of "normal" kidneys have melanosis), and only $48.1 \%$ of peamouth chub "nodular" livers have helminths ( $14.1 \%$ of "normal" livers have helminths). Furthermore, some expected associations were not seen, e.g., no increases in fatty or glycogen vacuolation, or other vacuolation, were associated with "coffee with cream" (fatty) livers.

The following table lists necropsy and histological abnormalities that were somewhat associated with each other, based on increased incidences of histological abnormalities in fish with necropsy abnormalities.

| Organ | Necropsy Abnormality | Histological Abnormality |
| :--- | :--- | :--- |
| GILLS | Frayed | Hyperplasia |
| LIVER | Nodular | Helminths, granulomas, and glycogen |
|  |  | vacuolation |
|  | Granular | Helminths and glycogen vacuolation |
| SPLEEN | Enlarged and granular | Hemorrhage and hemosideran |
|  | Nodular | Melanosis (MW) and Helminths (PC) |
| KIDNEY | Mottled | Lipid infiltration |
|  | All (except swollen and asymmetrical) | Melanosis |
| HINDGUT | Severe inflammation | Inflammation and granulomas |

## Gills

Hyperplasia was the most common gill abnormality, occurring in up to $100 \%$ of peamouth chub (Hansard, 1995), $71 \%$ of mountain whitefish (Hansard, 1994), and $70 \%$ of starry flounder (North Arm, 1994) (Table 107, Figure 143, Photo 40 to Photo 42). In mountain whitefish and starry flounder, hyperplasia was more prevalent in 1994 than in other years. Hyperplasia can be caused by exposure to poor water quality and high silt levels (Bagshaw 1997, Brand 1998). When hyperplasia occurs as discrete pockets, it is likely associated with some physical agent, whereas chemical damage generally takes on a more diffuse appearance (Bagshaw 1997). The hyperplasia was characteristically non-focal in these fish, that is, there was one or more completely hyperplastic lamella per tissue section, and was often severe enough to cause fusion of the lamellae (Brand 1998). The characteristic non-focal nature of the hyperplasia suggests that the agents of stress are more likely chemical rather than physical.

Incidences of hyperplasia were not highest in reaches where suspended sediment levels were highest, suggesting that a water quality condition other than high suspended sediment levels was causing the hyperplasia. Also, because incidences of hyperplasia were highest in most northern reaches where older fish were sampled, age may be a factor in the development of hyperplasia.

Parasites were also common on the gills of peamouth chub and mountain whitefish (Photo 38, Photo 43 to Photo 48). Up to $32 \%$ of peamouth chub (North Arm, 1994) and mountain whitefish (Agassiz, 1994) had helminths and $33 \%$ of peamouth chub had protozoans (North Thompson, 1994). Protozoans were rare in mountain whitefish, and parasites were rare in starry flounder.

Aneurysms were also very common, but were thought to be caused by physical trauma during handling. In 1996, when fish were killed by anesthetic, aneurysms were rare. All other abnormalities inflammation, necrosis, thrombosis, chubbing, hypertrophy, epithelial lifting, and branching, were rare.

## Liver

Liver abnormalities are presented in Photo 19 to Photo 36. The most common condition in peamouth chub was glycogen or fatty vacuolation (incidence up to $93 \%$ at Hansard in 1995) (Table 108, Figure 144 , Photo 22) which is associated with normal storage of energy in the liver. However, glycogen or
fatty vacuolation did not appear to be related to the levels of lipid in the liver, e.g. fatty/glycogen vacuolation was common at Marguerite in 1994 (83\%) where liver lipid levels (5.3\%) were lowest, and rare at Agassiz in 1995 (1.6\%) where liver lipid levels (24.3\%) were highest. Helminths were also common in the livers of peamouth chub (up to $92 \%$ at Nechako in 1995). Inflammation and granulomas occurred at similar levels within each reach and were probably due to parasitism (Brand 1998).

The most common abnormality in mountain whitefish was inflammation - in up to $56 \%$ of livers from the Thompson River in 1994 (Table 108, Figure 145, Photo 25 to Photo 27). All other abnormalities were rare.

Abnormalities were more rare in starry flounder. Helminths, granulomas, and inflammation, the most common abnormalities, were slightly more common in the Main Arm than North Arm (Figure 145).

Observation of hydropic vacuolation and hepatocellular steatosis (Photo 23 to Photo 25) in 1994 only was attributable to the histologist. Hydropic vacuolation, which has been associated with neoplasm development and contaminant exposure in Puget Sound (Myers et al. 1992, Myers et al. 1994), was most prevalent in peamouth chub at Agassiz (42\%) and in the Main Arm (34\%). Hydropic vacuolation in mountain whitefish was observed only at Agassiz (5.3\%) and in the Thompson basin ( $1.4 \%$ to $3.4 \%$ ), and in starry flounder in the North Arm (5.7\%). Hepatocellular steatosis, a degenerative lesion suggestive of a metabolic disorder which has been associated with both dietary deficiencies and exposure to toxic chemicals (Myers et al. 1987), was seen primarily in peamouth chub at Hansard ( $32 \%$ ) and in the lower Fraser ( $21 \%$ to $38 \%$ ), and in starry flounder in both the Main (9.7\%) and North (13\%) arms.

## Spleen

Spleen abnormalities are presented in Photo 1 to Photo 6. Close to $100 \%$ of peamouth chub spleens were infested by the protozoan Myxobolus cyprini (Photo 2, Table 109, Figure 146). Infested organs exhibited diffuse, chronic inflammation consisting primarily of macrophages but it is not known what impact the parasite had on the health of the fish (Kent et al. 1996). In 1994, spleen melanosis/melanin was common in peamouth chub downstream of Prince George at Woodpecker $(22 \%)$ and Marguerite (34\%). These were the same locations where black discolouration of the livers was also observed. Hemosiderin was observed in peamouth chub spleens from all reaches in 1994, with highest incidences in the North Thompson ( $78 \%$ ) and the mainstem from Hansard to Barnston ( $10 \%$ to $19 \%$, except at Marguerite ( $1.7 \%$ )). Helminths were also seen in spleens but at incidences less than $9 \%$ in all samples except Nechako in 1995 (22\%).

The melanosis/melanin and hemosiderin were the only abnormalities observed frequently in mountain whitefish spleens, and incidences were highest in 1994 (Table 109). Incidences of melanosis ranged from $4.3 \%$ in the Thompson River to $45 \%$ at Woodpecker in 1994; and incidences of hemosiderin ranged from 0\% (Marguerite, Agassiz, and Thompson) to 20\% in the Nechako River (Photo 4, Photo 5).

## Kidney

Kidney abnormalities are presented in Photo 13 to Photo 18. Results for peamouth chub kidneys were similar to those for spleen (Table 110, Figure 147), i.e., close to $100 \%$ of kidneys were infested by M. cyprini (Photo 14), melanosis was seen only at Woodpecker (3.4\%) and Marguerite (5.1\%) in 1994 (Photo 16), and helminths were common, with highest incidences at Nechako ( $20 \%$ to $26 \%$ ) and Agassiz ( $2 \%$ to $20 \%$ ). Patterns of occurrence of melanosis in mountain whitefish kidneys were also similar to those in spleens -- ranged from $5.7 \%$ in the Thompson to $34 \%$ in the Nechako in 1994 ( $31 \%$ at Marguerite). All starry flounder kidneys were normal.

## Hindgut

The most common hindgut abnormalities were helminths, M. cyprini (peamouth chub only), and inflammation (Table 111, Figure 148, Photo 7 to Photo 12). In peamouth chub, abnormalities, especially parasites, were most common in the Nechako River (helminths in $21 \%$ to $24 \%$ and $M$. cyprini in $0 \%$ to $25 \%$ ). Few abnormal hindguts were seen in mountain whitefish and starry flounder. Helminths (up to $5.0 \%$ ) and inflammation (up to $6.7 \%$ ) were found in mountain whitefish only at Nechako, Woodpecker, and in the Thompson basin.

## Gonads

Only 1995 data for gonad histology are presented in Table 112. Few abnormalities were observed bizarre mitosis in two mountain whitefish from the Thompson, and helminths in 7 peamouth chub from the Nechako and 2 starry flounder from the Main Arm.

## Conclusions and Recommendations

Peamouth chub and mountain whitefish were used successfully in assessing contaminant exposure to fish in the Fraser basin - the two fish species were captured in river reaches throughout the basin and showed differences in both contaminant levels and MFO induction between sites upstream and downstream of contaminant sources. Mountain whitefish were most abundant in reaches in the upper basin - Nechako River, Hansard, and North Thompson. Peamouth chub were more evenly distributed amongst the reaches, with the highest catches observed in the estuary. Starry flounder were useful as an indicator only of conditions in the estuary, since the species is not found outside of waters with marine influence.

Levels of most contaminants in both liver and muscle tissues were low throughout the basin, compared to either historical levels or available tissue residue guidelines for protection of humans or wildlife consumers of fish tissue. In many cases, concentrations in tissues were at or near analytical detection limits. Improvements in effluent control, process changes by pulp and paper mills, and local prohibition of persistent organochlorines have succeeded in reducing environmental release of many contaminants. However, residues of many chemicals, particularly of the persistent organochlorines, continue to be detected in fish tissues. MFO activity, indicative of contaminant effects, was still found to be elevated downstream of known point-source discharges in the basin. The cumulative effect of these contaminants and their metabolic transformation products should remain a subject of some concern.

Chlorinated dioxins and furans, toxaphene, and DDE residues may still pose a risk to piscivorous wildlife (such as osprey, eagles, mink) in some reaches. Dioxin concentrations, expressed in terms of 2,3,7,8-TCDD TEQs using recently derived World Health Organization TEF values, in many reaches far exceed the current CCME guideline level of $0.71 \mathrm{pg} / \mathrm{g}$ w.w. for protection of wildlife consuming contaminated tissue. Fish tissues collected from reaches upstream of Hope showed a clear dominance of pulp-mill-related dioxin and furan congeners, reflecting the dominance of these discharges in the upper basin. In the lower Fraser reaches, a wider array of dioxin and furan congeners were detected, a consequence of the greater diversity of potential contaminant sources. In addition, detectable residues of dioxin-like PCB congeners were found in many reaches, which could further contribute to the total "dioxin" TEQs and exacerbate potential toxic effects. Toxaphene and DDE were measured in fish at levels which exceeded current CCME tissue residue guidelines for protection of wildlife. Most PCBs and pesticides measured in this study are banned or have limited application in North America, although many continue to be used elsewhere in the world. Present levels in the Fraser basin probably result from a combination of both atmospheric transport from distant source areas and more local contamination from historical use, with subsequent food chain biomagnification and accumulation. These residues might be expected to persist into the near future, and therefore probably represent the present "background" levels.

Increasing use and emission of some compounds continues to be of concern. Some PAHs in fish tissue in the estuary were higher than historic levels, probably due to the continuing population growth
in the Lower Fraser Valley. Releases of PAHs to the Fraser River from residential runoff, stormwater, municipal wastewater plants and atmospheric emissions will be of continuing concern. Monitoring of levels and effects of these compounds must continue to assess the effectiveness of implemented urban pollution abatement activities.

Other chemicals with potentially more subtle effects are being discharged. These include the array of surfactants and other compounds which have been recently implicated as potential disrupters of endocrine function by mimicking or interfering with normal hormone action. In addition, pesticides in current use, though far less persistent than their predecessors, may have undetermined effects on biota. Study of environmental levels and effects of these "new" contaminants is necessary.

No clear relationships were observed between elevated contaminant or MFO levels and health measures such as elevated HAI or decreased energy reserves, condition, and growth. This may be because contaminant levels in the basin are too low or natural factors, such as flow, sediment loads, and temperature are more important in determining the fish health in the basin. Peamouth chub energy reserves, condition, and growth rates were highest in the Nechako, Thompson, and lower Fraser rivers, probably due to higher temperatures and lower sediment levels. Impaired fish health seen at Marguerite may be due of contaminant exposure, but was more likely resulting from natural high suspended sediment levels and river gradient. Trends in relevant biological variables in mountain whitefish differed and were more variable than found in peamouth chub, possibly due to fish movements. Whitefish can migrate long distances and may not be ideal indicators of local conditions although they may be useful in integrating effects of extensive river reaches. Energy reserves and condition in mountain whitefish were generally highest in the Thompson and North Thompson rivers, and similar among other reaches. Growth rates increased from north to south (upstream to downstream), indicating that latitude and altitude were the primary controlling factors.

HAI values at upstream reference reaches were comparable to downstream reaches for both species. The highest HAI values, indicating poorest overall health, were found in the Nechako River and were due to parasite infestations. HAI values were not correlated with other health indicators such as growth rate and condition.

Evidence for reproductive impairment in both species was found in the (central) middle Fraser reach at Marguerite and in the Thompson River through analysis of gonadosomatic index, age-at-maturity, and size-at-maturity. This impairment could be the result of several factors, such as the increased suspended sediment and river gradient at Marguerite or the behaviour of fish migrating elsewhere to spawn. However, both reaches are immediately downstream of discharges from both pulp mills and major urban centres suggesting that contaminant exposure could possibly be contributing to this impairment.

Intersex, an indicator of endocrine disruption, was observed in the testes of starry flounder from the estuary. However, no reference sites were sampled for starry flounder so the natural background incidence of intersex in this species is unknown. Consequently, this result requires further investigation to determine if the intersex is being caused by endocrine disrupting compounds.

Contaminant levels in peamouth chub and mountain whitefish changed little over the three years of this study. Therefore, these data represent a good baseline of contaminant levels in these three fish species in the Fraser basin. Greater year-to-year variability was observed in the fish condition and health measures and more sampling is recommended before a baseline can be established. The interpretation of contaminant and fish health data will be improved through research on patterns of fish movement and coupled with study of geographic and temporal variability in parasitism, condition, biochemistry, physiology, reproduction, and growth.

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## Tables

Table 1. Mean annual suspended sediment and mean high and low flow suspended solids concentrations at stations on the Fraser River and its tributaries.

| Site | Mean Annual <br> Suspended Sediment |  | Suspended Solids (mg/L) |  |
| :--- | :---: | :---: | :---: | :---: |
|  | (mg/L) | Period* $^{*}$ | High Flow | Low Flow |
| Nechako River | 83 | $1976-80$ | 30 | - |
| Hansard | 132 | $1974-86$ | 147 | 18 |
| Marguerite | 124 | $1966-79$ | 137 | 74 |
| Hope |  |  | 108 | 30 |
| Main Arm (Estuary) |  |  | 43 | 12 |
| Thompson River |  | 25 | - |  |
| North Thompson River |  |  | 4 |  |

*Years for which daily suspended sediment measurements were taken through the entire year.
From Environment Canada (1997) and Hall (1991).

Table 2. River gradients in the Fraser River mainstem from Hansard to Mission.

| Reach | Slope |
| :--- | :--- |
| Hansard | $17 \mathrm{~m} / 100 \mathrm{~km}$ |
| Prince George to Marguerite | $64 \mathrm{~m} / 100 \mathrm{~km}$ |
| Marguerite to Hope | $104 \mathrm{~m} / 100 \mathrm{~km}$ |
| Hope | $60 \mathrm{~m} / 100 \mathrm{~km}$ |
| Agassiz | $48 \mathrm{~m} / 100 \mathrm{~km}$ |
| Mission | $5 \mathrm{~m} / 100 \mathrm{~km}$ |

From Carson (1988)

Table 3. Sampling periods for the Fraser basin 1994 to 1996.

| Year | Reach | Dates |
| :--- | :--- | :--- |
| 1994 | Nechako | Sept. 25-28 |
|  | Hansard | Sept. 14-19 |
|  | Woodpecker | Sept. 21-23 |
|  | Marguerite | Oct. 24-30 |
|  | Agassiz | Nov. 02-06 |
|  | Mission | Aug. 18-23 |
|  | Barnston | Aug. 11-17 |
|  | Main Arm | July 28-29, Aug. 09-10 |
|  | North Arm | July 26-29, Aug. 02-05 |
|  | North Thompson | Oct. 15-19 |
|  | Thompson | Sept. 30-Oct. 04, Oct. 11-13 |
| 1995 | Nechako | Aug. 29-Sept. 05 |
|  | Hansard | Aug. 22-28 |
|  | Woodpecker | Sept. 06-09 |
|  | Marguerite | Sept. 10-13, Oct. 03-04 |
|  | Agassiz | Oct. 05-06, Oct. 11-17 |
|  | Main Arm | July 31-Aug. 3 |
|  | North Arm | July 24-28 |
|  | North Thompson | Sept. 19-25 |
|  | Thompson | Sept. 26-28 |
| 1996 | Hansard | Sept. 10-13 |
|  | Woodpecker | Sept. 14-19 |
|  | Main Arm - Summer | July 22-24 |
|  | Main Arm - Fall | Sept. 24-26 |

Table 4. Number of fish collected for analyses per reach in the Fraser basin 1994 to 1996.

| Reach | Peamouth Chub |  |  | Mountain Whitefish |  |  | Starry Flounder |  |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | 1994 | 1995 | 1996 | 1994 | 1995 | 1996 | 1994 | 1995 |
|  | 62 | 60 | - | 60 | 60 | - | - | - |
| Hansard | 60 | 56 | 60 | 60 | 60 | 52 | - | - |
| Woodpecker | 60 | 60 | 55 | 64 | 60 | 38 | - | - |
| Marguerite | 60 | 30 | - | 31 | 14 | - | - | - |
| Agassiz | 60 | 61 | - | 25 | 45 | - | - | - |
| Mission | 57 | - | - | - | - | - | - | - |
| Barnston | 62 | - | - | - | - | - | - | - |
| Main Arm | 62 | 60 | 60 | - | - | - | 62 | 62 |
| Main Arm (Sep) | - | - | 53 | - | - | - | - | - |
| North Arm | 62 | 60 | - | - | - | - | 65 | 60 |
| N. Thompson | 60 | 60 | - | 60 | 60 | - | - | - |
| Thompson | 60 | 60 | - | 78 | 60 | - | - | - |

Table 5. Analyses conducted on tissues from Fraser River fish during 1994 and 1995. Where not specified, tissue were muscle and liver composites. Species designations refer to peamouth chub (PC), mountain whitefish (MW) and starry flounder (SF).

| Reach | Year | Dioxins Furans | Chorophenols |  |  |  | $\begin{gathered} \text { Pesticide } \\ s \\ \hline \end{gathered}$ | Metals | PAHs |  | Resin <br> Acids |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  | Tissue | Bile | $\begin{gathered} \text { Congener } \\ \text { s } \end{gathered}$ | Coplana $\mathbf{r}$ |  |  | Tissue | Bile |  |
| Nechako | $\begin{aligned} & \hline 1994 \\ & 1995 \end{aligned}$ |  | PC, MW |  | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | PC, MW PC, MW | PC, MW PC, MW | PC, MW PC, MW |  | PC,MW |  |
| Hansard | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC, MW | PC, MW | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | PC, MW PC, MW | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ |  | PC,MW | $\begin{gathered} \mathrm{PC} \\ \mathrm{PC,MW} \end{gathered}$ |
| Woodpecker | $\begin{aligned} & \hline 1994 \\ & 1995 \end{aligned}$ | PC, MW | PC, MW | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | PC, MW PC, MW | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ |  | PC,MW | $\begin{aligned} & \text { PC,MW } \\ & \text { PC,MW } \end{aligned}$ |
| Marguerite | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC, MW | PC, MW | $\begin{array}{\|c} \hline \text { PC,MW } \\ \text { MW } \end{array}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ |  | PC,MW | PC |
| Agassiz | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC, MW | PC, MW |  | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ |  | PC,MW |  |
| Mission | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC | PC |  | PC | PC | PC | PC | PC, SF |  |  |
| Barnston | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC | PC |  | PC | PC | PC | PC | PC, SF |  |  |
| Main Arm | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC, SF | PC, SF | PC, SF | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | PC, SF |  | PC,SF |
| North Arm | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC, SF | PC, SF | PC, SF | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{SF} \\ & \mathrm{PC}, \mathrm{SF} \end{aligned}$ | PC, SF |  | PC,SF |
| North <br> Thompson | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC, MW | PC, MW | $\begin{array}{\|l\|} \hline \mathrm{PC}, \mathrm{MW} \\ \mathrm{PC}, \mathrm{MW} \end{array}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | PC, MW <br> PC, MW | PC, MW <br> PC, MW | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ |  | PC,MW | PC <br> PC |
| Thompson | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | PC, MW | PC, MW | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ | $\begin{aligned} & \hline \mathrm{PC}, \mathrm{MW} \\ & \mathrm{PC}, \mathrm{MW} \end{aligned}$ |  | PC,MW | $\begin{aligned} & \hline \mathrm{PC} \\ & \mathrm{PC} \end{aligned}$ |

Table 6. Necropsy-based health assessment - descriptions, codes, and HAI ratings.

| Variable | Condition | Code | Rating |
| :---: | :---: | :---: | :---: |
| Fins | No active erosion or previous erosion healed over | 0 | 0 |
|  | Light active erosion without bleeding | 1 | 10 |
|  | Severe active erosion with hemorrhage, and/or secondary infection | 2 | 20 |
| Skin | Normal; no aberrations | 0 | 0 |
|  | One to a few small lesions (<1cm) pink or red | 1 | 10 |
|  | Many small lesions and one large (cherry-red lesion) | 2 | 20 |
|  | Multiple large and/or cherry-red lesions (blistering) | 3 | 30 |
|  | Ulcerative lesions and/or blistering | 4 | 40 |
| Eyes | No aberrations; good "clear" eye | N | 0 |
|  | Generally an opaque eye (one or both) | B1/B2 | 30 |
|  | Swollen, protruding eye (one or both) | E1/E2 | 30 |
|  | Hemorrhaging or bleeding in the eye (one or both) | H1/H2 | 30 |
|  | Missing one or both eyes | M1/M2 | 30 |
|  | Other: not fitting above categories | OT | 30 |
| Gills | Normal: no apparent aberrations | N | 0 |
|  | Frayed: erosion of tips of gill lamellae resulting in "ragged gills" | F | 30 |
|  | Clubbed: swelling of the tips of the gill lamellae | C | 30 |
|  | Marginate: gills with light margin along the tips of the lamellae | M | 30 |
|  | Pale, very light in colour | P | 30 |
|  | Other: not fitting above categories | OT | 30 |
| Opercula | Normal opercula; gills completely covered | 0 | 0 |
|  | Slight shortening; small potion of gill exposed | 1 | 10 |
|  | Severe shortening; considerable portion of the gills exposed | 2 | 20 |
| Pseudobranch | Normal; flat, containing no aberrations | N | 0 |
|  | Swollen; convex in aspect | S | 30 |
|  | Lithic: mineral deposits, white, amorphous spots | L | 30 |
|  | Swollen and lithic | SL | 30 |
|  | Inflamed; redness, hemorrhage or other | 1 | 30 |
|  | Other: not fitting above categories | OT | 30 |
| Thymus | No hemorrhage | 0 | 0 |
|  | Mild hemorrhage | 1 | 10 |
|  | Severe hemorrhage | 2 | 20 |
| Parasites | No observed parasites | 0 | 0 |
|  | Few observed parasites | 1 | 10 |
|  | Moderate parasite infestation | 2 | 20 |
|  | Numerous parasites | 3 | 30 |
| Bile | Yellow or straw colour; bladder empty or part-full | 0 |  |
|  | Yellow or straw colour; bladder distended, full | 1 |  |
|  | Light-green to "grass-green" | 2 |  |
|  | dark green to dark blue-green | 3 |  |

Table 6. Necropsy-based health assessment - descriptions, codes, and HAI ratings.(continued)

| Variable | Condition | Code | Rating |
| :--- | :--- | :---: | :---: |
| Liver | Normal: Light red colour | B | 0 |
|  | Fatty liver, coffee-with-cream colour - peamouth chub | C | 0 |
|  | Fatty liver, coffee-with-cream colour - whitefish or flounder | C | 30 |
|  | Nodules in the liver; cysts or nodules | D | 30 |
|  | Focal discolouration; distinct localized colour changes | E | 30 |
|  | general discolouration; colour change in whole liver | F | 30 |
|  | Other: not fitting above categories | OT | 30 |
| Mesenteric Fat | No fat around pyloric caecae; no fat in evidence in the visceral <br> cavity | 0 |  |
|  | Slight; less than 50\% of each caecum covered in fat - might be <br> just a trailing fat body. | 1 |  |
|  | Moderate; about 50\% of each caecum covered in fat | 2 |  |
|  | Pyloric caecae more than 50\% covered in fat | 3 |  |
|  | Very Fatty; pyloric caecae completely covered with fat | 4 |  |
|  | Normal - black | B | 0 |
|  | Normal - red | R | 0 |
|  | Normal; granular, rough appearance of spleen | G | 0 |
|  | Nodular; containing fistulas or nodules of varying sizes | D | 30 |
|  | Enlarged: noticeably enlarged | E | 30 |
|  | Other: gross abnormalities not fitting above categories | OT | 30 |
|  | Normal; no inflammation or reddening | 0 | 0 |
|  | Slight inflammation or reddening | 1 | 10 |
|  | Severe inflammation or reddening | 2 | 20 |
| Kidney | Normal; firm, dark red colour, lying relatively flat along the length <br> of the vertebral column | N | 0 |
|  | Swollen: enlarged or swollen in whole or part | S | 30 |
|  | Mottled: grey discolouration | M | 30 |
|  | Granular: granular appearance or texture | G | 30 |
|  | Urolithiasis or nephrocalcinosis; white or creamy coloured <br> mineral material in kidney tubules | U | 30 |
|  | Other: not fitting above categories | OT | 30 |

Table 7. Fish species caught in the Fraser basin 1994 to 1996.

|  | Nechako |  | Hansard |  |  |  | $\begin{gathered} \hline \text { Woodpecke } \\ r \end{gathered}$ |  |  | Marguerite |  | Agassiz |  | Mission <br> 94 | Barnston <br> 94 | $\begin{aligned} & \hline \text { Main } \\ & \text { Arm } \\ & \hline \end{aligned}$ |  |  | $\begin{aligned} & \hline \text { North } \\ & \text { Arm } \\ & \hline \end{aligned}$ |  | NorthThompson |  | Thompson |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Year | 94 | 95 | 94 | 95 | 5 | 96 | 94 | 95 | 96 | 94 | 95 | 94 | 95 |  |  | 94 | 95 | 96 | 94 | 95 | 94 | 95 | 94 | 95 |
| black crappie |  |  |  |  |  |  |  |  |  |  |  |  |  | X | X |  |  |  |  |  |  |  |  |  |
| brown bullhead |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  |  |  |  | X |  |  |  |  |  |
| burbot | X | X | X | X | X | X | X |  | X | X | X |  |  |  |  |  | X |  |  |  |  |  | X |  |
| carp |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  |  |  |  |  |  |  |  | X |  |
| chinook salmon | X | X | X | X | X | X | X | X | X | X | X | X | X | X |  | X | X | X |  | X | X | X | X |  |
| chiselmouth chub |  |  |  |  |  |  | X | X | X | X |  | X | X |  |  |  |  |  |  |  |  |  |  |  |
| coho salmon | X |  |  |  |  |  |  |  |  | X | X | X | X |  |  |  |  | X |  |  | X | X | X | X |
| Cottus sp. | X | X | X | X | X | X | X |  | X | X | X | X | X |  | X | X | X | X |  | X | X | X | X | X |
| cutthroat trout |  |  |  |  |  |  |  |  |  |  |  | X | X | X | X |  |  |  | X | X |  |  |  |  |
| Dolly Varden/bull trout |  |  | X | X | X | X | X | X | X |  |  | X | X | X |  |  |  |  |  | X | X | X |  |  |
| eelpout |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  |  |  |  |  |  |  |  |
| herring |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  |  |  |  |  |  |  |  |
| lamprey |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  |  |  |
| leopard dace | X | X | X |  |  |  | X | X |  | X | X | X | X |  |  |  |  |  |  |  |  |  | X |  |
| longnose dace | X | X | X | X | X |  | X |  |  | X | X |  | X |  |  |  |  |  |  |  |  |  | X |  |
| mountain whitefish | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X |  | X | X | X | X | X |
| pacific shad |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  |  |  |  |  |  |  |  |
| peamouth chub | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X |
| pink salmon |  |  |  |  |  |  |  |  |  |  | X |  | X |  |  |  |  |  |  |  |  | X |  | X |
| prickly sculpin |  |  |  |  |  |  |  |  |  |  |  |  |  | X | X | X | X | X |  | X |  |  |  |  |
| rainbow trout | X | X | X |  |  | X | X | X | X | X | X | X | X | X | X |  |  | X |  |  | X | X | X | X |
| redsided shiner | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X |
| sanddabs |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X | X |  |  | X |  |  |  |  |
| sand sole |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  |  |  |  |  |  |  |  |
| shiner perch |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X | X | X | X | X | X |  |  |  |  |
| smelt |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X | X | X | X | X |  |  |  |  |
| sockeye salmon | X | X |  |  |  |  | X | X | X | X | X | X | X |  | X | X |  |  |  | X |  | X | X | X |
| squawfish | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X |
| staghorn sculpin |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  | X | X |  |  |  |  |  |
| starry flounder |  |  |  |  |  |  |  |  |  |  |  |  |  | X | X | X | X | X | X | X |  |  |  |  |
| steelhead |  |  |  |  |  |  |  |  |  | X |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| suckers | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X | X |
| threespine stickleback |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X |  | X | X |  |  |  |  |  |
| white sturgeon |  |  |  |  |  |  |  |  |  |  |  |  |  |  | X | X | X |  |  | X |  |  |  |  |
| \# of species | 13 | 12 | 12 | 10 |  | 10 | 14 | 11 | 12 | 15 | 14 | 14 | 16 | 14 | 14 | 20 | 14 | 15 | 11 | 16 | 11 | 12 | 14 | 10 |

Table 8. Catch per unit effort (CPUE) for species caught in the Fraser basin in 1994.

| Location | Unit Effort | $M W$ CPUE | $\begin{gathered} P C \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \text { SU } \\ \text { CPUE } \end{gathered}$ | $\begin{aligned} & \text { SQF } \\ & \text { CPUE } \end{aligned}$ | $\begin{gathered} \text { RSS } \\ \text { CPUE } \\ \hline \end{gathered}$ | $\begin{gathered} \text { CC } \\ \text { CPUE } \end{gathered}$ | CH CPUE | $\begin{gathered} \text { RBT } \\ \text { CPUE } \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Upper Fraser Basin: <br> Beach Seines |  |  |  |  |  |  |  |  |  |
| Swanson Ck./Targe Ck. | 7 | 67.43 | 0.00 | 19.71 | 0.00 | 26.14 | 1.71 | 0.00 | 0.57 |
| Vanderhoof/Sinkut Ck./Smith Ck. | 27 | 5.52 | 19.48 | 23.89 | 67.11 | 254.00 | 0.74 | 0.85 | 0.22 |
| Nechako Average |  | 18.26 | 15.47 | 23.03 | 53.29 | 207.09 | 0.94 | 0.68 | 0.29 |
| Hansard | 11 | 1.00 | 5.09 | 4.64 | 0.45 | 9.27 | 0.45 | 5.27 | 0.09 |
| Bowron River | 3 | 8.67 | 28.00 | 4.67 | 2.67 | 28.67 | 0.00 | 22.33 | 0.00 |
| Averil Creek | 23 | 1.00 | 8.65 | 7.39 | 2.22 | 6.83 | 0.30 | 14.39 | 0.00 |
| Dome Ck./Slim Ck. | 13 | 31.85 | 15.38 | 33.38 | 0.38 | 0.08 | 0.38 | 11.62 | 0.23 |
| Ptarmigan Ck./Trophy R. | 9 | 19.00 | 4.56 | 14.22 | 0.11 | 3.11 | 0.33 | 16.89 | 0.00 |
| Hansard Average |  | 10.93 | 9.83 | 13.51 | 1.19 | 6.34 | 0.34 | 12.86 | 0.07 |
| Woodpecker/Trapping C | 52 | 1.77 | 11.79 | 20.19 | 18.60 | 34.69 | 0.06 | 7.38 | 0.19 |
| Naver Creek | 3 | 37.67 | 25.67 | 17.33 | 8.67 | 4.33 | 0.00 | 13.67 | 0.67 |
| Woodpecker Average |  | 3.73 | 12.55 | 20.04 | 18.05 | 33.04 | 0.05 | 7.73 | 0.22 |
| Hawk Creek | 8 | 1.50 | 0.38 | 18.00 | 0.00 | 1.13 | 0.00 | 14.75 | 0.50 |
| Australia Creek/Alexandria | 37 | 3.22 | 19.81 | 85.46 | 12.19 | 8.81 | 0.11 | 7.49 | 0.11 |
| Marguerite/Macalister | 19 | 1.95 | 0.47 | 23.32 | 0.16 | 4.95 | 0.05 | 3.47 | 0.00 |
| Marguerite Average |  | 2.63 | 11.64 | 58.58 | 7.09 | 6.70 | 0.08 | 7.20 | 0.13 |
| Jones Creek/Ruby Creek | 3 | 0.67 | 4.67 | 1.67 | 0.00 | 3.67 | 0.33 | 0.00 | 0.33 |
| Herling Island/Agassiz | 45 | 4.13 | 10.47 | 38.98 | 3.42 | 0.67 | 4.40 | 3.40 | 0.11 |
| Agassiz Average |  | 3.92 | 10.10 | 36.65 | 3.21 | 0.85 | 4.15 | 3.19 | 0.13 |
| McLure | 8 | 26.25 | 1.25 | 9.75 | 1.50 | 0.63 | 0.50 | 16.88 | 0.25 |
| Barriere River/Barriere | 15 | 63.67 | 8.60 | 9.80 | 0.07 | 4.73 | 5.20 | 38.80 | 1.07 |
| Louis Creek | 25 | 36.88 | 5.92 | 8.16 | 1.00 | 30.80 | 4.60 | 20.84 | 1.12 |
| North Thompson Average |  | 43.48 | 5.98 | 8.94 | 0.79 | 17.63 | 4.10 | 25.79 | 0.96 |
| Juniper /Deadman/Oxbow/Battle | 22 | 2.91 | 0.00 | 18.68 | 0.73 | 149.73 | 2.77 | 2.82 | 0.32 |
| Bonaparte River | 7 | 1.00 | 0.00 | 5.57 | 0.00 | 14.29 | 2.00 | 2.57 | 1.43 |
| Savona | 41 | 20.29 | 1.93 | 4.44 | 0.20 | 23.51 | 6.93 | 2.46 | 0.37 |
| Spences Bridge/Nicola R. | 9 | 0.11 | 0.00 | 5.22 | 0.00 | 0.56 | 0.00 | 2.67 | 0.22 |
| Kamloops Lake | 14 | 4.21 | 34.79 | 5.07 | 12.07 | 2.93 | 11.36 | 0.71 | 0.07 |
| Thompson Average |  | 10.35 | 6.09 | 8.06 | 2.08 | 47.35 | 5.57 | 2.31 | 0.38 |

Table 8. Catch per unit effort (CPUE) for species caught in the Fraser basin in 1994. (continued)

| Location | $\begin{array}{c\|} \hline \text { Unit } \\ \text { Effort } \end{array}$ | $\begin{aligned} & \hline \text { MW } \\ & \text { CPUE } \end{aligned}$ | $\begin{gathered} \hline P C \\ \text { CPUE } \end{gathered}$ | $\begin{array}{\|c\|} \hline \text { SU } \\ \text { CPUE } \end{array}$ | $\begin{aligned} & \text { SQF } \\ & \text { CPUE } \end{aligned}$ | $\begin{aligned} & \text { RSS } \\ & \text { CPUE } \end{aligned}$ | $\begin{gathered} \text { CC } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \mathrm{CH} \\ \text { CPUE } \end{gathered}$ | $\begin{aligned} & \text { RBT } \\ & \text { CPUE } \end{aligned}$ | $\begin{gathered} \hline \text { SF } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \hline \text { ST } \\ \text { CUPE } \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Lower Fraser <br> Basin: <br> Beach Seines |  |  |  |  |  |  |  |  |  |  |  |
| Crescent Island | 4 | 0.00 | 6.75 | 4.50 | 7.25 | 6.50 | 0.00 | 0.00 | 0.00 | 0.25 | 0.00 |
| Mission Bridge | 1 | 6.00 | 14.00 | 10.00 | 15.00 | 10.00 | 0.00 | 2.00 | 0.00 | 0.00 | 0.00 |
| Matsqui Island | 11 | 0.73 | 9.45 | 8.18 | 6.09 | 5.27 | 2.91 | 0.45 | 0.00 | 0.36 | 0.00 |
| Hatzic Slough | 13 | 0.00 | 3.31 | 2.92 | 2.15 | 0.38 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Westwater | 2 | 0.00 | 10.50 | 3.50 | 4.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Small Creek | 3 | 0.00 | 13.67 | 6.00 | 6.00 | 0.00 | 0.00 | 0.67 | 0.00 | 0.00 | 0.00 |
| Mission Average |  | 0.41 | 7.35 | 5.32 | 4.85 | 2.91 | 0.94 | 0.26 | 0.00 | 0.15 | 0.00 |
| Coquitlam River Mouth | 3 | 1.00 | 41.67 | 17.00 | 15.67 | 3.00 | 0.00 | 0.00 | 58.33 | 6.67 | 0.00 |
| D/S Albion Ferry | 3 | 0.00 | 15.67 | 1.67 | 4.67 | 1.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| D/S/Barnston Island | 3 | 0.00 | 19.67 | 7.00 | 7.33 | 11.00 | 0.00 | 0.00 | 0.00 | 0.33 | 1.33 |
| Haney Bar | 1 | 0.00 | 18.00 | 3.00 | 6.00 | 3.00 | 0.00 | 0.00 | 0.00 | 0.00 | 2.00 |
| Barnston Average |  | 0.30 | 24.90 | 8.00 | 8.90 | 4.90 | 0.00 | 0.00 | 17.50 | 2.10 | 0.60 |
| Annacis Channel | 3 | 0.67 | 14.67 | 4.00 | 7.33 | 4.33 | 1.00 | 0.33 | 0.00 | 2.67 | 0.00 |
| Gunderson Slough | 4 | 0.00 | 12.75 | 2.25 | 2.25 | 0.00 | 1.50 | 0.00 | 0.00 | 1.75 | 0.00 |
| Deas Slough | 2 | 0.00 | 40.00 | 9.00 | 10.50 | 2.50 | 0.00 | 0.00 | 0.00 | 45.00 | 0.00 |
| Steveston Harbour | 2 | 0.00 | 36.00 | 20.00 | 10.00 | 2.00 | 0.00 | 0.00 | 0.00 | 21.50 | 0.00 |
| Main Arm Average |  | 0.18 | 22.45 | 7.18 | 6.55 | 2.00 | 0.82 | 0.09 | 0.00 | 13.45 | 0.00 |
| Tree Island Slough | 5 | 0.00 | 14.40 | 7.40 | 8.80 | 2.80 | 2.20 | 0.00 | 0.00 | 2.20 | 0.00 |
| Near MacDonald | 4 | 0.00 | 26.75 | 0.00 | 11.25 | 0.75 | 0.00 | 0.00 | 0.00 | 10.25 | 0.00 |
| Beach <br> North Arm Average |  | 0.00 | 19.89 | 4.11 | 9.89 | 1.89 | 1.22 | 0.00 | 0.00 | 5.78 | 0.00 |

Table 8. Catch per unit effort (CPUE) for species caught in the Fraser basin in 1994. (continued)

| Location | Unit Effort | $\begin{gathered} \hline \text { MW } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \hline P C \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \hline \text { SU } \\ \text { CPUE } \end{gathered}$ | $\begin{aligned} & \hline \text { SQF } \\ & \text { CPUE } \end{aligned}$ | $\begin{aligned} & \text { RSS } \\ & \text { CPUE } \end{aligned}$ | $\begin{gathered} \hline \text { CC } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \text { CH } \\ \text { CPUE } \end{gathered}$ | $\begin{aligned} & \hline R B T \\ & \text { CPUE } \end{aligned}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Upper Fraser Basin: Boat Electroshocking |  |  |  |  |  |  |  |  |  |
| Hansard | 6 | 0.17 | 0.17 | 2.83 | 0.00 | 2.00 | 0.00 | 0.33 | 0.17 |
| Bowron River | 3 | 0.33 | 0.33 | 3.00 | 0.00 | 1.67 | 0.00 | 1.33 | 0.00 |
| Hansard Average |  | 0.22 | 0.22 | 2.89 | 0.00 | 1.89 | 0.00 | 0.67 | 0.11 |
| Alexandria | 1 | 1.00 | 0.00 | 76.00 | 18.00 | 0.00 | 0.00 | 0.00 | 4.00 |
| Marguerite Average |  | 1.00 | 0.00 | 76.00 | 18.00 | 0.00 | 0.00 | 0.00 | 4.00 |
| Gee Traps |  |  |  |  |  |  |  |  |  |
| Vanderhoof | 5 | 0.00 | 0.00 | 0.00 | 0.40 | 0.40 | 0.60 | 0.00 | 0.00 |
| Nechako Average |  | 0.00 | 0.00 | 0.00 | 0.40 | 0.40 | 0.60 | 0.00 | 0.00 |
| Hansard | 15 | 0.00 | 1.60 | 0.00 | 0.27 | 0.00 | 0.07 | 0.00 | 0.33 |
| Averil Creek | 1 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Hansard Average |  | 0.00 | 1.60 | 0.00 | 0.27 | 0.00 | 0.07 | 0.00 | 0.33 |
| Woodpecker | 12 | 0.00 | 0.08 | 0.00 | 1.83 | 0.00 | 0.00 | 0.00 | 0.00 |
| Woodpecker Average |  | 0.00 | 0.08 | 0.00 | 1.83 | 0.00 | 0.00 | 0.00 | 0.00 |
| Agassiz | 1 | 0.00 | 0.00 | 1.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Agassiz Average |  | 0.00 | 0.00 | 1.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| McLure | 2 | 0.00 | 0.50 | 0.00 | 0.00 | 0.50 | 1.50 | 0.00 | 0.50 |
| N. Thompson Average |  | 0.00 | 0.50 | 0.00 | 0.00 | 0.50 | 1.50 | 0.00 | 0.50 |
| Oxbow | 3 | 0.00 | 0.00 | 0.00 | 2.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Savona | 3 | 0.00 | 0.00 | 0.00 | 0.33 | 0.00 | 1.00 | 0.00 | 0.00 |
| Thompson Average |  | 0.00 | 0.00 | 0.00 | 1.17 | 0.00 | 0.50 | 0.00 | 0.00 |

Table 8. Catch per unit effort (CPUE) for species caught in the Fraser basin in 1994. (continued)

| Location | Unit Effort | $\begin{array}{\|c\|} \hline \text { MW } \\ \text { CPUE } \end{array}$ | $\begin{gathered} P C \\ C P U E \end{gathered}$ | $\begin{gathered} \text { SU } \\ \text { CPUE } \end{gathered}$ | $\begin{aligned} & \text { SQF } \\ & \text { CPUF } \end{aligned}$ | $\begin{aligned} & \text { RSS } \\ & \text { CPUE } \end{aligned}$ | $\left\lvert\, \begin{gathered} C C \\ \text { CPUE } \end{gathered}\right.$ | $\begin{gathered} S T \\ C P U E \end{gathered}$ | $\begin{gathered} E P \\ C P U E \end{gathered}$ | $\begin{gathered} \hline \text { SF } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \text { SP } \\ \text { CPUE } \end{gathered}$ | $\begin{array}{\|c\|} \hline \text { SM } \\ \text { CUPE } \end{array}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Lower Fraser <br> Basin: <br> Bottom Trawls |  |  |  |  |  |  |  |  |  |  |  |  |
| D/S/Barnston Island | 1 | 0.00 | 1.00 | 0.00 | 3.00 | 0.00 | 6.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Haney Bar | 2 | 1.00 | 6.50 | 1.00 | 3.00 | 0.50 | 2.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Barnston Average |  | 0.67 | 4.67 | 0.67 | 3.00 | 0.33 | 3.33 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Deas Slough | 1 | 0.00 | 20.00 | 0.00 | 0.00 | 6.00 | 2.00 | 1.00 | 0.00 | 60.00 | 12.00 | 0.00 |
| Steveston Harbour | 2 | 0.00 | 14.50 | 0.00 | 2.00 | 5.00 | 27.50 | 0.00 | 52.50 | 107.5 | 11.00 | 0.50 |
| Albion Channel | 2 | 0.00 | 5.00 | 0.00 | 0.00 | 0.00 | 17.50 | 0.00 | 35.00 | 32.50 | 5.00 | 0.00 |
| Main Arm Average |  | 0.00 | 11.80 | 0.00 | 0.80 | 3.20 | 18.40 | 0.20 | 35.00 | 68.00 | 8.80 | 0.20 |
| Near MacDonald Beach | 7 | 0.00 | 1.86 | 0.00 | 1.43 | 0.86 | 7.86 | 0.00 | 0.00 | 36.14 | 4.57 | 0.29 |
| Sea Island Bridge | 1 | 0.00 | 3.00 | 0.00 | 1.00 | 2.00 | 45.00 | 0.00 | 0.00 | 115.0 | 10.00 | 4.00 |
| North Arm Average |  | 0.00 | 2.00 | 0.00 | 1.38 | 1.00 | 12.50 | 0.00 | 0.00 | 46.00 | 5.25 | 0.75 |
| Mid-Water Trawls |  |  |  |  |  |  |  |  |  |  |  |  |
| Haney Bar | 2 | 0.00 | 4.00 | 0.00 | 6.50 | 0.00 | 1.00 | 3.50 | 0.00 | 0.00 | 0.00 | 0.00 |
| Barnston <br> Average |  | 0.00 | 4.00 | 0.00 | 6.50 | 0.00 | 1.00 | 3.50 | 0.00 | 0.00 | 0.00 | 0.00 |
| Hoop Nets |  |  |  |  |  |  |  |  |  |  |  |  |
| Tree Island Slough | 2 | 0.00 | 0.00 | 4.50 | 10.50 | 15.00 | 5.50 | 24.50 | 0.50 | 0.00 | 0.00 | 0.00 |
| North Arm Average |  | 0.00 | 0.00 | 4.50 | 10.50 | 15.00 | 5.50 | 24.50 | 0.50 | 0.00 | 0.00 | 0.00 |
| MW=Mountain Whitefish $\mathrm{CC}=$ Cottus sp ST=Sturgeon RSS=Redsided Shiner |  |  | PC=Peamouth Chub $\mathrm{CH}=$ Chinook Salmon EP=Eelpout SQF=Squawfish |  |  | SU=Suckers RBT=Rainbow Trout SP=Shiner Perch |  |  | SM=Smelt |  |  |  |
|  |  |  |  |  |  |  | $=$ Starry | Flound |  |
|  |  |  |  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |  |

Table 9. Catch per unit effort (CPUE) for species caught in the Fraser basin in 1995.

| Location | Unit Effort | $\begin{gathered} M W \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \text { PC } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \text { SU } \\ \text { CPUE } \end{gathered}$ | $\begin{aligned} & \text { SQF } \\ & \text { CPUE } \end{aligned}$ | $\begin{aligned} & \text { RSS } \\ & \text { CPUE } \end{aligned}$ | $\begin{gathered} \text { CC } \\ \text { CPUE } \end{gathered}$ | CH CPUE | $\begin{gathered} \text { RBT } \\ \text { CPUE } \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Upper Fraser Basin: Beach Seines: |  |  |  |  |  |  |  |  |  |
| Vanderhoof/Sinkut Creek | 36 | 4.72 | 11.36 | 53.64 | 76.14 | 256.31 | 2.03 | 0.72 | 0.11 |
| Swanson Creek/Targe Ck | 8 | 133.63 | 0.00 | 11.38 | 0.38 | 0.00 | 0.25 | 2.50 | 1.63 |
| Nechako Average |  | 28.16 | 9.30 | 45.95 | 62.36 | 209.70 | 1.70 | 1.05 | 0.39 |
| Dome Ck./Slim Ck./King Ck. | 45 | 22.22 | 6.58 | 13.93 | 1.09 | 0.87 | 0.07 | 30.44 | 0.00 |
| Cottonwood River | 3 | 53.33 | 6.67 | 1.67 | 0.00 | 11.67 | 0.00 | 44.67 | 0.00 |
| Unnamed Creek | 3 | 88.67 | 3.33 | 1.33 | 0.00 | 0.00 | 0.00 | 11.67 | 0.00 |
| Holmes Creek | 3 | 42.33 | 0.33 | 1.67 | 0.00 | 0.33 | 0.00 | 26.67 | 0.00 |
| Dore River/McBride | 10 | 49.20 | 0.10 | 6.50 | 0.00 | 0.00 | 0.00 | 12.80 | 0.00 |
| Hansard Average |  | 31.95 | 5.13 | 11.03 | 0.77 | 1.17 | 0.05 | 27.30 | 0.00 |
| Woodpecker | 13 | 2.54 | 29.23 | 69.46 | 35.31 | 21.38 | 0.00 | 28.46 | 0.38 |
| Naver Creek | 8 | 14.75 | 0.25 | 9.25 | 2.50 | 4.00 | 0.00 | 6.25 | 0.13 |
| Blackwater River | 3 | 17.67 | 0.00 | 6.00 | 0.00 | 0.00 | 0.00 | 1.00 | 0.00 |
| Woodpecker Average |  | 8.50 | 15.92 | 41.46 | 19.96 | 12.92 | 0.00 | 17.63 | 0.25 |
| Australia Creek | 40 | 1.00 | 7.78 | 69.55 | 27.08 | 26.43 | 0.03 | 17.03 | 0.28 |
| Marguerite | 3 | 0.00 | 1.00 | 13.67 | 6.00 | 0.00 | 0.00 | 6.67 | 0.33 |
| Macalister/Soda Ck. | 8 | 1.25 | 0.38 | 23.50 | 10.75 | 1.13 | 0.00 | 31.13 | 0.00 |
| Narcosli River | 5 | 5.40 | 9.00 | 98.00 | 47.20 | 55.20 | 0.00 | 37.20 | 1.00 |
| Marguerite Average |  | 1.38 | 6.46 | 62.52 | 25.41 | 23.96 | 0.02 | 20.29 | 0.30 |
| Agassiz | 38 | 3.47 | 26.87 | 51.47 | 8.13 | 6.82 | 0.84 | 4.45 | 0.24 |
| Agassiz Average |  | 3.47 | 26.87 | 51.47 | 8.13 | 6.82 | 0.84 | 4.45 | 0.24 |
| Raft River | 17 | 12.18 | 0.00 | 0.94 | 0.00 | 0.12 | 0.47 | 9.12 | 0.29 |
| Mann Creek | 3 | 48.00 | 0.00 | 6.67 | 0.00 | 0.00 | 0.00 | 46.67 | 0.33 |
| McLure | 37 | 8.57 | 1.68 | 18.32 | 1.16 | 15.76 | 0.08 | 15.68 | 0.54 |
| Louis Creek/Barrier River | 27 | 12.30 | 0.78 | 3.00 | 0.33 | 2.19 | 0.22 | 8.63 | 0.70 |
| North Thompson Average |  | 11.90 | 0.99 | 9.46 | 0.62 | 7.67 | 0.20 | 13.19 | 0.54 |
| Savona | 12 | 28.33 | 13.33 | 4.08 | 0.17 | 0.00 | 0.25 | 0.00 | 0.75 |
| Deadman R./Juniper Beach | 1 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Juniper Beach | 5 | 3.00 | 0.00 | 4.20 | 0.00 | 0.00 | 0.00 | 0.00 | 0.40 |
| Thompson Average |  | 19.72 | 8.89 | 3.89 | 0.11 | 0.00 | 0.17 | 0.00 | 0.61 |

Table 9. Catch per unit effort (CPUE) for species caught in the Fraser basin in 1995. (continued)


Table 10. Catch per unit effort (CPUE) for common species caught in the Fraser basin in 1996.

| Location | Unit <br> Effort | MWW <br> CPUE | PC <br> CPUE | SU <br> CPUE | SQF <br> CPUE | RSS <br> CPUE | CC <br> CPUE | CH <br> CPUE | RBT <br> CPUE |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Upper Fraser Basin |  |  |  |  |  |  |  |  |  |
| September 1996 |  |  |  |  |  |  |  |  |  |
| Dome Creek | 22 | 10.23 | 4.82 | 15.68 | 1.27 | 0.18 | 0.05 | 20.05 | 0.00 |
| Slim Creek | 9 | 13.11 | 8.56 | 18.11 | 3.78 | 0.00 | 0.00 | 37.67 | 0.00 |
| Dore River | 3 | 35.00 | 11.67 | 25.33 | 0.33 | 7.67 | 0.00 | 21.00 | 0.00 |
| Hansard Average |  | $\mathbf{1 3 . 1 8}$ | $\mathbf{6 . 4 1}$ | $\mathbf{1 7 . 1 8}$ | $\mathbf{1 . 8 5}$ | $\mathbf{0 . 7 9}$ | $\mathbf{0 . 0 3}$ | $\mathbf{2 4 . 7 9}$ | $\mathbf{0 . 0 0}$ |
|  |  |  |  |  |  |  |  |  |  |
| Woodpecker |  |  |  |  |  |  |  |  |  |
| Naver Creek | 0.06 | 8.81 | 17.31 | 14.38 | 7.25 | 0.00 | 4.00 | 0.44 |  |
| Blackwater River | 39 | 1.90 | 5.08 | 19.05 | 2.90 | 1.03 | 0.05 | 2.67 | 0.03 |
| Trapping Creek | 7 | 0.29 | 12.57 | 24.71 | 11.71 | 1.86 | 0.00 | 3.29 | 0.00 |
| Woodpecker Average | 1 | 0.00 | 12.00 | 28.00 | 13.00 | 0.00 | 0.00 | 0.00 | 2.00 |
|  |  | $\mathbf{1 . 2 2}$ | $\mathbf{6 . 9 7}$ | $\mathbf{1 9 . 3 8}$ | $\mathbf{6 . 9 5}$ | $\mathbf{2 . 6 8}$ | $\mathbf{0 . 0 3}$ | $\mathbf{3 . 0 3}$ | $\mathbf{0 . 1 6}$ |


| Location | Unit Effort | $\begin{gathered} \hline P C \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \hline \text { SF } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \hline \text { SU } \\ \text { CPUE } \end{gathered}$ | $\begin{aligned} & \text { SQF } \\ & \text { CPUE } \end{aligned}$ | $\begin{gathered} \hline S P \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \hline \text { SB } \\ \text { CPUE } \end{gathered}$ | $\begin{gathered} \hline \text { CC } \\ \text { CPUE } \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Lower Fraser Basin        <br> July 1996        |  |  |  |  |  |  |  |  |
| Westwater | 10 | 11.00 | 13.30 | 6.10 | 4.20 | 0.00 | 4.50 | 10.60 |
| Deas Island/Tilbury Slough | 2 | 8.00 | 2.00 | 1.00 | 0.00 | 0.00 | 2.50 | 0.50 |
| Don Island | 10 | 2.10 | 0.60 | 0.60 | 0.00 | 0.00 | 1.00 | 2.20 |
| Main Arm Average |  | 6.68 | 6.50 | 3.14 | 1.91 | 0.00 | 2.73 | 5.86 |
| September 1996 |  |  |  |  |  |  |  |  |
| Tilbury Is//Tilbury Mudflats | 9 | 12.78 | 5.44 | 0.78 | 1.33 | 0.00 | 0.00 | 1.33 |
| Kirkland Island | 8 | 8.38 | 10.00 | 0.00 | 0.00 | 11.00 | 0.00 | 6.88 |
| Steveston Island | 3 | 4.67 | 10.67 | 0.00 | 0.00 | 6.00 | 0.00 | 5.67 |
| Don Island | 4 | 3.75 | 1.00 | 0.50 | 0.00 | 0.00 | 0.00 | 0.00 |
| Fishing Bar | 12 | 7.50 | 0.00 | 0.33 | 0.17 | 0.00 | 0.00 | 0.00 |
| Stock Pile | 3 | 3.00 | 0.00 | 0.00 | 0.00 | 2.00 | 0.00 | 0.67 |
| Cove Site/Deas Island | 2 | 6.00 | 1.00 | 0.00 | 1.50 | 0.00 | 0.00 | 0.50 |
| Main Arm Average |  | 7.85 | 4.07 | 0.32 | 0.41 | 2.73 | 0.00 | 2.12 |
| MW=Mountain Whitefish PC=Peamouth Chub $\quad$ SU=Suckers $\quad$ CC=Co |  |  |  |  |  |  |  |  |
| CH=Chinook Salmon |  | =Rainb | Trout | SF= | tarry Flo | nder | SP=Sh | ner Perch |
| RSS=Redsided Shiner |  | Squ | fishSB | ree S | ned S | leback |  |  |

Table 11. Description of distributions of mountain whitefish sampled, i.e. diffuse versus spawning aggregations.

| Reach | Year | Creek/river mouth | Max catch | $\begin{gathered} \text { SA } \\ ? \end{gathered}$ | Comments |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | 1994 | Targe/Smith/Sinkut | 45 | Y | Mostly mature 3-17 year olds |
|  | 1995 | Sinkut | 13 | ? | Mixed immature and mature |
|  | 1995 | Targe/Swanson | 48 | Y | Mostly mature 2-12 year olds |
| Hansard | 1994 | Slim/Dome | 15 | Y | Mostly mature 3-12 year olds |
|  | 1995 | Slim/King high water | 9 | ? | Mix of mature and immature 4-12 year olds |
|  | 1995 | Dore R | 21 | Y | All mature 4-15 year olds |
|  | 1996 | Slim/Dome high water | 7 | ? | Mixed mature and immature 3-14 year olds |
|  | 1996 | Dore R | 20 | Y | Mostly mature 6-17 year olds |
| Woodpecker | 1994 | Naver | 25 | Y | All mature 2-12 year olds |
|  | 1995 | Naver/Blackwater R | 36 | Y | Mostly mature 2-11 year olds |
|  | 1996 | Naver (new creek channel) | 6 | Y | Mostly mature 2-9 year olds, but several immature 2-6 year olds |
| Marguerite | 1994 | Australia/at Macalister | 5 | N | Some female fish with small, sometimes swollen gonads and no developed eggs (with parasites or few developed eggs?) (about 3 g ) see photos; fishing crew noted that most of the 39 fish captured were spawned out females. |
|  | 1995 | Australia | 4 | N | Mostly immature 2-3 year olds, four mature 4-10 year olds |
| Agassiz | 1994 | Between bridge and Herrling Island | 3 | N | Mostly immature 2 year olds, older fish mature |
|  | 1995 | Between bridge and Herrling Island | 11 | Y | Mostly mature 2-11 year olds |
| North <br> Thompson | 1994 | Barriere River | 550 | Y | All mature 2-10 year olds |
|  | 1995 | Raft R/Mann | 16 | Y | All mature 3-13 year olds, except two |
| Thompson | 1994 | At Savona | 67 | N | Mostly immature 2-4 year olds |
|  | 1995 | At Savona | 43 | ? | Many immature 1-4 year olds; rest mature 2-10 year olds |

Table 12. Summary analytical surrogate recoveries for variables measured only in 1994.

| Dioxins | mean | SE | min | max |
| :---: | :---: | :---: | :---: | :---: |
| 13C-2,3,7,8 T4CDD | 70.2 | 1.2 | 26.0 | 100.0 |
| 13C-1,2,3,7,8 P5CDD | 76.3 | 1.7 | 20.0 | 120.0 |
| 13C-1,2,3,4,7,8 H6CDD | 67.6 | 1.5 | 17.0 | 98.0 |
| 13C-1,2,3,4,6,7,8 H7CDD | 55.4 | 1.7 | 15.5 | 94.0 |
| 13C-O8CDD | 43.7 | 1.9 | 9.6 | 100.0 |
| 13C-2,3,7,8 T4CDF | 77.0 | 1.2 | 32.0 | 110.0 |
| 13C-1,2,3,7,8 P5CDF | 74.0 | 1.5 | 24.0 | 120.0 |
| 13C-1,2,3,4,7,8 H6CDF | 72.3 | 1.5 | 20.0 | 100.0 |
| 13C-1,2,3,4,6,7,8 H7CDF | 62.8 | 1.7 | 17.0 | 100.0 |
|  |  |  |  |  |
| Chlorophenols |  |  |  |  |
| 4,5-Dichlorocatechol-13C | 51 | 2.41 | 4 | 100 |
| 3,4,5,6-Tetrachlorocatechol-13C | 47 | 1.69 | 6 | 91 |
| 4-Chlorophenol-13C | 76 | 2.63 | 17 | 169 |
| 2,4-Dichlorophenol-13C | 78 | 1.66 | 40 | 136 |
| 2,4,5-Trichlorophenol-13C | 90 | 2.23 | 33 | 151 |
| 2,4,6-Trichlorophenol-13C | 82 | 1.43 | 46 | 119 |
| 2,3,4,5-Tetrachlorophenol-13C | 99 | 2.24 | 32 | 143 |
| Pentachlorophenol-13C | 80 | 1.42 | 46 | 122 |
| 4-Chloroguaiacol-13C | 88 | 1.91 | 56 | 180 |
| 4,5,6-Trichloroguaiacol-13C | 83 | 1.34 | 35 | 116 |
| 3,4,5,6-Tetrachloroguaiacol-13C | 75 | 1.39 | 38 | 110 |
| 5-Chlorovanillin-13C | 74 | 1.68 | 25 | 115 |
|  |  |  |  |  |
| PAHs |  |  |  |  |
| Benzo(a)pyrene-d12 | 73.9 | 2.1 | 47.7 | 98.4 |
| Benzo(ghi)perylene-d12 | 66.6 | 4.9 | 32.5 | 137.6 |
| Chrysene-d12 | 79.5 | 3.0 | 50.6 | 126.1 |
| Acenaphthene-d10 | 78.7 | 1.5 | 53.4 | 91.7 |
| Dibenzo(ah)anthracene-d14 | 48.2 | 5.5 | 12.2 | 129.1 |
| Naphthalene-d8 | 73.5 | 2.3 | 7.5 | 95.9 |
| Perylene-d12 | 76.9 | 2.1 | 48.8 | 98.5 |
| Phenanthrene-d10 | 73.3 | 2.4 | 53.8 | 121.2 |
| Pyrene-d10 | 86.1 | 2.1 | 65.7 | 129.1 |

Table 13. Summary surrogate results for variables measured in both 1994 and 1995.

|  | 1994 |  |  |  |  |  | max |  |  | mean | SE | min | max |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| non-ortho PCBs | mean | SE | min | max |  |  |  |  |  |  |  |  |  |
| 13CHCB | 62.4 | 1.0 | 18.5 | 101.0 | 69.2 | 1.2 | 41.0 | 121.7 |  |  |  |  |  |
| 13-PCB101 | 76.3 | 1.1 | 58.3 | 120.0 | 82.8 | 1.3 | 52.1 | 121.0 |  |  |  |  |  |
| 13-PCB105 | 78.9 | 1.0 | 60.0 | 118.2 | 92.0 | 1.3 | 57.4 | 129.8 |  |  |  |  |  |
| 13-PCB118 | 81.0 | 1.0 | 61.0 | 125.0 | 86.6 | 1.3 | 53.6 | 126.0 |  |  |  |  |  |
| 13-PCB153 | 104.2 | 0.6 | 100.0 | 115.0 | 102.8 | 0.6 | 100. | 121.0 |  |  |  |  |  |
| 13-PCB180 | 78.2 | 1.2 | 60.5 | 126.1 | 82.2 | 1.2 | 48.4 | 126.0 |  |  |  |  |  |
| 13-PCB209 | 77.9 | 1.2 | 54.5 | 122.6 | 67.8 | 1.3 | 41.1 | 122.0 |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Pesticides | mean | SE | min | max | mean | SE | min | max |  |  |  |  |  |
| 13C12-p,p'-DDE | 80.1 | 1.0 | 52.8 | 129.6 | 92.2 | 1.2 | 55.7 | 127.1 |  |  |  |  |  |
| 13C12-p,p'-DDT | 77.6 | 1.7 | 41.6 | 151.2 | 84.8 | 1.0 | 56.5 | 118.1 |  |  |  |  |  |
| 13C12-PCB 101 | 80.8 | 0.9 | 58.3 | 120.0 | 92.9 | 1.1 | 59.2 | 127.7 |  |  |  |  |  |
| 13C12-PCB 180 | 81.6 | 1.1 | 54.4 | 120.0 | 91.6 | 1.1 | 46.8 | 125.6 |  |  |  |  |  |
| 13C12-PCB 209 | 81.7 | 1.1 | 55.3 | 122.6 | 74.1 | 1.3 | 32.3 | 122.4 |  |  |  |  |  |
| 13C6-gamma HCH | 74.5 | 1.1 | 45.5 | 131.0 | 87.8 | 1.3 | 57.0 | 125.4 |  |  |  |  |  |
| 13C6-Hexachlorobenzene | 67.5 | 1.0 | 18.0 | 101.0 | 76.0 | 1.1 | 40.7 | 121.7 |  |  |  |  |  |
| 13C8-Mirex | 77.6 | 1.1 | 48.9 | 119.2 | 81.7 | 1.0 | 47.5 | 109.8 |  |  |  |  |  |
| d4-alpha-Endosulphan | 94.7 | 1.5 | 26.0 | 162.9 | 77.2 | 1.9 | 17.2 | 134.8 |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Coplanar PCBs | mean | SE | min | max | mean | SE | min | max |  |  |  |  |  |
| 13C-PCB \#126 | 81.8 | 2.0 | 38.0 | 120.0 | 70.6 | 2.8 | 22.7 | 130.0 |  |  |  |  |  |
| 13C-PCB \#77 | 80.9 | 2.0 | 43.8 | 130.0 | 77.5 | 2.9 | 26.2 | 140.0 |  |  |  |  |  |
| 13C-PCB \#169 | 77.7 | 2.5 | 23.4 | 120.9 | 75.0 | 2.7 | 24.0 | 140.0 |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Resin Acids in Bile | mean | SE | min | max | mean | SE | min | max |  |  |  |  |  |
| O-Methylpodocarpic | 61.0 | 11.6 | 16.0 | 140.0 | 62.4 | 4.5 | 30.0 | 111.4 |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |  |  |  |  |  |

Table 14. Lab duplicate results for organochlorine pesticides and PCBs, 1994 and 1994.

| OCPesticides |  | Lab Duplicates 1994 |  | 1995 |
| :---: | :---: | :---: | :---: | :---: |
|  | N | Mean RSD (min,max) | N | Mean RSD (min,max) |
| Dieldrin | 12 | 12.4 (0.0-38.6) | 13.0 | 27.9 (0.0-53.2) |
| alpha-Endosulphan (l) | 5 | 8.7 (0.0-15.7) | 11.0 | 19.4 (0.0-44.9) |
| alpha HCH | 8 | 9.8 (3.4-21.8) | 14.0 | 34.5 (0.0-66.8) |
| beta HCH | 3 | 48.4 (9.2-114.5) | 6.0 | 62.7 (0.0-121.7) |
| delta HCH | 2 | 53.0 (19.8-86.2) | 3.0 | 51.2 (0.0-153.5) |
| gamma HCH | 4 | 29.8 (4.0-78.6) | 5.0 | 52.5 (0.0-93.8) |
| Hexachlorobenzene | 13 | 7.3 (0.0-27.4) | 15.0 | 16.2 (0.0-31.8) |
| Heptachlor Epoxide | 12 | 12.9 (0.0-47.1) | 13.0 | 28.9 (0.0-73.2) |
| Mirex | 4 | 9.0 (0.0-20.2) | 7.0 | 27.9 (0.0-118.9) |
| o, $p^{\prime}$-DDT | 4 | 8.4 (0.0-12.9) | 11.0 | 25.2 (0.0-47.3) |
| $p, p^{\prime}-D D T$ | 4 | 8.9 (0.0-26.0) | 8.0 | 25.7 (0.0-118.9) |
| $p, p^{\prime}-D D D$ | 8 | 10.6 (0.0-33.7) | 14.0 | 21.4 (0.0-37.2) |
| $p, p^{\prime}-$ DDE | 14 | 6.7 (0.0-20.7) | 15.0 | 20.3 (0.0-29.7) |
| cis-Chlordane | 8 | 16.2 (0.0-35.4) | 13.0 | 23.8 (0.0-44.9) |
| Oxychlordane | 1 | 17.7 (17.7-17.7) | 4.0 | 84.1 (0.0-224.5) |
| trans-Chlordane | 2 | 10.1 (0.0-20.2) | 9.0 | 35.9 (0.0-68.7) |
| trans-Nonachlor | 12 | 9.8 (0.0-47.1) | 13.0 | 24.4 (0.0-50.6) |
| Total Toxaphene | 11 | 17.2 (2.9-58.6) | 14.0 | 29.9 (0.0-43.4) |
| Aroclor 1242 | 2 | 31.5 (8.3-54.6) | 8.0 | 26.9 (0.0-55.0) |
| Aroclor 1254 | 7 | 12.3 (0.0-35.7) | 13.0 | 23.8 (0.0-39.0) |
| Aroclor 1260 | 8 | 15.7 (0.0-60.6) | 13.0 | 24.4 (0.0-38.0) |
| Coplanar PCBs |  |  |  |  |
|  | 1994 |  |  | 1995 |
|  | N | Mean RSD (min,max) | N | Mean RSD (min,max) |
| PCB \#77 | 10 | 11.59 (2.15-33.10) | 7 | 3.03 (0.00-7.44) |
| PCB \#126 | 9 | 12.80 (4.37-29.53) | 6 | 2.87 (0.00-15.71) |
| PCB \#169 | 6 | 14.45 (2.77-37.94) | 7 | 4.59 (0.00-15.15) |


| PCB Congeners |  | Lab Duplicates 1994 |  | 1995 |
| :---: | :---: | :---: | :---: | :---: |
|  | N | Mean RSD (min,max) | N | Mean RSD ( $\min -\mathrm{max}$ ) |
| Tot.PCBs ( $\mathrm{ND}=1 / 2$ ) | 14 | 5.1 (0.2-13.5) | 15 | $5.4(0.8-11.8)$ |
| Tot.PCBs ( $\mathrm{ND}=0$ ) | 14 | 8.9 (0.8-35.9) | 15 | 7.0 (0.3-38.1) |
| PCB22 | - |  | 4 | 24.0 (8.3-47.1) |
| PCB31/28 | 7 | 8.8 (3.3-28.3) | 11 | 9.9 (0.0-23.6) |
| PCB33 | - | - | 6 | 29.2 (0.0-106.1) |
| PCB41/71/64 | 9 | 14.4 (0.0-47.1) | 10 | 30.1 (0.0-77.8) |
| PCB42 |  | - | 4 | 12.8 (6.1-25.6) |
| PCB44 | 6 | 6.4 (0.0-13.3) | 9 | 11.5 (0.0-28.3) |
| PCB47/48 | 7 | 9.5 (0.0-23.6) | 8 | 5.1 (0.0-11.8) |
| PCB49 | 9 | 17.1 (0.0-60.6) | 9 | 11.4 (0.0-31.9) |
| PCB52 | 11 | 8.3 (0.0-38.6) | 13 | 10.2 (0.0-25.0) |
| PCB56/60 | 5 | 12.1 (6.1-22.3) | 11 | 15.1 (0.0-28.3) |
| PCB66 | 9 | 7.7 (0.0-25.4) | 13 | 10.2 (0.0-25.0) |
| PCB70/76 | 11 | 17.2 (0.0-113.1) | 13 | 7.5 (0.0-17.0) |
| PCB74 | 8 | 14.5 (2.1-28.3) | 12 | 11.9 (0.0-28.3) |
| PCB84/89 | 4 | 2.2 (0.0-8.7) | 12 | 8.9 (0.0-28.3) |
| PCB85 | 5 | 13.7 (4.0-38.6) | 12 | 8.5 (0.0-35.4) |
| PCB87 | 9 | 3.7 (0.0-13.2) | 13 | 9.1 (0.0-28.3) |
| PCB91 | - | - | 8 | 11.6 (0.0-47.1) |
| PCB95 | 8 | 4.9 (0.0-12.1) | 13 | 6.6 (0.0-15.7) |
| PCB97 | 7 | 17.0 (0.0-59.4) | 13 | 9.3 (0.0-28.3) |
| PCB99 | 10 | 9.3 (0.0-23.3) | 13 | 7.2 (0.0-28.3) |
| PCB101/90 | 12 | 11.1 (0.0-47.1) | 14 | 4.2 (0.0-13.7) |
| PCB105 | 9 | 9.5 (0.0-23.6) | 13 | 6.8 (0.0-47.1) |
| PCB107 | - | - | 9 | 9.2 (0.0-47.1) |
| PCB110 | 12 | 14.0 (0.0-34.3) | 13 | 5.9 (0.0-12.9) |
| PCB118 | 11 | 8.6 (0.0-17.0) | 14 | 4.3 (0.0-20.2) |
| PCB128 | 8 | 12.8 (2.2-33.7) | 12 | 6.4 (0.0-24.0) |
| PCB136 | - | - | 9 | 10.3 (0.0-28.3) |
| PCB138/163/164 | 14 | 8.5 (0.0-30.7) | 15 | 6.3 (0.0-20.2) |
| PCB141 | 6 | 26.3 (6.7-76.1) | 11 | 7.6 (0.0-28.3) |
| PCB144/135 | 4 | 13.4 (0.0-28.3) | 11 | 8.5 (0.0-28.3) |
| PCB146 | 7 | 6.0 (0.0-20.2) | 12 | 3.4 (0.0-15.7) |
| PCB149 | 13 | 8.2 (0.0-26.5) | 13 | 6.4 (1.7-15.7) |
| PCB151 | 8 | 13.9 (0.0-34.8) | 11 | 8.5 (0.0-28.3) |
| PCB153 | 14 | 9.9 (2.2-32.6) | 15 | 5.0 (0.0-20.2) |
| PCB156 | 4 | 12.8 (0.0-40.4) | 5 | 11.7 (0.0-28.3) |
| PCB158 | 4 | 9.7 (0.0-21.8) | 10 | 14.4 (0.0-28.3) |
| PCB170/190 | 6 | 8.3 (0.0-18.1) | 12 | 9.8 (0.0-25.0) |
| PCB174 | 4 | 25.6 (13.9-41.6) | 9 | 9.7 (0.0-28.3) |
| PCB177 | 5 | 16.5 (6.7-25.7) | 9 | 5.1 (0.0-15.7) |
| PCB179 | - | - | 7 | 8.4 (0.0-28.3) |
| PCB180 | 14 | 9.5 (0.0-26.4) | 13 | 6.8 (2.4-20.2) |
| PCB183 | 5 | 22.5 (0.0-55.3) | 12 | 11.2 (0.0-40.4) |
| PCB187/182 | 11 | 10.2 (0.0-42.2) | 13 | 4.0 (0.0-12.9) |
| PCB193 | 5 | 21.5 (0.0-50.5) | - | - |
| PCB196/203 | 3 | 9.9 (0.0-20.2) | 7 | 15.2 (0.0-38.6) |
| PCB199 | 4 | 26.2 (0.0-64.3) | 7 | 9.8 (0.0-26.9) |

Table 15. Lab duplicate results for chlorophenolic and dioxin and furan analyses, 1994.

| Chlorophenolics |  | $\begin{gathered} \mathrm{n}=12 \\ \text { Duplicates } \end{gathered}$ |
| :---: | :---: | :---: |
|  | N | Mean RSD (min,max) |
| 4-Chlorophenol | 1 | 91.5 (91.5-91.5) |
| 24/25-Dichlorophenol | 5 | 6.7 (0.0-17.0) |
| 26-Dichlorophenol | 1 | 25.7 (25.7-25.7) |
| 34-Dichlorophenol | 2 | 68.4 (43.3-93.6) |
| 35-Dichlorophenol | 6 | 26.0 (1.0-78.1) |
| 234-Trichlorophenol | 1 | 121.9 (121.9-121.9) |
| 235-Trichlorophenol | 1 | 107.3 (107.3-107.3) |
| 236-Trichlorophenol | 1 | 91.9 (91.9-91.9) |
| 245-Trichlorophenol | 1 | 104.7 (104.7-104.7) |
| 246-Trichlorophenol | 3 | 14.4 (2.2-37.2) |
| 345-Trichlorophenol | 2 | 66.8 (36.6-97.0) |
| 2345-Tetrachlorophenol | 1 | 130.5 (130.5-130.5) |
| 2346-Tetrachlorophenol | 1 | 77.2 (77.2-77.2) |
| 2356-Tetrachlorophenol | 1 | 118.0 (118.0-118.0) |
| Pentachlorophenol | 2 | 24.2 (3.0-45.4) |
| 3-Chlorocatechol | 4 | 34.0 (0.0-69.9) |
| 4-Chlorocatechol | 1 | 85.4 (85.4-85.4) |
| 34-Dichlorocatechol | 2 | 79.1 (72.0-86.2) |
| 35-Dichlorocatechol | 1 | 80.8 (80.8-80.8) |
| 36-Dichlorocatechol | 1 | 103.5 (103.5-103.5) |
| 45-Dichlorocatechol | 1 | 76.1 (76.1-76.1) |
| 3456-Tetrachlorocatechol | 2 | 62.2 (42.4-81.9) |
| 4-Chloroguaiacol | 1 | 86.5 (86.5-86.5) |
| 5-Chloroguaiacol | 6 | 31.5 (2.2-87.3) |
| 6-Chloroguaiacol | 1 | 130.9 (130.9-130.9) |
| 34-Dichloroguaiacol | 1 | 85.8 (85.8-85.8) |
| 45-Dichloroguaiacol | 2 | 26.2 (5.2-47.1) |
| 46-Dichloroguaiacol | 1 | 104.9 (104.9-104.9) |
| 345-Trichloroguaiacol | 1 | 8.8 (8.8-8.8) |
| 346-Trichloroguaiacol | 1 | 97.5 (97.5-97.5) |
| 456-Trichloroguaiacol | 1 | 79.3 (79.3-79.3) |
| 3456-Tetrachloroguaiacol | 2 | 24.0 (4.3-43.7) |
| 3-Chlorosyringol | 2 | 62.5 (25.3-99.7) |
| 35-Dichlorosvringol | 1 | 112.8 (112.8-112.8) |
| 345-Trichlorosyringol | 1 | 108.6 (108.6-108.6) |
| 2-Chlorosyringaldehyde | 1 | 41.6 (41.6-41.6) |
| 26-Dichlorosyringaldehyde | 3 | 53.4 (19.5-118.8) |
| 345-Trichloroveratrole | 2 | 45.8 (35.4-56.3) |
| 3456-Tetrachloroveratrole | 3 | 53.2(8.3-112.0) |


| Dioxins/Furans |  |  |
| :--- | :---: | :---: |
|  | N | Mean RSD (min,max) |
| 2,3,7,8 TCDD | 7 | $2.3(0.0-10.9)$ |
| 1,2,3,6,7,8 H6CDD | 1 | $0.0(0.0-0.0)$ |
| 1,2,3,4,6,7,8 H7CDD | 1 | $5.7(5.7-5.7)$ |
| T4CDD (TOTAL) | 7 | $8.1(0.0-18.4)$ |
| H6CDD (TOTAL) | 1 | $14.1(14.1-14.1)$ |
| H7CDD (TOTAL) | 1 | $12.9(12.9-12.9)$ |
| O8CDD (TOTAL) | 2 | $40.5(22.7-58.2)$ |
|  |  |  |
| 2,3,7,8 T4CDF | 10 | $8.1(0.0-47.1)$ |
| 2,3,4,7,8 P5CDF | 1 | $0.0(0.0-0.0)$ |
| T4CDF (TOTAL) | 10 | $7.3(0.0-47.1)$ |
| P5CDF (TOTAL) | 2 | $23.6(0.0-47.1)$ |
| H6CDF (TOTAL) | 1 | $10.9(10.9-10.9)$ |
| H7CDF(TOTAL) |  |  |

Table 16. Blind duplicate results for trace metal measurements in fish tissues from the Fraser River.

|  | 1994 |  | 1995 |  |
| :---: | :---: | :---: | :---: | :---: |
|  | $\mathbf{N}$ | Mean RSD (Min/Max) | $\mathbf{N}$ | Mean RSD (Min/Max) |
| As-Total | 2 | $3.23(2.79-3.67)$ | 9 | $2.85(0.00-7.93)$ |
| Cd-Total | 1 | $3.97(3.97-3.97)$ | 5 | 4.26 (1.55-12.39) |
| Co-Total | 0 | No Detects | 0 | No Detects |
| Cr-Total | 1 | $1.61(1.61-1.61)$ | 0 | No Detects |
| Cu-Total | 11 | $5.57(1.54-14.59)$ | 19 | $2.96(0.00-9.02)$ |
| Fe-Total | 11 | $6.85(2.19-16.26)$ | 19 | $7.07(2.57-17.54)$ |
| Hg-Total | 11 | $5.13(2.29-11.25)$ | 16 | $7.07(2.16-15.84)$ |
| Mn-Total | 9 | $2.63(1.46-4.45)$ | 8 | $3.21(1.63-4.45)$ |
| Ni-Total | 10 | $3.90(1.20-18.63)$ | 5 | $1.60(1.09-2.22)$ |
| Pb-Total | 0 | No Detects | 0 | No Detects |
| Se-Total | 9 | $5.27(3.29-7.57)$ | 16 | $3.98(0.00-7.38)$ |
| Zn-Total | 11 | $5.14(2.86-8.55)$ | 19 | $8.31(2.50-33.67)$ |

Table 17. Comparison of dioxin/furan analyses conducted by the Regional Dioxin Lab at the Institute of Ocean Sciences, Sidney BC and Axys Analytical Services, Ltd.

| Reach | Barnston |  | Main Arm |  | North Arm |  | Mission |  | Barnston |  |  | Barnston |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Axys Lab ID | 2878-44 |  | 2878i-39 i |  | 2878-35 |  | 2878-46 R |  | 2878-42 |  |  | 2878-45 |  |  |  |
| Sample ID | IOS 1 | AXYS | IOS 2 | AXYS | IOS 3 | AXYS | IOS 4 | AXYS | IOS 5 (1) | IOS 5(2) | AXYS | IOS 6 (1) | IOS 6 (2) | AXYS | AXYS |
| Percent Lipid | <2 | 2.4 | <2 | 1.4 | <2 | 1.9 | <2 | 2.5 | <2 | <2 | 1.5 | <2 | <2 | 1.8 | 1.8 |
| Percent Moisture | 77.9 |  | 79.8 |  | 79.2 |  | 77.2 |  | 79.5 | 79.5 |  | 77.5 | 77.5 |  |  |
| 2,3,7,8 TCDD | 0.08 | $<0.10$ | 0.07 | < 0.08 | 0.08 | 0.10 | 0.10 | 0.10 | 0.10 | 0.08 | < 0.12 | 0.08 | 0.06 | 0.10 | 0.10 |
| 1,2,3,7,8 PCDD | $<0.08$ | $<0.09$ | < 0.08 | $<0.17$ | $<0.08$ | $<0.10$ | $<0.08$ | < 0.20 | 0.08 | 0.09 | $<0.11$ | $<0.08$ | $<0.08$ | $<0.10$ | $<0.10$ |
| 1,2,3,4,7,8 H6CDD | $<0.10$ | $<0.20$ | < 0.10 | $<0.17$ | $<0.10$ | $<0.20$ | $<0.10$ | < 0.20 | $<0.10$ | $<0.10$ | $<0.16$ | $<0.10$ | $<0.10$ | $<0.10$ | $<0.20$ |
| 1,2,3,6,7,8 H6CDD | 0.21 | < 0.20 | 0.22 | $<0.17$ | 0.32 | < 0.20 | 0.22 | < 0.20 | 0.21 | 0.24 | $<0.16$ | 0.21 | 0.25 | $<0.10$ | < 0.20 |
| 1,2,3,7,8,9 H6CDD | $<0.10$ | $<0.20$ | < 0.10 | $<0.17$ | $<0.10$ | < 0.20 | $<0.10$ | < 0.20 | $<0.10$ | $<0.10$ | < 0.16 | $<0.10$ | $<0.10$ | $<0.10$ | $<0.20$ |
| 1,2,3,4,6,7,8 H7CDD | 0.12 | $<0.20$ | < 0.12 | $<0.50$ | 0.23 | 0.53 | 0.20 | < 0.30 | 0.12 | 0.15 | $<0.15$ | $<0.12$ | 0.13 | $<0.20$ | $<0.20$ |
| O8CDD (TOTAL) | 0.50 | $<0.30$ | 0.38 | 1.60 | 1.08 | 3.30 | 0.56 | $<0.40$ | 0.54 | 0.52 | $<0.40$ | 0.43 | 0.61 | $<0.40$ | $<0.30$ |
| 2,3,7,8 T4CDF | 0.54 | 0.40 | 0.43 | 0.30 | 0.61 | 0.65 | 0.71 | 0.69 | 0.70 | 0.64 | 0.40 | 0.75 | 0.91 | 0.73 | 0.72 |
| 1,2,3,7,8 P5CDF | $<0.06$ | $<0.20$ | < 0.06 | < 0.20 | $<0.06$ | $<0.20$ | $<0.06$ | $<0.10$ | $<0.06$ | $<0.06$ | $<0.20$ | $<0.06$ | $<0.06$ | $<0.20$ | $<0.10$ |
| 2,3,4,7,8 P5CDF | $<0.06$ | $<0.20$ | < 0.06 | < 0.20 | $<0.06$ | $<0.20$ | $<0.06$ | < 0.10 | 0.07 | 0.07 | < 0.20 | 0.07 | 0.08 | $<0.20$ | $<0.10$ |
| 1,2,3,4,7,8 H6CDF | $<0.08$ | < 0.20 | < 0.08 | $<0.15$ | $<0.08$ | $<0.30$ | $<0.08$ | < 0.30 | < 0.08 | $<0.08$ | < 0.20 | $<0.08$ | $<0.08$ | $<0.20$ | < 0.20 |
| 1,2,3,6,7,8 H6CDF | $<0.08$ | < 0.20 | < 0.08 | $<0.15$ | $<0.08$ | < 0.30 | $<0.08$ | < 0.30 | $<0.08$ | $<0.08$ | < 0.20 | $<0.08$ | $<0.08$ | $<0.20$ | < 0.20 |
| 1,2,3,7,8,9 H6CDF | $<0.08$ | < 0.20 | < 0.08 | $<0.15$ | < 0.08 | $<0.30$ | $<0.08$ | < 0.30 | $<0.08$ | < 0.08 | < 0.20 | $<0.08$ | $<0.08$ | $<0.20$ | $<0.20$ |
| 2,3,4,6,7,8 H6CDF | $<0.08$ | < 0.20 | < 0.08 | $<0.15$ | $<0.08$ | $<0.30$ | $<0.08$ | < 0.30 | $<0.08$ | $<0.08$ | < 0.20 | $<0.08$ | $<0.08$ | $<0.20$ | $<0.20$ |
| 1,2,3,4,6,7,8 H7CDF | $<0.10$ | $<0.30$ | < 0.10 | $<0.27$ | $<0.10$ | < 0.30 | $<0.10$ | < 0.30 | $<0.10$ | $<0.10$ | < 0.20 | $<0.10$ | $<0.10$ | $<0.20$ | $<0.20$ |
| 1,2,3,4,7,8,9 H7CDF | $<0.10$ | $<0.30$ | < 0.10 | $<0.27$ | $<0.10$ | $<0.30$ | $<0.10$ | < 0.30 | $<0.10$ | $<0.10$ | < 0.20 | $<0.10$ | $<0.10$ | $<0.20$ | < 0.20 |
| O8CDF (TOTAL) | $<0.12$ | $<0.30$ | < 0.12 | $<0.73$ | $<0.12$ | $<0.30$ | $<0.12$ | < 0.10 | $<0.12$ | $<0.12$ | < 0.20 | $<0.12$ | $<0.12$ | $<0.20$ | <0.20 |
| T4CDD (TOTAL) | 0.08 | $<0.10$ | 0.07 | $<0.08$ | $<0.06$ | 0.20 | $<0.06$ | 0.20 | 0.10 | 0.08 | $<0.12$ | 0.16 | 0.09 | 0.10 | 0.10 |
| P5CDD (TOTAL) | $<0.08$ | $<0.09$ | < 0.08 | $<0.17$ | $<0.08$ | $<0.10$ | $<0.08$ | < 0.20 | 0.08 | $<0.08$ | $<0.11$ | $<0.08$ | $<0.08$ | $<0.10$ | $<0.10$ |
| H6CDD (TOTAL) | 0.35 | $<0.20$ | 0.22 | $<0.17$ | 0.29 | $<0.20$ | 0.22 | < 0.20 | 0.21 | $<0.10$ | $<0.16$ | $<0.10$ | $<0.10$ | $<0.10$ | $<0.20$ |
| H7CDD (TOTAL) | 0.28 | $<0.20$ | 0.12 | $<0.50$ | 0.25 | 0.53 | 0.20 | < 0.30 | 0.12 | 0.15 | $<0.15$ | 0.12 | 0.34 | $<0.20$ | $<0.20$ |
| O8CDD (TOTAL) | 0.50 | $<0.30$ | 0.38 | 1.60 | 1.08 | 3.30 | 0.56 | < 0.40 | 0.54 | 0.52 | $<0.40$ | 0.43 | 0.61 | < 0.40 | < 0.30 |
| T4CDF (TOTAL) | 0.54 | 0.40 | 0.43 | 0.30 | 0.61 | 0.65 | 0.71 | 0.69 | 0.70 | 0.64 | 0.40 | 0.75 | 0.91 | 0.73 | 0.72 |
| P5CDF (TOTAL) | $<0.06$ | $<0.20$ | < 0.06 | < 0.20 | 0.36 | 0.46 | 0.13 | 0.20 | $<0.06$ | $<0.06$ | $<0.20$ | 0.08 | 0.15 | $<0.20$ | $<0.10$ |
| H6CDF (TOTAL) | $<0.08$ | $<0.20$ | < 0.08 | $<0.15$ | 0.26 | $<0.30$ | 0.23 | $<0.30$ | $<0.08$ | $<0.08$ | < 0.20 | 0.09 | $<0.08$ | $<0.20$ | $<0.20$ |
| H7CDF (TOTAL) | $<0.10$ | $<0.30$ | < 0.10 | $<0.27$ | $<0.10$ | $<0.30$ | 0.35 | $<0.30$ | $<0.10$ | $<0.10$ | < 0.20 | $<0.10$ | $<0.10$ | $<0.20$ | $<0.20$ |
| O8CDF (TOTAL) | < 0.12 | $<0.30$ | < 0.12 | $<0.73$ | $<0.12$ | < 0.30 | $<0.12$ | < 0.10 | $<0.12$ | < 0.12 | < 0.20 | $<0.12$ | $<0.12$ | < 0.20 | < 0.20 |

Table 18. Chlorodioxin and chlorofurans measured in fish tissues from the Fraser basin, 1994.

| Dioxins | Furans |
| :--- | :--- |
| 2,3,7,8-Tetrachlorodibenzodioxin | 2,3,7,8-Tetrachlorodibenzofuran |
| Total Tetrachlorodibenzodioxin | Total Tetrachlorodibenzofuran |
| 1,2,3,7,8-Pentachlorodibenzodioxin | $1,2,3,7,8$-Pentachlorodibenzofuran |
| Total Pentachlorodioxin | $2,3,4,7,8$-Pentachlorodibenzofuran |
| 1,2,3,4,7,8-Hexachlorodibenzodioxin | Total Pentachlorodibenzofuran |
| 1,2,3,6,7,8-Hexachlorodibenzodioxin | $1,2,3,4,7,8$-Hexachlorodibenzofuran |
| 1,2,3,7,8,9-Hexachlorodibenzodioxin | $1,2,3,6,7,8$-Hexachlorodibenzofuran |
| Total Hexachlorodibenzodioxin | $2,3,4,6,7,8$-Hexachlorodibenzofuran |
| 1,2,3,4,6,7,8-Heptachlorodibenzodioxin | 1,2,3,7,8,9-Hexachlorodibenzofuran |
| Total Heptachlorodibenzodioxin | Total Hexachlorodibenzofuran |
| Octachlorodibenzodioxin | 1,2,3,4,6,7,8-Heptachlorodibenzofuran |
|  | 1,2,3,4,7,8,9-Heptachlorodibenzofuran |
|  | Total Heptachlorodibenzofuran |
|  |  |

Table 19. Chlorinated dibenzodioxin and dibenzofuran TEQs and homolog totals in peamouth chub muscle from the Fraser basin, 1994. TEQs are from van den Berg et al. (1998), with NATO TEQs shown for comparison. Non-detects were set equal to zero. Values shown are means and standard error of $4-5$ tissue composites of 5 individuals each, in $\mathrm{pg} / \mathrm{g}$ wet weight. Values below the mean indicate the range of measured values.


Table 20. Chlorinated dibenzodioxin and dibenzofuran TEQs and homolog totals in flounder and whitefish muscle from the Fraser basin, 1994. TEQs are from van den Berg et al. (1998), with NATO TEQs shown for comparison. Non-detects were set equal to zero. Values shown are means and standard error of 4-5 tissue composites of 5 individuals each, in $\mathrm{pg} / \mathrm{g}$ wet weight. Values below the mean indicate the range of measured values.


Table 21. Chlorinated dibenzodioxin and dibenzofuran TEQs and homolog totals in peamouth chub liver from the Fraser basin, 1994. TEQs are from van den Berg et al. (1998), with NATO TEQs shown for comparison. Non-detects were set equal to zero. Values below the mean indicate the range of measured values.

|  | 2,3,7,8-TEQ | Peamouth Chub Liver Tissue |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Hansard | Woodpecker | Marguerite | Mission | Barnston | Main Arm | North Arm | N. <br> Thompson | Thompson |
|  | Mammals <br> Fish <br> Birds | $\begin{aligned} & \hline 0.16 \\ & 0.08 \\ & 1.50 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.17 \\ & 0.09 \\ & 1.70 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.12 \\ & 0.06 \\ & 1.20 \\ & \hline \end{aligned}$ | $\begin{gathered} 3.16 \\ 2.75 \\ 11.07 \\ \hline \end{gathered}$ | $\begin{gathered} \hline 3.13 \\ 2.69 \\ 11.49 \\ \hline \end{gathered}$ | $\begin{gathered} \hline 3.28 \\ 2.76 \\ 11.83 \\ \hline \end{gathered}$ | $\begin{gathered} 3.52 \\ 2.89 \\ 11.94 \\ \hline \end{gathered}$ | $\begin{gathered} 2.05 \\ 1.5 \\ 11.95 \\ \hline \end{gathered}$ | $\begin{gathered} \hline 5.7 \\ 3.83 \\ 39.45 \\ \hline \end{gathered}$ |
|  | NATO | 0.16 | 0.17 | 0.12 | 2.70 | 2.64 | 2.82 | 2.98 | 1.95 | 5.70 |
|  | Detect Lims |  |  |  |  |  |  |  |  |  |
| T4CDD | $\begin{array}{r} 0.60(0.30-1.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{gathered} \begin{array}{c} \text { ND } \\ (<0.50) \end{array} \end{gathered}$ | $\begin{gathered} \begin{array}{c} \text { ND } \\ (<0.50) \end{array} \end{gathered}$ | $0.40$ | $1.80$ | $\begin{gathered} 1.80 \\ - \end{gathered}$ | $\begin{gathered} 2.35 \\ (1.70-3.00) \\ \hline \end{gathered}$ | $\begin{gathered} 2.10 \\ (2.00-2.20) \\ \hline \end{gathered}$ | $\begin{gathered} 1.25 \\ (1.10-1.40) \\ \hline \end{gathered}$ | $\begin{gathered} 1.95 \\ (1.30-2.60) \\ \hline \end{gathered}$ |
| P5CDD | $\begin{array}{r} 0.60(0.30-2.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{gathered} \mathrm{ND} \\ (<0.50) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.40) \end{gathered}$ | $0.90$ | $0.98$ | $\begin{gathered} 0.99 \\ (0.98-1.00) \\ \hline \end{gathered}$ | $\begin{gathered} 1.10 \\ (1.00-1.20) \\ \hline \end{gathered}$ | $\begin{gathered} 0.28 \\ (<0.30-0.40) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.80) \end{gathered}$ |
| H6CDD | $\begin{array}{r} 1.00(0.24-3.00) \\ \text { range } \end{array}$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.50) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.80) \end{gathered}$ | $3.20$ | $2.00$ | $\begin{gathered} 2.06 \\ (<0.24-4.00) \\ \hline \end{gathered}$ | $\begin{gathered} 3.10 \\ (2.80-3.40) \\ \hline \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.60-<0.60) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ |
| H7CDD | $\begin{array}{r} 2.00(0.80-4.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{array}{c\|} \hline 2.45 \\ (<4.00-2.90) \\ \hline \end{array}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.20) \\ \hline \end{gathered}$ | $7.10$ | $3.30$ | $\begin{gathered} \hline 3.80 \\ (2.90-4.70) \\ \hline \end{gathered}$ | $\begin{gathered} 6.40 \\ (5.40-7.40) \\ \hline \end{gathered}$ | $\begin{gathered} \hline \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.50) \end{gathered}$ |
| O8CDD | $\begin{array}{r} 3.00(1.00-6.00) \\ \text { range } \end{array}$ | $\begin{gathered} \hline \text { ND } \\ (<3.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.50) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $3.50$ | $1.60$ | $\begin{gathered} 4.05 \\ (2.40-5.70) \\ \hline \end{gathered}$ | $\begin{gathered} 3.30 \\ (2.80-3.80) \\ \hline \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.40) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<4.00) \end{gathered}$ |
| T4CDF | $\begin{array}{r} 0.60(0.60-0.60) \\ \text { range } \\ \hline \end{array}$ | $\begin{array}{c\|} \hline 1.50 \\ (1.10-1.90) \\ \hline \end{array}$ | $1.70$ | $1.20$ | $8.90$ | $9.40$ | $\begin{gathered} 9.50 \\ (9.00-10.00) \\ \hline \end{gathered}$ | $\begin{gathered} 9.45 \\ (9.30-9.60) \\ \hline \end{gathered}$ | $\begin{gathered} \hline 11.00 \\ (11.00-11.00) \\ \hline \end{gathered}$ | $\begin{gathered} \hline 37.50 \\ (32.00-43.00) \\ \hline \end{gathered}$ |
| P5CDF | $\begin{array}{r} 0.60(0.30-2.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{gathered} \begin{array}{c} \mathrm{ND} \\ (<0.50) \end{array} \end{gathered}$ | $\begin{gathered} \hline \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \begin{array}{c} \text { ND } \\ (<0.40) \end{array} \end{gathered}$ | $3.20$ | $2.70$ | $\begin{gathered} 3.00 \\ (2.20-3.80) \\ \hline \end{gathered}$ | $\begin{gathered} 4.25 \\ (3.90-4.60) \\ \hline \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.30) \end{gathered}$ | $\begin{gathered} \hline N D \\ (<0.80) \end{gathered}$ |
| H6CDF | $\begin{array}{r} 1.00(0.40-3.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{gathered} \hline \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \hline \mathrm{ND} \\ (<1.50) \end{gathered}$ | $\begin{gathered} \hline \mathrm{ND} \\ (<0.80) \end{gathered}$ | $3.40$ | $2.00$ | $\begin{gathered} 2.20 \\ (2.00-2.40) \\ \hline \end{gathered}$ | $\begin{gathered} 4.45 \\ (4.20-4.70) \\ \hline \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.60) \end{gathered}$ | $\begin{gathered} \hline \mathrm{ND} \\ (<1.00) \end{gathered}$ |
| H7CDF | $\begin{array}{r} 1.20(0.19-4.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.20) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.70) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.50) \end{gathered}$ | $\begin{gathered} 0.48 \\ (<0.79-0.57) \\ \hline \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.60) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.50) \end{gathered}$ |
| O8CDF | $\begin{array}{r} 2.00(0.49-6.00) \\ \text { range } \end{array}$ | $\begin{gathered} \mathrm{ND} \\ (<3.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.50) \end{gathered}$ | $\begin{gathered} \text { ND } \\ (-2.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.70) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.50) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.60) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.64) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.40) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<4.00) \end{gathered}$ |

Table 22. Chlorinated dibenzodioxin and dibenzofuran TEQs and homolog totals in starry flounder and whitefish liver from the Fraser basin, 1994. TEQs are from van den Berg et al. (1998), with NATO TEQs shown for comparison. Non-detects were set equal to zero. Values below the mean indicate the range of measured values.

| 2,3,7,8-TEQ |  | Starry Flounder |  | Mountain Whitefish Liver Tissue |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Main Arm | North Arm | Hansard | Woodpecker | Marguerite | N. Thompson | Thompson |
|  | Mammals <br> Fish <br> Birds | $\begin{aligned} & \hline 1.07 \\ & 0.88 \\ & 3.51 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 1.28 \\ & 1.05 \\ & 3.75 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.00 \\ & 0.00 \\ & 0.00 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 2.61 \\ & 2.48 \\ & 3.91 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 1.73 \\ & 1.62 \\ & 3.80 \\ & \hline \end{aligned}$ | $\begin{aligned} & 1.04 \\ & 0.72 \\ & 6.75 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.56 \\ & 0.28 \\ & 5.55 \\ & \hline \end{aligned}$ |
|  | NATO | 1.03 | 1.11 | 0.00 | 2.55 | 1.73 | 1.04 | 0.56 |
|  | Detect Lims |  |  |  |  |  |  |  |
| T4CDD | $\begin{array}{r} \hline 0.60(0.30-1.50) \\ \text { range } \\ \hline \end{array}$ | $0.80$ | $0.72$ | $\begin{gathered} \hline \text { ND } \\ (<0.60) \end{gathered}$ | $\begin{gathered} 2.40 \\ (1.90-2.90) \end{gathered}$ | 1.50 | $\begin{gathered} 0.55 \\ (<0.6-0.80) \\ \hline \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.30) \end{gathered}$ |
| P5CDD | $\begin{gathered} 0.60(0.30-2.00) \\ \text { range } \end{gathered}$ | $0.30$ | $0.30$ | $\begin{gathered} \mathrm{ND} \\ (<0.60) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.30) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<1.00) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.60) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (0.40) \\ \hline \end{gathered}$ |
| H6CDD | $\begin{array}{r} 1.00(0.24-3.00) \\ \text { range } \\ \hline \end{array}$ | $0.58$ | $1.00$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} 0.80 \\ (<1.1-1.1) \\ \hline \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.60) \end{gathered}$ |
| H7CDD | $\begin{gathered} 2.00(0.80-4.00) \\ \text { range } \end{gathered}$ | $0.90$ | $1.10$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \\ \hline \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<1.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<3.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.80) \end{gathered}$ |
| O8CDD | $\begin{array}{r} \hline 3.00(1.00-6.00) \\ \text { range } \end{array}$ | $1.70$ | $1.50$ | $\begin{gathered} \mathrm{ND} \\ (<3.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.50) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<4.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<3.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ |
| T4CDF | $\begin{gathered} 0.60(0.60-0.60) \\ \text { range } \end{gathered}$ | $2.70$ | $2.60$ | $\begin{gathered} \mathrm{ND} \\ (<0.60) \\ \hline \end{gathered}$ | $1.50$ | $2.30$ | $\begin{gathered} 6.35 \\ (5.40-7.30) \\ \hline \end{gathered}$ | $\begin{gathered} 5.55 \\ (3.10-8.00) \\ \hline \end{gathered}$ |
| P5CDF | $\begin{array}{r} 0.60(0.30-2.00) \\ \text { range } \\ \hline \end{array}$ | $1.10$ | $1.10$ | $\begin{gathered} \begin{array}{c} \text { ND } \\ (<0.60) \end{array} \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.30) \end{gathered}$ | $\begin{gathered} \hline \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.60) \end{gathered}$ | $\begin{gathered} \hline \text { ND } \\ (<0.40) \end{gathered}$ |
| H6CDF | $\begin{array}{r} 1.00(0.40-3.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{gathered} \hline \text { ND } \\ (<0.40) \end{gathered}$ | $2.90$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.50) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<0.60) \end{gathered}$ |
| H7CDF | $\begin{array}{r} 1.20(0.19-4.00) \\ \text { range } \\ \hline \end{array}$ | $\begin{gathered} \mathrm{ND} \\ (<0.19) \end{gathered}$ | $0.25$ | $\begin{gathered} \begin{array}{c} \text { ND } \\ (<2.00) \end{array} \end{gathered}$ | $\begin{gathered} \begin{array}{c} N D \\ (<1.00) \end{array} \end{gathered}$ | $\begin{gathered} \begin{array}{c} N D \\ (<3.00) \end{array} \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<2.00) \end{gathered}$ | $\begin{gathered} \begin{array}{c} \text { ND } \\ (<0.80) \end{array} \end{gathered}$ |
| O8CDF | $\begin{array}{r} 2.00(0.49-6.00) \\ \text { range } \end{array}$ | $1.70$ | $0.25$ | $\begin{gathered} \mathrm{ND} \\ (<3.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<1.50) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<4.00) \end{gathered}$ | $\begin{gathered} \mathrm{ND} \\ (<3.00) \end{gathered}$ | $\begin{gathered} \begin{array}{c} \mathrm{ND} \\ (<1.00) \end{array} \end{gathered}$ |

Table 23. Detection frequencies (\%) of measured dioxin and furan congeners in fish tissues from the Fraser basin, 1994.

|  | Liver Analyses |  |  |
| :---: | :---: | :---: | :---: |
|  | Peamouth | Whitefish | Flounder |
| Number of Analyses | 14 | 8 | 2 |
| 2,3,7,8 TCDD | 71.4 | 50.0 | 100.0 |
| 1,2,3,7,8 PCDD | 37.5 | ND | 100.0 |
| 1,2,3,4,7,8 H6CDD | 12.5 | ND | ND |
| 1,2,3,6,7,8 H6CDD | 37.5 | 12.5 | 100.0 |
| 1,2,3,7,8,9 H6CDD | 12.5 | ND | ND |
| 1,2,3,4,6,7,8 H7CDD | 37.5 | ND | 100.0 |
| T4CDD (TOTAL) | 70.8 | 50.0 | 100.0 |
| P5CDD (TOTAL) | 37.5 | ND | 100.0 |
| H6CDD (TOTAL) | 33.3 | 12.5 | 100.0 |
| H7CDD (TOTAL) | 37.5 | ND | 100.0 |
| O8CDD (TOTAL) | 33.3 | ND | 100.0 |
| 2,3,7,8 T4CDF | 95.8 | 87.5 | 100.0 |
| 1,2,3,7,8 P5CDF | 8.3 | ND | ND |
| 2,3,4,7,8 P5CDF | 33.3 | ND | 100.0 |
| 1,2,3,4,7,8 H6CDF | ND | ND | ND |
| 1,2,3,6,7,8 H6CDF | ND | ND | ND |
| 1,2,3,7,8,9 H6CDF | ND | ND | ND |
| 2,3,4,6,7,8 H6CDF | 4.2 | ND | ND |
| 1,2,3,4,6,7,8 H7CDF | 8.3 | ND | 50.0 |
| 1,2,3,4,7,8,9 H7CDF | ND | ND | ND |
| T4CDF (TOTAL) | 95.8 | 87.5 | 100.0 |
| P5CDF (TOTAL) | 33.3 | ND | 100.0 |
| H6CDF (TOTAL) | 29.2 | ND | 50.0 |
| H7CDF (TOTAL) | 4.2 | ND | ND |
| O8CDF (TOTAL) | 4.2 | ND | 50.0 |


| Muscle Analyses |  |  |
| :---: | :---: | :---: |
| Peamouth | Whitefish | Flounder |
| 45 | 25 | 11 |
| 35.6 | 76.0 | ND |
| ND | ND | ND |
| ND | ND | ND |
| ND | 8.0 | ND |
| ND | ND | ND |
| 4.4 | 8.0 | ND |
| 37.8 | 76.0 | ND |
| ND | ND | ND |
| 4.4 | 8.0 | ND |
| 6.7 | 8.0 | ND |
| 28.9 | 16.0 | 36.4 |
|  |  |  |
| 95.6 | 80.0 | 18.2 |
| ND | ND | ND |
| ND | 16.0 | ND |
| ND | ND | ND |
| ND | ND | ND |
| ND | ND | ND |
| 2.2 | 24.0 | ND |
| 2.2 | 4.0 | ND |
| ND | ND | ND |
| 95.6 | 80.0 | 18.2 |
| 8.9 | 40.0 | ND |
| 2.2 | 12.0 | ND |
| 2.2 | 12.0 | ND |
| 4.4 | ND | ND |

Table 24. TEFs for chlorinated dioxin and furan congeners.

|  |  | TEF relative to 2,3,7,8-TCDD $^{1}$ |  |  |
| :---: | :---: | :---: | :---: | :---: |
| Congener | NATO | Mammals | Fish | Birds |
| dioxins |  |  |  |  |
| 2378 | 1 | 1 | 1 | 1 |
| 12378 | 0.5 | 1 | 1 | 1 |
| 123478 | 0.1 | 0.1 | 0.5 | 0.05 |
| 123678 | 0.1 | 0.1 | 0.01 | 0.01 |
| 123789 | 0.1 | 0.1 | 0.1 | 0.1 |
| 1234678 | 0.01 | 0.01 | 0.001 | $<0.001$ |
| OCDD | 0.001 | 0.0001 | $<0.0001$ | 0.0001 |
|  |  |  |  |  |
| furans |  |  |  |  |
| 2378 | 0.1 | 0.1 | 0.05 | 1 |
| 12378 | 0.05 | 0.05 | 0.05 | 0.1 |
| 23478 | 0.5 | 0.5 | 0.5 | 1 |
| 123478 | 0.1 | 0.1 | 0.1 | 0.1 |
| 123678 | 0.1 | 0.1 | 0.1 | 0.1 |
| 123789 | 0.1 | 0.1 | 0.1 | 0.1 |
| 234678 | 0.1 | 0.1 | 0.1 | 0.1 |
| 1234678 | 0.01 | 0.01 | 0.01 | 0.01 |
| 1234789 | 0.01 | 0.01 | 0.01 | 0.01 |
| OCDF | 0.001 | 0.0001 | $<0.0001$ | 0.0001 |

[^0]Table 25.Chlorophenolic congeners measured in fish tissues from the Fraser basin, 1994/1995

| Chlorophenols | Chlorocatechols | Chloroguaiacols |
| :---: | :---: | :---: |
| 4-Chlorophenol | 3-Chlorocatechol | 4-Chloroguaiacol |
| 2,3-Dichlorophenol | 4-Chlorocatechol | 5-Chloroguaiacol |
| 2,4/2,5-Dichlorophenol | 3,4-Dichlorocatechol | 6-Chloroguaiacol |
| 2,6-Dichlorophenol | 3,5-Dichlorocatechol | 3,4-Dichloroguaiacol |
| 3,4-Dichlorophenol | 3,6-Dichlorocatechol | 4,5-Dichloroguaiacol |
| 3,5-Dichlorophenol | 4,5-Dichlorocatechol | 4,6-Dichloroguaiacol |
| 2,3,4-Trichlorophenol | 3,4,5-Trichlorocatechol | 3,4,5-Trichloroguaiacol |
| 2,3,5-Trichlorophenol | 3,4,6-Trichlorocatechol | 3,4,6-Trichloroguaiacol |
| 2,3,6-Trichlorophenol | 3,4,5,6-Tetrachlorocatechol | 4,5,6-Trichloroguaiacol |
| 2,4,5-Trichlorophenol | Chloroveratroles | 3,4,5,6-Tetrachloroguaiacol |
| 2,4,6-Trichlorophenol | 4,5-Dichloroveratrole | Chlorosyringols |
| 3,4,5-Trichlorophenol | 3,4,5-Trichloroveratrole | 3-Chlorosyringol |
| 2,3,4,5-Tetrachlorophenol | 3,4,6-Trichloroveratrole | 3,5-Dichlorosyringol |
| 2,3,4,6-Tetrachlorophenol | 3,4,5,6-Tetrachloroveratrole | 3,4,5-Trichlorosyringol |
| 2,3,5,6-Tetrachlorophenol | Chlorosyringaldehydes. | Chlorovanillins |
| Pentachlorophenol | 2-Chlorosyringaldehyde 2,6-Dichlorosyringaldehyde | 5-Chlorovanillin 6-Chlorovanillin 56-Dichlorovanillin |

Table 26. Maximum concentrations and detections of chlorophenolics in fish tissues and bile in the Fraser basin, 1994. Maxima are shown as ng/g wet wt. for muscle/liver and $\mathrm{ng} / \mathrm{mL}$ bile. The number of detections/number of samples are shown in parentheses.

| Congener | Peamouth Chub |  |  | Mountain Whitefish |  |  | Starry Flounder |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Muscle | Liver | Bile | Muscle | Liver | Bile | Muscle | Liver |
| 4-Chlorophenol | 0.12 (4/44) | 8.90 (9/16) | 1.70 (1/11) | 0.09 (1/22) | 1.00 (5/8) | 16.00 (6/9) | ND | 0.56 (1/2) |
| 2,3-Dichlorophenol | ND | ND | ND | ND | ND | 4.90 (4/9) | ND | ND |
| 2,4/2,5-Dichlorophenol | 0.36 (21/44) | 1.40 (5/16) | 77.00 (4/11) | 0.28 (14/22) | 0.97 (3/8) | 320.00 (9/9) | 0.55 (5/12) | 1.20 (1/2) |
| 2,6-Dichlorophenol | ND | ND | 1.30 (1/11) | ND | 0.35 (1/8) | 12.00 (5/9) | ND | ND |
| 3,4-Dichlorophenol | 0.82 (9/44) | 19.00 (1/16) | 2.70 (1/11) | 1.20 (4/22) | 2.30 (4/8) | 7.70 (2/9) | 0.12 (1/12) | ND |
| 3,5-Dichlorophenol | 0.83 (18/44) | 1.90 (7/16) | 7.30 (3/11) | 2.40 (15/22) | 2.00 (3/8) | 19.00 (6/9) | ND | 1.00 (1/2) |
| 2,3,4-Trichlorophenol | ND | ND | 7.40 (1/11) | ND | ND | 1.10 (1/9) | ND | ND |
| 2,3,5-Trichlorophenol | ND | 0.26 (2/16) | 5.10 (1/11) | ND | ND | 2.20 (1/9) | ND | ND |
| 2,3,6-Trichlorophenol | ND | 0.26 (1/16) | 3.30 (1/11) | ND | ND | ND | ND | ND |
| 2,4,5-Trichlorophenol | ND | ND | 6.70 (2/11) | ND | ND | 3.70 (4/9) | ND | ND |
| 2,4,6-Trichlorophenol | 0.03 (1/44) | 0.72 (5/16) | 160.00 (8/11) | 0.06 (1/22) | 0.95 (1/8) | 180.00 (9/9) | ND | 0.19 (1/2) |
| 3,4,5-Trichlorophenol | ND | 0.51 (1/16) | 7.30 (3/11) | ND | ND | 18.00 (9/9) | ND | ND |
| 2,3,4,5-Tetrachlorophenol | ND | ND | 11.00 (1/11) | ND | ND | 3.60 (2/9) | ND | ND |
| 2,3,4,6-Tetrachlorophenol | ND | ND | 23.00 (7/11) | ND | ND | 19.00 (5/9) | ND | ND |
| 2,3,5,6-Tetrachlorophenol | 2.80 (1/44) | ND | 21.00 (3/11) | ND | ND | ND | ND | ND |
| Pentachlorophenol | 0.15 (4/44) | 2.00 (6/16) | 150.00 (11/11) | 0.06 (1/22) | 1.30 (1/8) | 38.00 (9/9) | ND | ND |
| 3-Chlorocatechol | 1.50 (6/44) | ND | 6.20 (1/11) | 0.90 (5/22) | ND | ND | 1.80 (6/12) | 10.00 (2/2) |
| 4-Chlorocatechol | 0.31 (5/44) | 1.50 (3/16) | ND | 0.13 (3/22) | ND | ND | 1.10 (8/12) | 2.00 (1/2) |
| 3,4-Dichlorocatechol | 0.56 (3/44) | 1.40 (1/16) | 4.30 (1/11) | 0.42 (1/22) | ND | ND | 0.70 (1/12) | ND |
| 3,5-Dichlorocatechol | ND | ND | 8.80 (1/11) | 0.23 (1/22) | ND | ND | ND | ND |
| 3,6-Dichlorocatechol | ND | ND | 10.00 (1/11) | 0 | ND | 12.00 (2/9) | ND | ND |
| 4,5-Dichlorocatechol | ND | 1.70 (2/16) | ND | 0 | 0.51 (1/8) | 9.40 (1/9) | 0.50 (1/12) | ND |
| 3,4,5-Trichlorocatechol | ND | ND | ND | ND | ND | 7.50 (1/9) | ND | ND |
| 3,4,6-Trichlorocatechol | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachlorocatechol | 0.15 (1/44) | ND | 1.30 (1/11) | ND | ND | 1.80 (1/9) | ND | ND |
| 4-Chloroguaiacol | ND | ND | 8.30 (3/11) | ND | ND | 28.00 (7/9) | ND | 0.21 (1/2) |
| 5-Chloroguaiacol | 0.22 (8/44) | 1.60 (6/16) | 87.00 (8/11) | 0.62 (11/22) | 1.30 (3/8) | 730.00 (7/9) | ND | ND |
| 6-Chloroguaiacol | ND | ND | 5.80 (2/11) | ND | ND | 1.30 (2/9) | ND | ND |
| 3,4-Dichloroguaiacol | ND | 0.74 (1/16) | 10.00 (3/11) | ND | 0.84 (2/8) | 160.00 (7/9) | ND | ND |
| 4,5-Dichloroguaiacol | 0.42 (4/44) | 12.00 (5/16) | 970.00 (4/11) | 0.14 (3/22) | 2.80 (1/8) | 1,400.00 (6/9) | ND | ND |
| 4,6-Dichloroguaiacol | ND | 0.24 (1/16) | 20.00 (3/11) | ND | ND | 55.00 (6/9) | ND | ND |
| 3,4,5-Trichloroguaiacol | 0.11 (3/44) | 5.80 (2/16) | 830.00 (4/11) | ND | 3.80 (1/8) | 240.00 (6/9) | ND | ND |
| 3,4,6-Trichloroguaiacol | ND | ND | 40.00 (4/11) | ND | ND | 103.00 (1/9) | ND | ND |
| 4,5,6-Trichloroguaiacol | 0.06 (2/44) | 1.20 (2/16) | 230.00 (4/11) | 0.07 (1/22) | 0.80 (1/8) | 667.00 (4/9) | ND | ND |
| 3,4,5,6-Tetrachloroguaiacol | ND | 0.54 (1/16) | 64.00 (10/11) | ND | 0.56 (1/8) | 180.00 (9/9) | ND | ND |
| 3-Chlorosyringol | 1.80 (1/44) | 2.00 (2/16) | 2.40 (2/11) | 0.50 (9/22) | ND | 2.80 (3/9) | ND | ND |
| 3,5-Dichlorosyringol | ND | ND | 20.00 (1/11) | ND | ND | 5.50 (1/9) | ND | ND |
| 3,4,5-Trichlorosyringol | ND | ND | 6.10 (1/11) | ND | ND | 13.00 (2/9) | ND | ND |
| 2-Chlorosyringaldehyde | 1.60 (6/44) | 0.56 (3/16) | ND | 1.20 (6/22) | ND | ND | ND | ND |
| 2,6-Dichlorosyringaldehyde | 2.10 (4/44) | 5.60 (4/16) | 3.30 (2/11) | 10.00 (5/22) | ND | 4.60 (5/9) | ND | ND |
| 5-Chlorovanillin | ND | ND | ND | ND | ND | ND | ND | ND |
| 6-Chlorovanillin | ND | ND | ND | ND | ND | 65.00 (4/9) | ND | ND |
| 5,6-Dichlorovanillin | ND | 1.30 (5/16) | ND | ND | ND | ND | ND | 0.92 (1/2) |
| 4,5-Dichloroveratrole | ND | ND | ND | ND | ND | ND | 0 | 0 |
| 3,4,5-Trichloroveratrole | 0.92 (6/44) | 2.50 (7/16) | 7.20 (2/11) | 2.00 (7/22) | ND | ND | ND | ND |
| 3,4,6-Trichloroveratrole | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachloroveratrole | 0.38 (11/44) | 1.80 (8/16) | 6.90 (1/11) | 0.76 (11/22) | ND | ND | ND | ND |

Table 27. Chlorophenolic congener group totals in peamouth chub from the Fraser basin, 1994 and bile in 1995. Values are means and ranges, in ng/g wet weight, with non-detects set to zero for calculating sums and averages.

|  |  | Peamouth Chub |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Total | Tissue | Hansard | Woodpecker | Marguerite | Mission | Barnston | North Arm | Main Arm | N. Thompson | Thompson |
| CI-phenolics | Muscle | 1.1 (0.7-1.8) | 1.6 (1.0-2.1) | 2.9 (1.4-6.2) | 0.4 (<0.05-1.7) | 0.3 (<0.08-0.7) | 0.5 (<0.05-0.8) | 0.1 (<0.06-0.4) | 1.8 (0.8-3.8) | 1.0 (0.2-1.8) |
|  | Liver | 1.9 (0.8-2.9) | 2.0 (1.8-2.1) | 28.5 | 9.7 | 14.4 | 10.6 (7.9-13.4) | 10.3 (9.1-11.5) | 4.5 (2.4-6.7) | 15.3 (8.0-22.7) |
|  | Bile 94 | 109.5 (36.8-182.3) | 226 (32-420) | 1,523 (482-2,565) | no samples | no samples | no samples | no samples | 2 | 32.1 (17.3-46.9) |
|  | Bile 95 | 19.8 (15.8-23.9) | 203.5 (170.7-236.2) | 321.9 |  |  | 261.2 (151.7-345.3) | 182.6 (169.0-196.3) | 19.5 (10.0-29.0) | 550.8 (494.2-607.5) |
| Cl-phenols | Muscle | 0.6 (0.2-1.6) | 0.5 (0.2-1.1) | 0.89 (0.07-2.92) | 0.1 (<0.08-0.3) | 0.3 (<0.08-0.7) | 0.4 (<0.09-0.8) | 0.1 (<0.10-0.4) | 1.25 (0.7-1.6) | 0.1 (<0.03-0.2) |
|  | Liver | 1.0 (<0.1-2.0) | 0.3 (<0.1-0.5) | 4.4 | 3.6 | 6.4 | 6.3 (3.4-9.2) | 5.1 (2.2-8.0) | 1.20 (1.10-1.30) | 11.1 (1.9-20.3) |
|  | Bile 94 | 106.5 (32.8-180.1) | 85.1 (21.7-148.5) | 223 (138-309) | no samples | no samples | no samples | no samples | 2 | 5.1 (4.7-5.6) |
|  | Bile 95 | 7.0 (6.9-7.1) | 27.1 (26.4-27.8) | 82.9 |  |  | 101.7 (45.9-150) | 67.1 (65.8-68.3) | 8.8 (5.8-11.8) | 9.4 (8.7-10.1) |
| CI-catechols | Muscle | 0.1 (<0.04-0.2) | 0.3 (<0.04-0.6) | 0.2 (0.1-0.3) | 0.4 (<0.06-1.7) | <0.05 | 0.03 (<0.08-0.15) | <0.03 | 0.1 (<0.02-0.2) | <0.01 |
|  | Liver | <0.12 | <0.25 | <0.09 | 0.8 | 2.4 | <0.05 | 1.6 (<0.04-3.2) | <0.09 | 0.7 (<0.22-1.4) |
|  | Bile 94 | <0.21 | 15.3 (<0.3-30.6) | <0.49 | no samples | no samples | no samples | no samples | <0.22 | <0.17 |
|  | Bile 95 | <1.72 | 7.3 (2.0-12.5) | <2.5 |  |  | <0.93 | 1.2 (<1.6-2.5) | <1.1 | 0.7 (<1.10-1.5) |
| Cl-guaiacols | Muscle | 0.04 (<0.03-0.14) | 0.04 (<0.02-0.2) | 0.4 (0.3-0.5) | <0.02 | <0.04 | <0.06 | <0.07 | 0.1 (<0.01-0.2) | 0.10 (<0.02-0.2) |
|  | Liver | 0.9 (0.8-0.9) | <0.14 | 22.1 | 0.8 | 1.4 | 0.2 (<0.16-0.5) | 0.2 (<0.12-0.3) | <0.04 | 1.2 (1.0-1.3) |
|  | Bile 94 | 3.1 (2.2-4.0) | 102.6 (10.2-194.9) | 1,299 (343-2,254) | no samples | no samples | no samples | no samples | $<0.21$ | 26.9 (11.7-42.2) |
|  | Bile 95 | 4.07 (2.2-5.9) | 64.7 (53.4-76) | 79.3 |  |  | 29.5 (21.5-33.8) | 23.4 (19.5-27.3) | $1.7(<0.41-3.3)$ | 265.5 (236.9-294.2) |
| Other | Muscle | 0.41 (0.05-0.77) | 0.7 (0.5-1.2) | 1.4 (0.8-2.5) | <0.03 | <0.06 | <0.05) | <0.04 | $0.5(<0.01-2.1)$ | 0.8 (<0.03-1.6) |
|  | Liver | $<0.07$ | 1.7 (1.6-1.8) | 2 | 4.5 | 4.2 | 4.1 (3.7-4.6) | 3.5 (3.2-3.7) | 3.4 (1.1-5.6) | 2.4 (1.4-3.4) |
|  | Bile 94 | <0.17 | 22.95 (<0.41-45.90) | 0.8 (<0.3-1.6) | no samples | no samples | no samples | no samples | <0.14 | <0.17 |
|  | Bile 95 | <1.4 | 5.4 (3.6-7.2) | <1.6 |  |  | <1.2 | <1.1 | <0.61 | <0.45 |


|  |  | Flounder |  | Whitefish |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Total | Tissue | North Arm | Main Arm | Hansard | Woodpecker | Marguerite | N. Thompson | Thompson |
| CI-phenolics | Muscle | 1.1 (0.4-2.9) | 1.4 (0.7-2.9) | 1.1 (0.5-1.8) | 1.3 (0.6-2.5) | 2.2 (0.3-3.5) | 4.7 (0.9-13.3) | 2.2 (0.7-3.4) |
|  | Liver | 8.1 | 12.2 | 1.3 | 3.1 (2.9-3.2) | 15.1 | 3.1 (2.8-3.4) | 2.1 (1.3-2.9) |
|  | Bile 94 | no samples | no samples | 188 (141-234) | 345 (270-419) | 3680 | 315 (304-325) | 784 (632-935) |
|  | Bile 95 | 56.9 (50.7-63.1) | 145.7 (103.5-187.9) | 131.8 | 777.7 (677.4-878.0) | 1561.40 | 672.4 (646.2-698.6) | 2,178 (1,860-2496) |
| Cl-phenols | Muscle | 0.4 (<0.02-0.5) | 0.02 (<0.03-0.1) | 0.4 (0.1-0.5) | 0.16 (<0.02-0.28) | 0.4 (<0.04-1.0) | 0.8 (<0.03-2.5) | 0.85 (0.2-1.7) |
|  | Liver | 2.9 | -0.08 | 1.3 | 3.1 (2.9-3.2) | 5.3 | 2.9 (2.8-3.2) | 0.9 (<0.10-1.8) |
|  | Bile 94 | no samples | no samples | 156 (120-192) | 81.2 (69.6-92.8) | 584.5 | 265 (248-281) | 58.6 (48.4-68.9) |
|  | Bile 95 | 21.3 (17.7-24.8) | 24.6 (14.4-34.8) | 56.3 | 111.20 (99.00-123.4) | 153.2 | 256.0 (254.1-258) | 84.3 (71.4-97.2) |
| Cl-catechols | Muscle | 0.7 (<0.08-2.9) | 1.3 (0.5-2.9) | <0.04 | 0.3 (<0.03-1.0) | 0.3 (<0.05-0.4) | <0.04 | 0.1 (<0.12-0.4) |
|  | Liver | 4.2 | 12 | <0.1 | <0.09 | 0.5 | <0.08 | <0.08 |
|  | Bile 94 | no samples | no samples | <1.10 | <0.29 | 23.8 | <1.70 | 6.0 (<1.50-12.0) |
|  | Bile 95 | <1.26 | 1.6 (<0.58-3.1) | <1.3 | 0.8 (<0.86-1.7) | 4.6 | 25.4(13.1-37.6) | 18.8 (17.2-20.4) |
| Cl-guaiacols | Muscle | <0.01 | <0.01 | 0.2 (<0.02-0.3) | 0.09 (<0.02-0.20) | 0.1 (<0.03-0.3) | 0.1 (<0.02-0.4) | 0.2 (<0.03-0.6) |
|  | Liver | $<0.07$ | 0.21 | <0.11 | <0.08 | 9.2 | 0.1 (<0.04-0.2) | 1.2 (1.1-1.3) |
|  | Bile 94 | no samples | no samples | 27.5 (21.0-34.0) | 200 (134-267) | 3013 | 47.0 (44.2-49.7) | 707 (562-853) |
|  | Bile 95 | 6.8 (5.2-8.3) | 46.9 (34.2-59.5) | 7.6 | 238.3 (198.9-277.6) | 596.4 | 49.9 (49.7-50.2) | 973.0 (828.7-1117.4) |
| Other | Muscle | <0.02 | <0.01 | 0.5 (<0.04-1.0) | 0.8 (0.2-2.0) | 1.3 (0.3-2.8) | 3.7 (0.9-10.4) | 1.1 (0.4-2.4) |
|  | Liver | 0.9 | <0.06 | <0.07 | <0.08 | <0.11 | <0.06 | <0.08 |
|  | Bile 94 | no samples | no samples | 4.0 (<4.9-8.0) | 63.5 (60.0-67.0) | 58.7 | 3.2 (<0.73-6.4) | 11.8 (1.9-21.7) |
|  | Bile 95 | 2.2 (<1.20-4.4) | <0.37 | 5.2 | 77.6 (72.6-82.5) | 53.0 | 9.7 (4.6-14.8) | 25.7 (18.8-32.6) |

Table 28. Measured chlorophenolic concentrations in peamouth chub muscle from the Fraser basin, 1994. Detection limits shown as the average (range) and concentrations shown as mean (SE) in ng/g wet wt.


Table 29. Measured chlorophenolic concentrations in whitefish and flounder muscle from the Fraser basin, 1994. Detection limits shown as the average (range) and concentrations shown as mean (SE) in ng/g wet wt.

|  | Starry Flounder |  |  |  | Mountain Whitefish Muscle |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Detect Limits | Main Arm | North Arm | Detect Limits | Hansard | Woodpecker | Marguerite | N. Thompson | Thompson |
| 4-Chlorophenol | 0.14 (0.06-0.26) | ND | ND | 2.65 (0.86-7.70) | ND | ND | 0.06 (0.01) | ND | ND |
| 2,3-Dichlorophenol | 0.09 (0.04-0.21) | ND | ND | 3.46 (0.96-8.90) | ND | ND | ND | ND | ND |
| 2,4/2,5-Dichlorophenol | 0.06 (0.02-0.12) | ND | 0.38 (0.09) | 2.02 (0.49-6.80) | 0.13 (0.03) | ND | 0.11 (0.02) | 0.15 (0.04) | 0.13 (0.04) |
| 2,6-Dichlorophenol | 0.06 (0.02-0.14) | ND | ND | 1.97 (0.32-6.90) | ND | ND | ND | ND | ND |
| 3,4-Dichlorophenol | 0.07 (0.02-0.15) | 0.04 (0.02) | ND | 0.19 (0.07-0.48) | ND | 0.06 (0.03) | 0.12 (0.08) | ND | 0.39 (0.27) |
| 3,5-Dichlorophenol | 0.11 (0.04-0.24) | ND | ND | 0.22 (0.07-0.66) | 0.25 (0.06) | 0.15 (0.06) | 0.22 (0.07) | 0.74 (0.56) | 0.29 (0.07) |
| 2,3,4-Trichlorophenol | 0.07 (0.02-0.16) | ND | ND | 0.22 (0.07-0.54) | ND | ND | ND | ND | ND |
| 2,3,5-Trichlorophenol | 0.06 (0.02-0.12) | ND | ND | 0.30 (0.07-1.00) | ND | ND | ND | ND | ND |
| 2,3,6-Trichlorophenol | 0.06 (0.02-0.12) | ND | ND | 0.93 (0.07-2.92) | ND | ND | ND | ND | ND |
| 2,4,5-Trichlorophenol | 0.06 (0.02-0.12) | ND | ND | 2.81 (0.48-7.30) | ND | ND | ND | ND | ND |
| 2,4,6-Trichlorophenol | 0.05 (0.02-0.09) | ND | ND | 1.25 (0.31-3.00) | ND | ND | 0.02 (0.01) | ND | ND |
| 3,4,5-Trichlorophenol | 0.07 (0.02-0.16) | ND | ND | 1.60 (0.22-5.30) | ND | ND | ND | ND | ND |
| 2,3,4,5-Tetrachlorophenol | 0.09 (0.02-0.21) | ND | ND | 0.29 (0.08-0.80) | ND | ND | ND | ND | ND |
| 2,3,4,6-Tetrachlorophenol | 0.08 (0.01-0.14) | ND | ND | 0.38 (0.13-0.90) | ND | ND | ND | ND | ND |
| 2,3,5,6-Tetrachlorophenol | 0.14 (0.02-0.24) | ND | ND | 0.26 (0.10-0.72) | ND | ND | ND | ND | ND |
| Pentachlorophenol | 0.07 (0.02-0.14) | ND | ND | 0.45 (0.18-0.98) | ND | ND | 0.04 (0.01) | ND | ND |
| 3-Chlorocatechol | 0.17 (0.04-0.45) | 0.82 (0.33) | 0.35 (0.20) | 0.11 (0.04-0.34) | ND | 0.26 (0.21) | 0.16 (0.04) | ND | 0.17 (0.04) |
| 4-Chlorocatechol | 0.10 (0.02-0.28) | 0.53 (0.16) | 0.32 (0.15) | 0.81 (0.29-2.10) | ND | 0.07 (0.03) | ND | ND | 0.07 (0.02) |
| 3,4-Dichlorocatechol | 0.37 (0.03-1.60) | ND | 0.30 (0.14) | 0.19 (0.08-0.65) | ND | ND | 0.20 (0.09) | ND | ND |
| 3,5-Dichlorocatechol | 0.63 (0.05-2.60) | ND | ND | 0.13 (0.04-0.25) | ND | ND | 0.23 (0.08) | ND | ND |
| 3,6-Dichlorocatechol | 0.41 (0.03-1.70) | ND | ND | 0.20 (0.07-0.66) | ND | ND | 0.12 (0.07) | ND | ND |
| 4,5-Dichlorocatechol | 0.33 (0.02-1.40) | ND | 0.25 (0.12) | 0.50 (0.09-1.60) | ND | ND | 0.10 (0.05) | ND | ND |
| 3,4,5-Trichlorocatechol | 0.06 (0.02-0.11) | ND | ND | 0.58 (0.21-1.30) | ND | ND | ND | ND | ND |
| 3,4,6-Trichlorocatechol | 0.08 (0.03-0.14) | ND | ND | 3.97 (0.62-4.00) | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachlorocatechol | 0.07 (0.01-0.15) | ND | ND | 0.23 (0.08-0.48) | ND | ND | ND | ND | ND |
| 4-Chloroguaiacol | 0.04 (0.01-0.08) | ND | ND | 1.87 (0.35-5.10) | ND | ND | ND | ND | ND |
| 5-Chloroguaiacol | 0.06 (0.02-0.12) | ND | ND | 1.76 (0.20-6.30) | 0.21 (0.05) | 0.10 (0.05) | 0.05 (0.03) | 0.15 (0.09) | 0.21 (0.14) |
| 6-Chloroguaiacol | 0.04 (0.02-0.09) | ND | ND | 0.41 (0.18-1.10) | ND | ND | ND | ND | ND |
| 3,4-Dichloroguaiacol | 0.08 (0.02-0.21) | ND | ND | 0.12 (0.05-0.32) | ND | ND | ND | ND | ND |
| 4,5-Dichloroguaiacol | 0.06 (0.01-0.14) | ND | ND | 0.44 (0.17-0.79) | ND | ND | 0.07 (0.02) | ND | ND |
| 4,6-Dichloroguaiacol | 0.07 (0.02-0.16) | ND | ND | 0.84 (0.14-3.10) | ND | ND | ND | ND | ND |
| 3,4,5-Trichloroguaiacol | 0.07 (0.03-0.15) | ND | ND | 0.90 (0.14-3.40) | ND | ND | ND | ND | ND |
| 3,4,6-Trichloroguaiacol | 0.06 (0.02-0.12) | ND | ND | 2.38 (0.04-6.00) | ND | ND | ND | ND | ND |
| 4,5,6-Trichloroguaiacol | 0.04 (0.02-0.08) | ND | ND | 0.36 (0.13-1.10) | ND | ND | 0.04 (0.01) | ND | ND |
| 3,4,5,6-Tetrachloroguaiacol | 0.06 (0.01-0.10) | ND | ND | 0.19 (0.08-0.45) | ND | ND | ND | ND | ND |
| 3-Chlorosyringol | 0.05 (0.02-0.10) | ND | ND | 0.15 (0.05-0.41) | 0.20 (0.04) | 0.11 (0.05) | ND | 0.17 (0.09) | 0.16 (0.11) |
| 3,5-Dichlorosyringol | 0.06 (0.01-0.13) | ND | ND | 4.03 (1.10-9.00) | ND | ND | ND | ND | ND |
| 3,4,5-Trichlorosyringol | 0.04 (0.02-0.08) | ND | ND | 0.10 (0.04-0.19) | ND | ND | ND | ND | ND |
| 2-Chlorosyringaldehyde | 0.23 (0.03-0.58) | ND | ND | 2.39 (0.07-0.89) | 0.40 (0.21) | 0.07 (0.04) | ND | 0.56 (0.26) | 0.11 (0.08) |
| 2,6-Dichlorosyringaldehyde | 0.20 (0.09-0.38) | ND | ND | 2.57 (0.55-1.00) | ND | 0.29 (0.27) | ND | 2.99 (2.38) | 0.87 (0.56) |
| 5-Chlorovanillin | 0.19 (0.07-0.38) | ND | ND | 0.10 (0.06-0.17) | ND | ND | ND | ND | ND |
| 6-Chlorovanillin | 0.23 (0.10-0.42) | ND | ND | 0.48 (0.12-1.60) | ND | ND | ND | ND | ND |
| 5,6-Dichlorovanillin | 0.09 (0.07-0.12) | ND | ND | 0.71 (0.17-2.70) | ND | ND | ND | ND | ND |
| 4,5 -Dichloroveratrole | 0.27 (0.07-0.67) | ND | ND | 2.14 (0.65-5.80) | ND | ND | ND | ND | ND |
| 3,4,5-Trichloroveratrole | 0.17 (0.04-0.42) | ND | ND | 0.13 (0.04-0.36) | ND | 0.18 (0.01) | 0.87 (0.28) | ND | ND |
| 3,4,6-Trichloroveratrole | 0.06 (0.03-0.11) | ND | ND | 2.79 (0.37-8.27) | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachloroveratrole | 0.21 (0.06-0.38) | ND | ND | 0.28 (0.09-0.50) | ND | 0.34 (0.06) | 0.35 (0.12) | 0.30 (0.09) | 0.13 (0.05) |

Table 30. Measured chlorophenolic concentrations in peamouth chub liver from the Fraser basin, 1994. Detection limits shown as the average (range) and concentrations shown as mean in ng/g wet wt.

|  |  | Peamouth_Chub_iver |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Detect Limits | Hansard | Woodpecker | Marguerite | Mission | Barnston | Main Arm | North Arm | N. Thompson | Thompson |
| 4-Chlorophenol | 0.19 (0.01-0.68) | ND | ND | 0.40 | 2.50 | 4.70 | 5.65 | 2.84 | ND | ND |
| 2,3-Dichlorophenol | 0.19 (0.04-0.66) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,4/2,5-Dichlorophenol | 0.27 (0.06-0.89) | ND | ND | 1.40 | 0.68 | 0.45 | ND | 0.51 | ND | ND |
| 2,6-Dichlorophenol | 0.17 (0.03-0.43) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4-Dichlorophenol | 0.11 (0.01-0.47) | ND | ND | ND | ND | ND | ND | ND | ND | 9.60 |
| 3,5-Dichlorophenol | 0.20 (0.01-0.79) | ND | ND | 0.86 | ND | ND | ND | 0.85 | 0.93 | 1.60 |
| 2,3,4-Trichlorophenol | 0.14 (0.01-0.32) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,3,5-Trichlorophenol | 0.16 (0.02-0.36) | ND | ND | ND | ND | 0.18 | 0.20 | ND | ND | ND |
| 2,3,6-Trichlorophenol | 0.13 (0.01-0.37) | ND | ND | ND | ND | ND | 0.20 | ND | ND | ND |
| 2,4,5-Trichlorophenol | 0.11 (0.03-0.25) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,4,6-Trichlorophenol | 0.17 (0.04-0.64) | ND | ND | 0.72 | 0.40 | 0.37 | 0.18 | 0.18 | ND | ND |
| 3,4,5-Trichlorophenol | 0.13 (0.02-0.31) | ND | 0.30 | ND | ND | ND | ND | ND | ND | ND |
| 2,3,4,5-Tetrachlorophenol | 0.12 (0.01-0.35) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,3,4,6-Tetrachlorophenol | 0.10 (0.01-0.29) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,3,5,6-Tetrachlorophenol | 0.21 (0.02-0.51) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| Pentachlorophenol | 0.16 (0.01-0.50) | 1.13 | ND | 1.00 | ND | 0.70 | 0.35 | 0.82 | ND | ND |
| 3-Chlorocatechol | 0.23 (0.02-0.85) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 4-Chlorocatechol | 0.14 (0.01-0.53) | ND | ND | ND | 0.78 | 1.40 | ND | 0.78 | ND | ND |
| 3,4-Dichlorocatechol | 0.37 (0.01-2.20) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,5-Dichlorocatechol | 0.55 (0.01-3.90) | ND | ND | ND | ND | ND | ND | ND | ND | 0.65 |
| 3,6-Dichlorocatechol | 0.36 (0.02-2.50) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 4,5-Dichlorocatechol | 0.43 (0.01-2.00) | ND | ND | ND | ND | 1.00 | ND | 1.13 | ND | ND |
| 3,4,5-Trichlorocatechol | 0.26 (0.03-0.65) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,6-Trichlorocatechol | 0.28 (0.01-0.59) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachlorocatechol | 0.10 (0.02-0.39) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 4-Chloroguaiacol | 0.09 (0.01-0.20) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 5-Chloroguaiacol | 0.14 (0.01-0.31) | 0.86 | ND | 1.60 | ND | ND | ND | ND | 0.12 | 1.15 |
| 6-Chloroguaiacol | 0.12 (0.03-0.24) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4-Dichloroguaiacol | 0.19 (0.02-0.59) | ND | ND | 0.74 | ND | ND | ND | ND | ND | ND |
| 4,5-Dichloroguaiacol | 0.13 (0.01-0.39) | ND | ND | 12.00 | 0.76 | 0.92 | 0.30 | 0.20 | ND | ND |
| 4,6-Dichloroguaiacol | 0.14 (0.02-0.43) | ND | ND | 0.24 | ND | ND | ND | ND | ND | ND |
| 3,4,5-Trichloroguaiacol | 0.21 (0.01-0.54) | ND | ND | 5.80 | ND | 0.24 | ND | ND | ND | ND |
| 3,4,6-Trichloroguaiacol | 0.19 (0.02-0.43) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 4,5,6-Trichloroguaiacol | 0.16 (0.01-0.58) | ND | ND | 1.20 | ND | 0.20 | ND | ND | ND | ND |
| 3,4,5,6-Tetrachloroguaiacol | 0.13 (0.02-0.31) | ND | ND | 0.54 | ND | ND | ND | ND | ND | ND |
| 3-Chlorosyringol | 0.11 (0.01-0.28) | ND | ND | ND | ND | ND | ND | ND | ND | 1.70 |
| 3,5-Dichlorosyringol | 0.32 (0.01-0.74) | ND | ND | ND | 0.20 | ND | ND | ND | ND | ND |
| 3,4,5-Trichlorosyringol | 0.23 (0.01-0.61) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2-Chlorosyringaldehyde | 0.13 (0.04-0.35) | ND | ND | ND | ND | 0.20 | 0.37 | ND | ND | ND |
| 2,6-Dichlorosyringaldehyde | 0.40 (0.03-1.20) | ND | ND | ND | ND | ND | ND | ND | 2.98 | 0.90 |
| 5-Chlorovanillin | 0.63 (0.01-1.90) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 6-Chlorovanillin | 0.70 (0.01-2.20) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 5,6-Dichlorovanillin | 0.20 (0.01-0.59) | ND | ND | ND | 0.53 | 1.30 | 0.30 | 0.35 | ND | ND |
| 4,5 -Dichloroveratrole | 0.50 (0.02-2.00) | ND | ND | ND | NQ | NQ | ND | ND | ND | ND |
| 3,4,5-Trichloroveratrole | 0.31 (0.01-0.87) | ND | ND | 2.00 | 2.40 | 1.50 | 2.20 | 1.80 | ND | ND |
| 3,4,6-Trichloroveratrole | 0.19 (0.01-0.55) | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachloroveratrole | 0.35 (0.02-0.86) | ND | 1.70 | ND | 1.40 | 1.20 | 1.40 | 1.30 | ND | ND |

Table 31. Measured chlorophenolic concentrations in whitefish and flounder liver from the Fraser basin, 1994. Detection limits shown as the average (range) and concentrations shown as mean in ng/g wet wt.

|  |  | arry Flound |  |  |  | Mo | ain Whitefish |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Detect Limits | Main Arm | North Arm | Detect Limits | Hansard | Woodpecker | Marguerite | N. Thompson | Thompson |
| 4-Chlorophenol | 0.30 (0.22-0.38) | 0.56 | ND | 0.27 (0.02-0.82) | ND | 0.96 | 0.77 | 0.61 | ND |
| 2,3-Dichlorophenol | 0.16 (0.13-0.19) | ND | ND | 0.31 (0.02-0.72) | ND | ND | ND | ND | ND |
| 2,4/2,5-Dichlorophenol | 0.10 (0.08-0.12) | 1.20 | ND | 0.25 (0.14-0.41) | ND | ND | 0.97 | 0.42 | ND |
| 2,6-Dichlorophenol | 0.11 (0.09-0.12) | ND | ND | 0.34 (0.19-0.49) | ND | ND | ND | ND | ND |
| 3,4-Dichlorophenol | 0.12 (0.10-0.13) | ND | ND | 0.32 (0.13-0.55) | 1.30 | 2.10 | ND | ND | 1.02 |
| 3,5-Dichlorophenol | 0.19 (0.16-0.22) | 1.00 | ND | 0.44 (0.14-0.82) | ND | ND | 1.00 | 1.95 | ND |
| 2,3,4-Trichlorophenol | 0.17 (0.15-0.19) | ND | ND | 0.10 (0.01-0.21) | ND | ND | ND | ND | ND |
| 2,3,5-Trichlorophenol | 0.11 (0.10-0.12) | ND | ND | 0.11 (0.01-0.28) | ND | ND | ND | ND | ND |
| 2,3,6-Trichlorophenol | 0.11 (0.10-0.12) | ND | ND | 0.12 (0.03-0.26) | ND | ND | ND | ND | ND |
| 2,4,5-Trichlorophenol | 0.14 (0.12-0.15) | ND | ND | 0.14 (0.08-0.20) | ND | ND | ND | ND | ND |
| 2,4,6-Trichlorophenol | 0.09 (0.08-0.10) | 0.19 | ND | 0.15 (0.12-0.20) | ND | ND | 0.95 | ND | ND |
| 3,4,5-Trichlorophenol | 0.17 (0.15-0.18) | ND | ND | 0.38 (0.04-1.20) | ND | ND | ND | ND | ND |
| 2,3,4,5-Tetrachlorophenol | 0.26 (0.19-0.33) | ND | ND | 0.15 (0.02-0.54) | ND | ND | ND | ND | ND |
| 2,3,4,6-Tetrachlorophenol | 0.18 (0.17-0.19) | ND | ND | 0.05 (0.01-0.10) | ND | ND | ND | ND | ND |
| 2,3,5,6-Tetrachlorophenol | 0.32 (0.30-0.33) | ND | ND | 0.09 (0.01-0.17) | ND | ND | ND | ND | ND |
| Pentachlorophenol | 0.17 (0.12-0.22) | ND | ND | 0.14 (0.03-0.33) | ND | ND | 1.30 | ND | ND |
| 3-Chlorocatechol | 0.24 (0.19-0.30) | 4.20 | 10.00 | 0.29 (0.04-0.62) | ND | ND | ND | ND | ND |
| 4-Chlorocatechol | 0.15 (0.11-0.18) | ND | 2.00 | 0.19 (0.04-0.39) | ND | ND | ND | ND | ND |
| 3,4-Dichlorocatechol | 0.69 (0.43-0.94) | ND | ND | 0.63 (0.32-1.10) | ND | ND | ND | ND | ND |
| 3,5-Dichlorocatechol | 1.18 (0.75-1.60) | ND | ND | 0.73 (0.16-1.90) | ND | ND | ND | ND | ND |
| 3,6-Dichlorocatechol | 0.74 (0.48-1.00) | ND | ND | 0.46 (0.06-1.20) | ND | ND | ND | ND | ND |
| 4,5-Dichlorocatechol | 0.61 (0.38-0.83) | ND | ND | 0.34 (0.04-1.00) | ND | ND | 0.51 | ND | ND |
| 3,4,5-Trichlorocatechol | 0.17 (0.16-0.18) | ND | ND | 0.33 (0.21-0.53) | ND | ND | ND | ND | ND |
| 3,4,6-Trichlorocatechol | 0.20 (0.19-0.21) | ND | ND | 0.19 (0.08-0.34) | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachlorocatechol | 0.19 (0.12-0.25) | ND | ND | 0.37 (0.08-1.50) | ND | ND | ND | ND | ND |
| 4-Chloroguaiacol | 0.06 (0.06-0.07) | ND | 0.21 | 0.11 (0.03-0.23) | ND | ND | ND | ND | ND |
| 5-Chloroguaiacol | 0.09 (0.08-0.10) | ND | ND | 0.20 (0.05-0.40) | ND | ND | 0.41 | ND | 1.20 |
| 6-Chloroguaiacol | 0.08 (0.07-0.08) | ND | ND | 0.11 (0.03-0.28) | ND | ND | ND | ND | ND |
| 3,4-Dichloroguaiacol | 0.16 (0.10-0.21) | ND | ND | 0.30 (0.15-0.60) | ND | ND | 0.84 | 0.19 | ND |
| 4,5-Dichloroguaiacol | 0.10 (0.06-0.14) | ND | ND | 0.09 (0.03-0.19) | ND | ND | 2.80 | ND | ND |
| 4,6-Dichloroguaiacol | 0.12 (0.08-0.16) | ND | ND | 0.13 (0.04-0.20) | ND | ND | ND | ND | ND |
| 3,4,5-Trichloroguaiacol | 0.14 (0.10-0.18) | ND | ND | 0.74 (0.19-2.99) | ND | ND | 3.80 | ND | ND |
| 3,4,6-Trichloroguaiacol | 0.11 (0.08-0.14) | ND | ND | 0.20 (0.11-0.38) | ND | ND | ND | ND | ND |
| 4,5,6-Trichloroguaiacol | 0.08 (0.06-0.10) | ND | ND | 0.12 (0.04-0.18) | ND | ND | 0.80 | ND | ND |
| 3,4,5,6-Tetrachloroguaiacol | 0.13 (0.09-0.16) | ND | ND | 0.46 (0.10-1.60) | ND | ND | 0.56 | ND | ND |
| 3-Chlorosyringol | 0.10 (0.06-0.13) | ND | ND | 0.08 (0.04-0.19) | ND | ND | ND | ND | ND |
| 3,5-Dichlorosyringol | 0.45 (0.41-0.49) | ND | ND | 0.53 (0.29-1.04) | ND | ND | ND | ND | ND |
| 3,4,5-Trichlorosyringol | 0.17 (0.14-0.19) | ND | ND | 0.14 (0.05-0.30) | ND | ND | ND | ND | ND |
| 2-Chlorosyringaldehyde | 0.09 (0.08-0.09) | ND | ND | 0.48 (0.09-1.20) | ND | ND | ND | ND | ND |
| 2,6-Dichlorosyringaldehyde | 0.64 (0.48-0.80) | ND | ND | 0.22 (0.13-0.39) | ND | ND | ND | ND | ND |
| 5-Chlorovanillin | 0.50 (0.44-0.55) | ND | ND | 0.40 (0.02-1.10) | ND | ND | ND | ND | ND |
| 6-Chlorovanillin | 0.55 (0.49-0.60) | ND | ND | 0.43 (0.03-1.20) | ND | ND | ND | ND | ND |
| 5,6-Dichlorovanillin | 0.16 (0.15-0.16) | 0.92 | ND | 0.15 (0.05-0.26) | ND | ND | ND | ND | ND |
| 4,5-Dichloroveratrole | 0.95 (0.95-0.95) | NQ | ND | 0.46 (0.04-1.30) | ND | ND | ND | ND | ND |
| 3,4,5-Trichloroveratrole | 0.47 (0.40-0.54) | ND | ND | 0.42 (0.08-1.20) | ND | ND | ND | ND | ND |
| 3,4,6-Trichloroveratrole | 0.32 (0.27-0.37) | ND | ND | 0.36 (0.11-0.77) | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachloroveratrole | 0.42 (0.39-0.45) | ND | ND | 0.56 (0.14-1.00) | ND | ND | ND | ND | ND |

Table 32. Measured chlorophenolic concentrations in peamouth and whitefish bile from the Fraser basin, 1994. Detection limits shown as the average (range) and concentrations shown as mean in $\mathrm{ng} / \mathrm{mL}$ bile.

|  |  |  |  | eamouth Bile |  |  |  | White | Bile |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Detect Limits | Hansard | Woodpecker | Marguerite | N. Thompson | Thompson | Hansard | Woodpecker | Marguerite | N. Thompson | Thompson |
| 4-Chlorophenol | 0.37 (0.02-1.60) | ND | ND | 1.10 | ND | ND | 4.95 | ND | 2.4 | 11.9 | 2.45 |
| 2,3-Dichlorophenol | 0.41 (0.03-2.40) | ND | ND | ND | ND | ND | 5.70 | ND | ND | 4.1 | 2.25 |
| 2,4/2,5-Dichlorophenol | 0.26 (0.02-1.55) | ND | 3.55 | 41.10 | ND | ND | 65.00 | 22.00 | 320 | 180 | 26.00 |
| 2,6-Dichlorophenol | 0.43 (0.01-2.40) | ND | 0.80 | ND | ND | ND | 8.75 | ND | 6.6 | 6.3 | 2.55 |
| 3,4-Dichlorophenol | 0.31 (0.04-1.71) | ND | 1.06 | ND | ND | ND | ND | 5.75 | ND | ND | ND |
| 3,5-Dichlorophenol | 0.53 (0.02-2.83) | ND | 4.05 | 1.84 | ND | ND | 13.75 | ND | 3.5 | 6.6 | 4.55 |
| 2,3,4-Trichlorophenol | 0.31 (0.04-1.16) | ND | 2.20 | ND | ND | ND | 1.89 | ND | 1.1 | ND | ND |
| 2,3,5-Trichlorophenol | 0.36 (0.03-1.50) | ND | 1.67 | ND | ND | ND | 3.18 | ND | ND | ND | ND |
| 2,3,6-Trichlorophenol | 0.33 (0.03-1.51) | ND | 1.21 | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,4,5-Trichlorophenol | 0.35 (0.02-2.00) | ND | 2.09 | ND | ND | ND | ND | 2.35 | 2.4 | 0.82 | ND |
| 2,4,6-Trichlorophenol | 0.25 (0.01-1.11) | ND | 11.30 | 116.50 | ND | 2.93 | ND | 11.00 | 180 | 30.50 | 5.70 |
| 3,4,5-Trichlorophenol | 0.27 (0.05-1.01) | ND | 3.10 | 1.61 | ND | ND | ND | 10.70 | 7.9 | 17.50 | 12.00 |
| 2,3,4,5-Tetrachlorophenol | 0.36 (0.03-2.00) | ND | 3.04 | ND | ND | ND | ND | ND | 3.6 | ND | ND |
| 2,3,4,6-Tetrachlorophenol | 0.44 (0.03-2.99) | 14.85 | 6.98 | 22.50 | ND | ND | ND | 6.40 | 19 | 1.15 | ND |
| 2,3,5,6-Tetrachlorophenol | 0.72 (0.02-5.08) | 4.10 | 6.13 | ND | ND | ND | ND | ND | ND | ND | ND |
| Pentachlorophenol | 0.48 (0.02-2.24) | 87.50 | 20.25 | 39.50 | 2.00 | 2.23 | 32.00 | 23.50 | 38 | 6.60 | 4.10 |
| 3-Chlorocatechol | 0.76 (0.01-4.71) | ND | 2.55 | ND | ND | ND | ND | ND | ND | ND | ND |
| 4-Chlorocatechol | 0.15 (0.03-0.54) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4-Dichlorocatechol | 0.40 (0.02-2.30) | ND | 1.98 | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,5-Dichlorocatechol | 1.05 (0.03-5.60) | ND | 3.75 | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,6-Dichlorocatechol | 0.71 (0.03-3.70) | ND | 3.49 | ND | ND | ND | ND | ND | 5.1 | ND | 6.88 |
| 4,5-Dichlorocatechol | 0.27 (0.02-1.39) | ND | ND | ND | ND | ND | ND | ND | 9.4 | ND | ND |
| 3,4,5-Trichlorocatechol | 0.36 (0.04-3.30) | ND | ND | ND | ND | ND | ND | ND | 7.5 | ND | ND |
| 3,4,6-Trichlorocatechol | 0.36 (0.02-2.90) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachlorocatechol | 0.61 (0.04-4.94) | ND | 0.57 | ND | ND | ND | ND | ND | 1.8 | ND | ND |
| 4-Chloroguaiacol | 0.44 (0.02-2.70) | ND | 2.66 | 1.22 | ND | ND | ND | 4.55 | 8.5 | 3.45 | 23.00 |
| 5-Chloroguaiacol | 0.23 (0.03-0.80) | ND | 10.90 | 63.50 | ND | 25.43 | ND | 76.50 | 200 | 5.60 | 605.00 |
| 6-Chloroguaiacol | 0.16 (0.03-0.64) | ND | 1.61 | 0.82 | ND | ND | ND | ND | 0.43 | ND | 0.71 |
| 3,4-Dichloroguaiacol | 0.70 (0.04-3.30) | ND | 1.66 | 6.70 | ND | ND | ND | 11.60 | 160 | 20.00 | 11.50 |
| 4,5-Dichloroguaiacol | 0.62 (0.02-3.30) | ND | 6.88 | 580.00 | ND | ND | ND | 31.00 | 1400 | 3.66 | 44.50 |
| 4,6-Dichloroguaiacol | 0.30 (0.01-1.60) | ND | 1.69 | 10.55 | ND | ND | ND | 5.75 | 55 | 2.05 | 1.70 |
| 3,4,5-Trichloroguaiacol | 0.64 (0.03-5.00) | ND | 28.50 | 441.00 | ND | ND | ND | 48.00 | 240 | 5.08 | 11.65 |
| 3,4,6-Trichloroguaiacol | 0.76 (0.06-5.40) | ND | 2.78 | 22.05 | ND | ND | ND | ND | 100 | ND | ND |
| 4,5,6-Trichloroguaiacol | 0.27 (0.02-1.90) | ND | 10.41 | 126.50 | ND | ND | ND | 6.30 | 670 | ND | 1.40 |
| 3,4,5,6-Tetrachloroguaiacol | 0.24 (0.03-1.40) | 3.10 | 16.95 | 47.00 | ND | 1.43 | 27.50 | 16.50 | 180 | 7.85 | 8.50 |
| 3-Chlorosyringol | 0.23 (0.04-0.89) | ND | 0.81 | 0.88 | ND | ND | ND | ND | 1.7 | 1.58 | 0.91 |
| 3,5-Dichlorosyringol | 0.79 (0.03-4.66) | ND | 5.96 | ND | ND | ND | ND | ND | ND | ND | 3.68 |
| 3,4,5-Trichlorosyringol | 0.36 (0.04-1.66) | ND | 2.00 | ND | ND | ND | 4.70 | ND | 13 | ND | ND |
| 2-Chlorosyringaldehyde | 0.19 (0.01-0.75) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,6-Dichlorosyringaldehyde | 0.41 (0.01-1.98) | ND | 1.73 | ND | ND | ND | 5.55 | 1.43 | ND | 2.50 | 2.35 |
| 5-Chlorovanillin | 1.52 (0.01-1.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 6-Chlorovanillin | 1.65 (0.02-3.00) | ND | ND | ND | ND | ND | ND | 62.50 | 44 | ND | 8.75 |
| 5,6-Dichlorovanillin | 0.28 (0.02-1.20) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 4,5 -Dichloroveratrole | 1.43 (0.02-6.60) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5-Trichloroveratrole | 0.68 (0.05-3.20) | ND | 3.18 | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,6-Trichloroveratrole | 0.44 (0.04-2.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachloroveratrole | 0.47 (0.05-2.13) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |

Table 33. Measured chlorophenolic concentrations in peamouth, flounder, and whitefish bile from the Fraser basin, 1995. Detection limits shown as the average (range) and concentrations shown as mean in $\mathrm{ng} / \mathrm{mL}$ bile.

|  | Peamouth |  |  |  |  |  |  | Starry Flounder |  | Whitefish |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Detect Limits | Hansard | Woodpecker | Main Arm | North Arm | N. Thompson | Thompson | Main Arm | North Arm | Hansard | Woodpecker | Marguerite | N. Thompson | Thompson |
| 4-Chlorophenol | 3.05 (0.48-18.00) | ND | ND | ND | ND | ND | ND | 0.8 | ND | 5.4 | 9.4 | 7.0 | ND | 0.4 |
| 2,3-Dichlorophenol | 1.26 (0.49-4.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | 2.8 | ND |
| 2,4/2,5-Dichlorophenol | 1.24 (0.40-5.10) | 0.2 | 4.7 | 1.8 | 7.0 | 4.6 | 3.0 | 9.6 | 7.8 | 33.0 | 45.0 | 9.8 | 195.0 | 38.0 |
| 2,6-Dichlorophenol | 1.23 (0.42-4.50) | ND | 0.0 | ND | ND | ND | ND | 1.8 | ND | 2.3 | 2.1 | 0.8 | 5.0 | ND |
| 3,4-Dichlorophenol | 3.20 (1.20-16.00) | ND | ND | ND | ND | ND | 2.9 | ND | ND | ND | ND | ND | 0.8 | 1.8 |
| 3,5-Dichlorophenol | 1.79 (0.63-6.60) | ND | ND | 1.3 | ND | ND | ND | 1.5 | ND | ND | 5.0 | ND | 0.8 | 6.3 |
| 2,3,4-Trichlorophenol | 1.31 (0.42-4.80) | ND | ND | ND | ND | ND | 0.1 | ND | ND | ND | ND | ND | 14.7 | 20.5 |
| 2,3,5-Trichlorophenol | 1.81 (0.74-6.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | 0.2 | ND | 3.1 | 1.9 |
| 2,3,6-Trichlorophenol | 1.63 (0.53-5.80) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,4,5-Trichlorophenol | 1.50 (0.60-6.40) | ND | 0.2 | ND | ND | ND | ND | 0.5 | ND | ND | 2.0 | 0.8 | 2.0 | 1.1 |
| 2,4,6-Trichlorophenol | 1.09 (0.37-4.50) | 3.6 | 13.0 | 46.0 | 89.0 | 4.3 | 2.9 | 11.4 | 13.5 | 10.0 | 17.5 | 84.0 | 18.0 | 2.9 |
| 3,4,5-Trichlorophenol | 2.17 (0.68-7.47) | ND | ND | ND | ND | ND | ND | ND | ND | 5.6 | 20.5 | 0.8 | 8.6 | 9.9 |
| 2,3,4,5-Tetrachlorophenol | 1.96 (0.64-7.40) | ND | ND | ND | ND | ND | ND | ND | ND | ND | 3.5 | ND | ND | ND |
| 2,3,4,6-Tetrachlorophenol | 1.98 (0.61-8.40) | ND | 7.1 | 11.8 | 16.9 | ND | ND | ND | ND | ND | 4.6 | 9.6 | 2.4 | ND |
| 2,3,5,6-Tetrachlorophenol | 2.63 (0.45-11.00) | ND | ND | 2.0 | 2.2 | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| Pentachlorophenol | 1.99 (0.65-7.90) | 13.8 | 23.0 | 52.0 | 63.0 | 2.5 | 1.9 | 19.3 | 60.0 | 11.0 | 26.0 | 18.0 | 15.0 | 3.5 |
| 3-Chlorocatechol | 2.39 (0.82-6.80) | ND | 1.5 | ND | ND | ND | ND | 1.5 | ND | ND | 0.2 | 0.6 | 3.7 | 3.3 |
| 4-Chlorocatechol | 1.53 (0.43-4.94) | ND | 0.1 | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4-Dichlorocatechol | 2.20 (0.72-6.13) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,5-Dichlorocatechol | 2.05 (0.49-6.80) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | 15.7 | ND |
| 3,6-Dichlorocatechol | 1.50 (0.47-5.29) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 4,5-Dichlorocatechol | 2.15 (0.51-7.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5-Trichlorocatechol | 1.88 (0.02-6.20) | ND | 0.5 | ND | ND | ND | ND | ND | ND | ND | ND | ND | 2.1 | 7.9 |
| 3,4,6-Trichlorocatechol | 1.92 (0.48-8.50) | ND | 0.9 | ND | ND | ND | ND | ND | ND | ND | ND | ND | 2.1 | 5.9 |
| 3,4,5,6-Tetrachlorocatechol | 1.33 (0.45-3.09) | ND | 1.6 | ND | ND | ND | 0.2 | ND | ND | ND | ND | ND | ND | ND |
| 4-Chloroguaiacol | 2.02 (0.58-7.20) | ND | 4.1 | ND | ND | ND | 3.1 | 1.7 | ND | ND | 5.9 | 5.6 | 0.1 | 29.0 |
| 5-Chloroguaiacol | 1.43 (0.52-3.60) | ND | 46.0 | 11.9 | 10.8 | ND | 245.0 | 20.0 | 4.5 | 1.4 | 140.0 | 92.0 | 14.0 | 855.0 |
| 6-Chloroguaiacol | 0.67 (0.14-1.76) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | 2.1 |
| 3,4-Dichloroguaiacol | 1.52 (0.45-4.40) | ND | ND | ND | 0.1 | ND | ND | 1.7 | ND | ND | 28.5 | 19.9 | 20.5 | 9.5 |
| 4,5-Dichloroguaiacol | 1.90 (0.73-6.80) | ND | 9.9 | 9.1 | 13.6 | ND | 17.5 | 13.5 | 2.7 | ND | 40.0 | 138.0 | ND | 72.5 |
| 4,6-Dichloroguaiacol | 1.14 (0.30-4.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | 5.3 | 3.3 | 2.5 | ND |
| 3,4,5-Trichloroguaiacol | 3.05 (0.93-16.00) | ND | ND | ND | ND | ND | ND | 4.8 | ND | ND | 2.8 | 36.7 | ND | ND |
| 3,4,6-Trichloroguaiacol | 3.14 (0.97-16.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | 0.9 | ND | ND |
| 4,5,6-Trichloroguaiacol | 1.63 (0.70-5.52) | 0.1 | ND | ND | 1.1 | ND | ND | 3.3 | ND | ND | 2.3 | 17.3 | ND | 0.4 |
| 3,4,5,6-Tetrachloroguaiacol | 1.46 (0.44-3.38) | 3.6 | 4.8 | ND | ND | 0.6 | ND | 2.2 | ND | 6.2 | 13.5 | 21.9 | 11.5 | 3.9 |
| 3-Chlorosyringol | 2.09 (0.61-9.20) | ND | ND | ND | ND | ND | ND | ND | 2.5 | ND | 3.1 | ND | ND | 1.6 |
| 3,5-Dichlorosyringol | 1.47 (0.52-5.20) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5-Trichlorosyringol | 1.78 (0.42-5.75) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | 0.1 | ND |
| 2-Chlorosyringaldehyde | 1.96 (0.50-7.70) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 2,6-Dichlorosyringaldehyde | 1.53 (0.53-5.80) | ND | 1.0 | ND | ND | ND | ND | ND | ND | ND | ND | ND | 3.6 | 9.9 |
| 5-Chlorovanillin | 1.64 (0.52-6.60) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 6-Chlorovanillin | 2.20 (0.37-11.00) | ND | 4.1 | ND | ND | ND | ND | ND | ND | ND | 74.5 | 24.3 | ND | 9.7 |
| 5,6-Dichlorovanillin | 3.81 (1.00-11.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 4,5-Dichloroveratrole | 0.99 (0.37-2.80) | ND | ND | ND | ND | ND | ND | ND | ND | 5.2 | ND | ND | 1.2 | 1.5 |
| 3,4,5-Trichloroveratrole | 1.32 (0.44-4.90) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,6-Trichloroveratrole | 2.46 (1.20-8.28) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
| 3,4,5,6-Tetrachloroveratrole | 1.67 (0.21-5.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |

Table 34. Comparison of maximal chlorophenol concentrations relative to established chlorophenol BC criteria for fish tainting.

| Criterion Level <br> $\boldsymbol{\mu g} / \mathbf{g}$ | Chlorophenol <br> Congener | Maximum <br> Measured |
| :---: | :---: | :---: |
| 10 | 2-monochlorophenol | $n m$ |
| 20 | 3-monochlorophenol | $n m$ |
| 40 | 4-monochlorophenol | 0.12 |
| 80 | 2,3-dichlorophenol | 0.13 |
| 0.2 | 2,4-dichlorophenol | $0.55^{\star}$ |
| 20 | 2,5-dichlorophenol | $0.55^{\star}$ |
| 30 | 2,6-dichlorophenol | 0.09 |
| 50 | 2,4,6-dichlorophenol | 0.09 |
| 20 | Pentachlorophenol | 0.15 |

* 2,4-2,5-dichlorophenol reported as a combined total in analyses.
$n m$ - not measured

Table 35. Resin acids measured in fish bile from the Fraser basin, 1994/1995.

```
Abietic Acid
Dehydroabietic Acid
Dehydroisopimaric Acid
Isopimaric Acid
Neoabietic Acid
Palustric Acid
Pimaric Acid
Sandaracopimaric Acid
12,14 - Dichlorodehydroabietic Acid
12 - Monochlorodehydroabietic Acid
```

Table 36. Measured concentrations of resin acids in peamouth and whitefish bile from reaches in the Fraser basin, 1994. Table shows average ( $\mu \mathrm{g} / \mathrm{mL}$ ) and range of values.

| Number of Analyses Total Resin Acids Total Chlorinated | Peamouth |  |  |  | Whitefish |
| :---: | :---: | :---: | :---: | :---: | :---: |
|  | Hansard | Woodpecker | North Thompson | Thompson | Woodpecker |
|  | $\begin{array}{\|c\|} \hline 2 \\ 424.94(317.57-532.30) \\ <0.016 \end{array}$ | $\begin{gathered} 2 \\ 11.80(11.42-12.18) \\ 0.055 \end{gathered}$ | $\begin{gathered} 3 \\ 7.12(1.28-13.02) \\ 0.007 \end{gathered}$ | $\begin{gathered} 2 \\ 0.73(0.51-0.94) \\ <0.007 \end{gathered}$ | $\begin{gathered} 2 \\ 16.51(16.32-16.70) \\ 0.161 \end{gathered}$ |
| Abietic | 145.00 (100.00-190.00) | 4.25 (3.30-5.20) | 3.88 (0.25-7.60) | 0.12 (0.06-0.18) | 4.70 (4.00-5.40) |
| Dehydroabieitic | 79.00 (72.00-86.00) | 2.50 (2.00-3.00) | 1.04 (0.23-1.90) | <0.46 | 1.50 (1.40-1.60) |
| Dehydroisopimaric | $<0.020$ | 0.051 (0.044-0.057) | 0.004 (0.002-0.005) | <0.004 | 0.075 (0.053-0.097) |
| Isopimaric | 8.75 (5.50-12.00) | 3.20 (2.60-3.80) | 1.15 (0.60-1.70) | 0.18 (<0.20-0.26) | 6.40 (5.80-7.00) |
| Neoabietic | 1.23 (0.36-2.10) | 0.06 (0.06-0.06) | 0.06 (0.003-0.12) | <0.003 | 0.02 (<0.003-0.04) |
| Palustric | 59.00 (29.00-89.00) | 0.52 (0.40-0.64) | 0.57 (0.04-1.10) | 0.04 (0.02-0.07) | 0.45 (0.35-0.55) |
| Pimaric | 1.95 (0.71-3.20) | 0.88 (0.83-0.92) | 0.11 (0.05-0.17) | 0.10 (0.05-0.15) | 3.05 (2.60-3.50) |
| Sandaracopimaric | 130.00 (110.00-150.00) | 0.38 (0.26-0.50) | 0.32 (0.10-0.54) | 0.04 (<0.08-0.05) | 0.31 (0.14-0.48) |
| 12,14-ChloroDHAA | <0.033 | 0.01 (0.01-0.01) | 0.004 (0.003-0.005) | <0.004 | 0.04 (0.03-0.04) |
| 12-ChloroDHAA | $<0.016$ | 0.04 (0.04-0.05) | $0.004(0.004-0.004)$ | <0,004 | $0.11(0.10-0.12)$ |

Table 37. Measured concentrations of resin acids in peamouth, whitefish, and flounder bile from reaches in the Fraser basin, 1995. Table shows average $(\mu \mathrm{g} / \mathrm{mL})$ and range of values.


|  | Whitefish |  | Flounder |  |
| :---: | :---: | :---: | :---: | :---: |
|  | Hansard | Woodpecker | Main Arm | North Arm |
| Number of | 2 | 2 | 3 | 2 |
| Analyses |  |  |  |  |
| Total Resin Acids | $36.75(20.34-53.15)$ | $25.36(16.55-34.17)$ | $15.66(1.99-29.21)$ | $4.68(3.73-5.62)$ |
| Total Chlorinated | ND | $0.13(0.10-0.17)$ | $0.01(<0.01-0.01)$ | $0.01(0.01-0.01)$ |
|  |  |  |  |  |
| Abietic | $18.00(9.00-27.00)$ | $15.30(6.60-24.00)$ | $8.40(0.68-16.00)$ | $1.55(1.10-2.00)$ |
| Dehydroabietic | $4.80(4.70-4.90)$ | $1.35(1.30-1.40)$ | $1.86(0.480-3.20)$ | $0.71(0.64-0.79)$ |
| Dehydroisopimaric | $0.01(<0.01-0.02)$ | $0.06(0.05-0.07)$ | $<0.012$ | $<0.04$ |
| Isopimaric | $4.80(2.70-6.90)$ | $4.95(4.80-5.10)$ | $1.87(0.43-3.30)$ | $0.70(0.69-0.72)$ |
| Neoabietic | $0.69(0.47-0.90)$ | $0.33(0.11-0.56)$ | $0.55(<0.01-1.10)$ | $0.03(<0.01-0.06)$ |
| Palustric | $6.45(1.90-11.00)$ | $0.64(0.54-0.73)$ | $1.40(0.19-2.60)$ | $0.21(0.18-0.24)$ |
| Pimaric | $0.50(0.45-0.55)$ | $2.30(2.20-2.40)$ | $0.12(0.04-0.20)$ | $0.11(0.10-0.11)$ |
| Sandaracopimaric | $1.50(1.10-1.90)$ | $0.29(0.26-0.33)$ | $1.44(0.08-2.80)$ | $1.35(1.00-1.70)$ |
| 12,14-ChloroDHAA | $<0.007$ | $0.10(0.07-0.13)$ | $0.01(<0.01-0.01)$ | $0.01(0.01-0.01)$ |
| 12- Chloro DHAA | $<0.001$ | $0.03(0.02-0.04)$ | $<0.01$ | $<0.01$ |

Table 38. Non-ortho and coplanar PCB congeners analyzed from fish tissues in the Fraser basin, 1994/1995. Numbers are IUPAC congener designations, and co-eluting congener sets are shown as, for example, $8 / 5$ or 24/27.

| PCB 8/5 | PCB 49 | PCB 118 | PCB 171 | PCB 198 |
| :--- | :--- | :--- | :--- | :--- |
| PCB 15 | PCB 52 | PCB 128 | PCB 172 | PCB 199 |
| PCB 16/32 | PCB 56/60 | PCB 129 | PCB 174 | PCB 201 |
| PCB 17 | PCB 66 | PCB 130 | PCB 175 | PCB 205 |
| PCB 18 | PCB 70/76 | PCB 131 | PCB 176 | PCB 206 |
| PCB 19 | PCB 74 | PCB 134 | PCB 177 | PCB 207 |
| PCB 22 | PCB 83 136 | PCB 178 | PCB 208 |  |
| PCB 24/27 | PCB 84/89 | PCB 137 | PCB 179 | PCB 209 |
| PCB 25 | PCB 85 | PCB 138/163/164 | PCB 180 |  |
| PCB 26 | PCB 87 | PCB 141 | PCB 183 | Coplanar Congeners: |
| PCB 31/28 | PCB 91 | PCB 144/135 | PCB 185 | PCB 77 |
| PCB 33 | PCB 95 | PCB 146 | PCB 187/182 | PCB 126 |
| PCB 40 | PCB 97 | PCB 149 | PCB 189 |  |
| PCB 41/71/64 | PCB 99 169 | PCB 151 | PCB 191 |  |
| PCB 42 | PCB 101/90 44 | PCB 153 | PCB 193 |  |
| PCB 45 | PCB 105 | PCB 156 | PCB 194 |  |
| PCB 46 | PCB 107 | PCB 157 | PCB 195 |  |
| PCB 47/48 | PCB 114 | PCB 158 | PCB 196/203 |  |

Table 39. Total PCB concentrations in fish muscle tissue from the Fraser basin, 1994 and 1995. Totals were calculated as the sum of 84 measured congeners, where non-detects were assigned a value equal to $1 / 2$ the sample detection limit. Concentrations are shown as the average and range of values in $\mathrm{ng} / \mathrm{g}$ wet weight.

| Peamouth |  |  |
| :---: | :---: | :---: |
|  | n | 1994 |
| Nechako | 4 | 3.97 (3.14-6.05) |
| Hansard | 4 | 2.76 (2.16-3.48) |
| Woodpecker | 4 | 5.39 (2.47-11.62) |
| Marguerite | 4 | 16.63 (5.13-27.95) |
| Agassiz | 4 | 27.42 (14.11-42.02) |
| Mission | 5 | 24.57 (10.65-57.77) |
| Barnston | 5 | 11.01 (8.92-13.03) |
| Main | 5 | 15.06 (10.41-23.50) |
| North Arm | 5 | 15.84 (10.30-23.20) |
| N. Thompson | 4 | 5.52 (3.80-6.37) |
| Thompson | 4 | 3.95 (2.45-4.96) |


| $\mathbf{n}$ | $\mathbf{1 9 9 5}$ |
| :---: | :---: |
| 4 | $2.02(1.89-2.25)$ |
| 4 | $2.19(1.16-4.19)$ |
| 4 | $1.54(0.93-2.16)$ |
| 4 | $1.80(0.91-2.41)$ |
| 4 | $10.50(9.11-11.81)$ |
|  | no samples |
|  | no samples |
| 4 | $13.07(4.86-22.21)$ |
| 4 | $11.58(10.55-13.00)$ |
| 4 | $4.30(2.25-6.20)$ |
| 4 | $3.40(2.98-3.99)$ |

## Flounder

|  | $\mathbf{n}$ | $\mathbf{1 9 9 4}$ |
| :--- | :---: | :---: |
| Main | 5 | $5.46(4.34-7.13)$ |
| North Arm | 5 | $11.48(7.66-19.99)$ |


| $\mathbf{n}$ | $\mathbf{1 9 9 5}$ |
| :---: | :---: |
| 4 | $2.90(2.03-4.00)$ |
| 4 | $5.33(3.04-8.89)$ |

## Whitefish

|  | $\mathbf{n}$ | $\mathbf{1 9 9 4}$ |
| :--- | :--- | :---: |
| Nechako | 4 | $4.29(2.20-10.15)$ |
| Hansard | 4 | $2.80(2.24-3.63)$ |
| Woodpecker | 4 | $4.03(1.65-7.56)$ |
| Marguerite | 4 | $5.19(3.51-6.82)$ |
| Agassiz | 4 | $30.83(10.86-82.44)$ |
| N. Thompson | 4 | $12.16(9.57-14.57)$ |
| Thompson | 4 | $10.74(4.31-22.36)$ |


| $\mathbf{n}$ | $\mathbf{1 9 9 5}$ |
| :--- | :---: |
| 4 | $2.36(1.47-4.08)$ |
| 4 | $2.58(1.70-3.67)$ |
| 4 | $2.20(0.95-3.72)$ |
| 2 | $5.02(4.24-5.80)$ |
| 4 | $23.81(16.64-36.18)$ |
| 4 | $8.68(4.52-18.53)$ |
| 4 | $7.16(6.44-8.05)$ |

Table 40. Total PCB concentrations in fish liver tissue from the Fraser basin, 1994 and 1995. Totals were calculated as the sum of 84 measured congeners, where non-detects were assigned a value equal to $1 / 2$ the sample detection limit. Concentrations are shown as the average and range of values in $\mathrm{ng} / \mathrm{g}$ wet weight.

## Peamouth

|  | $\mathbf{n}$ | $\mathbf{1 9 9 4}$ |
| :--- | :--- | :---: |
| Nechako | 2 | $24.48(19.66-29.31)$ |
| Hansard | 2 | $23.96(19.62-28.29)$ |
| Woodpecker | 1 | 24.69 |
| Marguerite | 1 | 22.8 |
| Agassiz | 3 | $78.65(21.98-143.44)$ |
| Mission | 1 | 214.43 |
| Barnston | 1 | 162.07 |
| Main | 2 | $181.04(156.46-205.63)$ |
| North Arm | 2 | $239.09(229.54-248.63)$ |
| N. Thompson | 2 | $27.19(26.17-28.21)$ |
| Thompson | 2 | $36.38(33.78-38.98)$ |


| $\mathbf{n}$ | $\mathbf{1 9 9 5}$ |
| :---: | :---: |
| 2 | $12.15(10.41-13.89)$ |
| 2 | $13.81(12.08-15.56)$ |
| 2 | $14.40(13.50-15.30)$ |
| 1 | 11.6 |
| 3 | $92.67(70.53-107.29)$ |
|  | no samples |
|  | no samples |
| 2 | $165.58(158.80-172.36)$ |
| 3 | $238.86(183.49-271.14)$ |
| 2 | $21.37(21.02-21.72)$ |
| 3 | $26.55(23.16-31.52)$ |

## Flounder

|  | n | $\mathbf{1 9 9 4}$ |
| :--- | :---: | :---: |
| Main | 1 | 52.93 |
| North Arm | 1 | 90.98 |


| $\mathbf{n}$ | $\mathbf{1 9 9 5}$ |
| :---: | :---: |
| 1 | 40.01 |
| 1 | 52.35 |

## Whitefish

|  | $\mathbf{n}$ | $\mathbf{1 9 9 4}$ |
| :--- | :--- | :---: |
| Nechako | 2 | $5.10(5.03-5.17)$ |
| Hansard | 1 | 3.22 |
| Woodpecker | 2 | $4.33(3.86-4.80)$ |
| Marguerite | 1 | 7.52 |
| Agassiz | 1 | 21.97 |
| N. Thompson | 2 | $5.16(4.86-5.46)$ |
| Thompson | 2 | $7.33(4.00-10.66)$ |


| $\mathbf{n}$ | $\mathbf{1 9 9 5}$ |
| :---: | :---: |
| 2 | $7.72(6.55-8.90)$ |
| 2 | $5.79(5.48-6.09)$ |
| 2 | $4.10(3.96-4.23)$ |
| 4 | $6.75(6.75-6.75)$ |
| 2 | $8.89(8.70-9.08)$ |
| 2 | $7.36(6.46-8.27)$ |
| 2 | $12.33(8.46-16.19)$ |

Table 41. Mean and ranges of coplanar PCB concentrations in fish muscle tissues in the Fraser basin, in $\mathrm{pg} / \mathrm{g}$ wet weight.

| Peamouth |  | n | PCB 77 | PCB 126 | PCB 169 |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 4 4 | $\begin{gathered} \hline 1.3(<1.2-1.7) \\ 0.7(0.6-0.8) \end{gathered}$ | $\begin{gathered} \hline \text { ND }(<0.6) \\ \text { ND }(<0.15) \end{gathered}$ | $\begin{aligned} & \text { ND (<0.9) } \\ & \text { ND (<0.2) } \end{aligned}$ |
| Hansard | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 4 4 | $\begin{gathered} \hline 7.1(1.7-21.0) \\ 1.4(0.7-3.0) \end{gathered}$ | $\begin{aligned} & \hline 0.6 \text { (0.5-0.9) } \\ & 0.4(0.2-0.8) \end{aligned}$ | $\begin{gathered} \hline 0.3(0.2-0.3) \\ 0.1(<0.2-0.2) \end{gathered}$ |
| Woodpecker | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 4 4 | $\begin{gathered} 14.8 \text { (2.3-48.0) } \\ 1.7 \text { (1.1-2.5) } \end{gathered}$ | $\begin{aligned} & \hline 0.7 \text { (0.5-1.3) } \\ & 0.4(0.2-0.5) \end{aligned}$ | $\begin{gathered} \hline 0.2(<0.2-0.2) \\ \text { ND }(<0.2) \end{gathered}$ |
| Marguerite | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 4 | $\begin{gathered} 13.1 \text { (5.3-21.0) } \\ 2.8 \text { (0.9-6.5) } \end{gathered}$ | $\begin{aligned} & \hline 2.6 \text { (1.1-4.8) } \\ & 0.4(0.2-0.4) \end{aligned}$ | $\begin{gathered} \hline 0.4(0.3-0.5) \\ N D(<0.3) \end{gathered}$ |
| Agassiz | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & 4 \\ & 4 \end{aligned}$ | $\begin{gathered} \hline 32.3 \text { (18.0-62.0) } \\ 8.8 \text { (6.2-13.0) } \end{gathered}$ | $\begin{aligned} & 3.8 \text { (2.6-4.9) } \\ & 1.3 \text { (1.1-1.4) } \end{aligned}$ | $\begin{gathered} \hline \mathrm{ND}(<1.1) \\ 0.2(<0.2-0.2) \end{gathered}$ |
| Mission | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ |  | $\begin{gathered} 34.2 \text { (13.0-73.0) } \\ \text { - No Sample - } \end{gathered}$ | 3.7 (1.9-6.4) <br> - No Sample - | ND (<0.6) <br> - No Sample - |
| Barnston | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 5 | $14.5 \text { (11.0-18.0) }$ <br> - No Sample - | $2.1 \text { (1.9-2.5) }$ <br> - No Sample - | ND (<0.62) <br> - No Sample - |
| Main Arm | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 5 4 | $\begin{gathered} \hline 16.7(11.0-27.0) \\ 10.9(7.0-15.5) \end{gathered}$ | $\begin{aligned} & \hline 2.5(1.6-4.2) \\ & 1.6(0.9-2.5) \end{aligned}$ | $\begin{aligned} & \hline 0.6(<1.3-0.7) \\ & 0.2(<0.3-0.4) \end{aligned}$ |
| North Arm | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 5 4 | $\begin{aligned} & \hline 19.5 \text { (13.0-29.0) } \\ & 18.0 \text { (15.0-23.0) } \end{aligned}$ | $\begin{aligned} & \hline 2.0 \text { (1.2-3.0) } \\ & 1.6 \text { (1.3-2.0) } \end{aligned}$ | $\begin{aligned} & \hline 0.5(<0.9-0.6) \\ & 0.2(<0.3-0.3) \end{aligned}$ |
| N. Thompson | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & 4 \\ & 4 \end{aligned}$ | $\begin{aligned} & \hline 5.0 \text { (3.2-6.9) } \\ & 4.3 \text { (2.9-6.7) } \end{aligned}$ | $\begin{aligned} & 1.0 \text { (0.6-1.1) } \\ & 0.8(0.5-1.0) \end{aligned}$ | $\begin{gathered} \hline 0.3(0.2-0.4) \\ 0.2(<0.3-0.3) \end{gathered}$ |
| Thompson | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | 4 4 | $\begin{aligned} & 5.1 \text { (3.2-7.8) } \\ & 4.3 \text { (3.9-4.8) } \end{aligned}$ | $\begin{aligned} & \hline 0.8(0.5-1.1) \\ & 0.7(0.7-0.8) \end{aligned}$ | $\begin{gathered} \hline 0.3(0.2-0.5) \\ N D(<0.3) \end{gathered}$ |


| Flounder | $\mathbf{n}$ | PCB 77 | PCB 126 | PCB 169 |  |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Main Arm | $\mathbf{1 9 9 4}$ | 5 | $1.9(<1.1-3.3)$ | ND $(<0.6)$ | ND $(<0.3)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $1.5(1.2-1.9)$ | $0.4(0.3-0.5)$ | ND $(<0.2)$ |
| North Arm | $\mathbf{1 9 9 4}$ | 5 | $9.6(3.9-21.0)$ | $1.4(<3.3-2.2)$ | ND $(<0.7)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $3.9(1.8-6.3)$ | $0.6(0.3-0.9)$ | ND $(<0.15)$ |


| Whitefish | $\mathbf{n}$ | PCB 77 | PCB 126 | PCB 169 |  |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Nechako | $\mathbf{1 9 9 4}$ | 4 | $2.2(1.7-3.4)$ | $0.9(<0.6-1.6)$ | $1.2(<1.1-1.9)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $0.9(0.7-1.0)$ | $0.3(<0.2-0.5)$ | $0.7(0.6-0.8)$ |
| Hansard | $\mathbf{1 9 9 4}$ | 4 | $1.0(0.7-1.3)$ | $0.3(0.2-0.4)$ | $0.4(0.4-0.5)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $1.0(0.8-1.5)$ | $0.4(0.4-0.5)$ | $0.3(0.2-0.5)$ |
| Woodpecker | $\mathbf{1 9 9 4}$ | 4 | $1.9(1.6-2.1)$ | $0.6(0.5-0.7)$ | $0.5(0.4-0.6)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $2.2(1.1-3.2)$ | $0.4(0.2-0.6)$ | $0.3(0.3-0.3)$ |
| Marguerite | $\mathbf{1 9 9 4}$ | 4 | $9.0(5.9-17.0)$ | $1.0(0.8-1.4)$ | $0.2(<0.2-0.3)$ |
|  | $\mathbf{1 9 9 5}$ | 3 | $3.7(3.3-4.4)$ | $0.7(0.6-0.9)$ | $0.2(0.2-0.2)$ |
| Agassiz | $\mathbf{1 9 9 4}$ | 4 | $9.9(7.8-14.0)$ | $2.7(1.6-3.5)$ | $0.6(<0.9-1.1)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $15.8(10.0-28.0)$ | $2.4(1.8-3.1)$ | $0.5(0.4-0.6)$ |
| N. Thompson | $\mathbf{1 9 9 4}$ | 4 | $11.7(9.8-15.0)$ | $2.0(1.7-2.4)$ | $0.5(0.4-0.7)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $7.7(1.9-19.0)$ | $1.2(0.7-2.1)$ | $0.8(0.5-1.0)$ |
| Thompson | $\mathbf{1 9 9 4}$ | 4 | $9.0(5.0-16.0)$ | $2.4(0.8-6.8)$ | $0.6(<0.8-1.7)$ |
|  | $\mathbf{1 9 9 5}$ | 4 | $5.9(4.7-7.2)$ | $1.0(0.9-1.1)$ | $0.3(0.3-0.4)$ |

Table 42. Mean and ranges of coplanar PCB concentrations in fish liver tissues in the Fraser basin, in pg/g wet weight.

| Peamouth | $\mathbf{n}$ | PCB 77 | PCB 126 | PCB 169 |  |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Nechako | $\mathbf{1 9 9 4}$ | 2 | $11.4(9.9-13.0)$ | $4.4(3.0-5.8)$ | $3.3(2.8-3.7)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $9.0(7.9-10.0)$ | $1.9(1.8-2.0)$ | $2.5(2.3-2.7)$ |
| Hansard | $\mathbf{1 9 9 4}$ | 2 | $29.0(17.0-41.0)$ | $4.2(3.7-4.6)$ | $1.3(1.2-1.3)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $8.5(7.8-9.2)$ | $2.6(2.4-2.8)$ | $1.3(1.0-1.5)$ |
| Woodpecker | $\mathbf{1 9 9 4}$ | 1 | 16.5 | 2.9 | ND (<1) |
|  | $\mathbf{1 9 9 5}$ | 2 | $16.0(16.0-16.0)$ | $3.4(3.2-3.5)$ | $1.5(1.3-1.7)$ |
| Marguerite | $\mathbf{1 9 9 4}$ | 1 | 18 | 4.2 | 0.9 |
|  | $\mathbf{1 9 9 5}$ | 1 | 19 | 3.1 | 1.2 |
| Agassiz | $\mathbf{1 9 9 4}$ | 2 | $107.0(84.0-130.0)$ | $15.5(11-18)$ | $2.1(1.4-2.8)$ |
|  | $\mathbf{1 9 9 5}$ | 3 | $75.7(61.0-100.0)$ | $10.8(8.3-14.0)$ | $1.3(1.1-1.5)$ |
| Mission | $\mathbf{1 9 9 4}$ | 1 | 340 | 36 | 4.9 |
|  | $\mathbf{1 9 9 5}$ | - | - No Sample - | - No Sample - | - No Sample - |
| Barnston | $\mathbf{1 9 9 4}$ | 1 | 180 | 34 | 3.2 |
|  | $\mathbf{1 9 9 5}$ | - | - No Sample - | - No Sample - | - No Sample - |
| Main Arm | $\mathbf{1 9 9 4}$ | 1 | 420 | 48 | 5.1 |
|  | $\mathbf{1 9 9 5}$ | 2 | $185.0(140.0-230.0)$ | $25.5(25.0-26.0)$ | $3.2(2.7-3.6)$ |
| North Arm | $\mathbf{1 9 9 4}$ | 2 | $370.0(330.0-410.0)$ | $39.0(32.0-46.0)$ | $4.3(3.2-5.4)$ |
|  | $\mathbf{1 9 9 5}$ | 3 | $350.0(240.0-430.0)$ | $34.0(26.0-38.0)$ | $3.3(3.1-3.5)$ |
| N. Thompson | $\mathbf{1 9 9 4}$ | 2 | $21.8(21.0-22.5)$ | $4.8(4.8-4.9)$ | $1.3(1.2-1.4)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $19.8(19.5-20.0)$ | $4.1(4.0-4.3)$ | $1.4(1.2-1.5)$ |
| Thompson | $\mathbf{1 9 9 4}$ | 2 | $43.5(40.0-47.0)$ | $8.4(7.1-9.7)$ | $2.3(2.0-2.6)$ |
|  | $\mathbf{1 9 9 5}$ | 3 | $33.5(32.0-36.0)$ | $5.1(4.8-5.6)$ | $1.8(1.6-1.9)$ |


| Flounder | $\mathbf{n}$ | PCB 77 | PCB 126 | PCB 169 |  |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Main Arm | $\mathbf{1 9 9 4}$ | 1 | 31 | 9.4 | 4.1 |
|  | $\mathbf{1 9 9 5}$ | 1 | 24 | 6.1 | 1.3 |
| North Arm | $\mathbf{1 9 9 4}$ | 1 | 74 | 18 | ND (<2.9) |
|  | $\mathbf{1 9 9 5}$ | 1 | 45 | 7.2 | 1.3 |


| Whitefish | $\mathbf{n}$ | PCB 77 | PCB 126 | PCB 169 |  |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Nechako | $\mathbf{1 9 9 4}$ | 2 | $1.0(0.6-1.4)$ | $0.6(0.6-0.6)$ | $\mathrm{ND}(<0.9)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $\mathrm{ND}(<0.6)$ | $\mathrm{ND}(<0.3)$ | $\mathrm{ND}(<0.6)$ |
| Hansard | $\mathbf{1 9 9 4}$ | 1 | $\mathrm{ND}(<1.3)$ | $\mathrm{ND}(<0.5)$ | $\mathrm{ND}(<1)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $\mathrm{ND}(<0.6)$ | $\mathrm{ND}(<0.3)$ | $\mathrm{ND}(<0.3)$ |
| Woodpecker | $\mathbf{1 9 9 4}$ | 2 | $6.9(1.8-12.0)$ | $0.6(0.5-0.8)$ | $\mathrm{ND}(<1)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $2.2(2.1-2.2)$ | $\mathrm{ND}(<0.46)$ | $\mathrm{ND}(<0.6)$ |
| Marguerite | $\mathbf{1 9 9 4}$ | 1 | 11 | 1.4 | $\mathrm{ND}(<0.7)$ |
|  | $\mathbf{1 9 9 5}$ | 1 | 2.8 | 0.9 | $\mathrm{ND}(<0.5)$ |
| Agassiz | $\mathbf{1 9 9 4}$ | 1 | 12 | 3.6 | 1.3 |
|  | $\mathbf{1 9 9 5}$ | 1 | 6.8 | 1.3 | 0.3 |
| N. Thompson | $\mathbf{1 9 9 4}$ | 2 | $4.9(4.8-4.9)$ | $1.2(1.1-1.2)$ | $\mathrm{ND}(<0.4)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $1.8(1.4-2.1)$ | $0.3(0.2-0.3)$ | $\mathrm{ND}(<0.6)$ |
| Thompson | $\mathbf{1 9 9 4}$ | 2 | $4.4(3.1-5.7)$ | $\mathrm{ND}(<0.5)$ | $\mathrm{ND}(<0.4)$ |
|  | $\mathbf{1 9 9 5}$ | 2 | $4.7(3.8-5.5)$ | $1.0(0.9-1.2)$ | $\mathrm{ND}(<0.7)$ |

Table 43. World Health Organization toxic equivalency factors (TEF) for PCB congeners relative to 2,3,7,8TCDD (van den Berg et al. 1998).

|  | TEF relative to 2,3,7,8-TCDD |  |  |
| :---: | :---: | :---: | :---: |
| UIPAC Number | Mammals | Fish | Birds |
| 81 | 0.0001 | 0.0005 | 0.1 |
| 77 | 0.0001 | 0.00001 | 0.05 |
| 126 | 0.1 | 0.005 | 0.1 |
| 169 | 0.01 | 0.00005 | 0.001 |
|  |  |  |  |
| 105 | 0.0001 | $<0.000005$ | 0.0001 |
| 114 | 0.0005 | $<0.000005$ | 0.0001 |
| 118 | 0.0001 | $<0.000005$ | 0.00001 |
| 123 | 0.0001 | $<0.000005$ | 0.00001 |
| 156 | 0.0005 | $<0.000005$ | 0.0001 |
| 157 | 0.0005 | $<0.000005$ | 0.0001 |
| 167 | 0.00001 | $<0.000005$ | 0.00001 |
| 189 | 0.0001 | $<0.000005$ | 0.00001 |

Table 44. WHO PCB TEQs for fish muscle from the Fraser basin in1994/5. Values shown as pg TCDD TEQ/g wet weight (mean of $4-5$ composites and standard error) using TEFs values for mammals, fish and birds from van den Berg et al. (1998). For the purpose of TEG calculation, non-detects were set to zero. ns = no sample

|  |  | Peamouth Chub |  |  | Starry Founder |  |  | Whitefish |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Reach | Year | Mammal | Fish | Bird | Mammal | Fish | Bird | Mammal | Fish | Bird |
| Nechako | 1994 | 0.00 (0.00) | 0.00 (0.00) | 0.06 (0.02) | -ns- | -ns- | -ns- | 0.13 (0.06) | 0.01 (0.00) | 0.21 (0.06) |
|  | 1995 | 0.00 (0.00) | 0.00 (0.00) | 0.03 (0.00) | -ns- | -ns- | -ns- | 0.10 (0.01) | 0.00 (0.00) | 0.12 (0.00) |
| Hansard | 1994 | 0.08 (0.01) | 0.00 (0.00) | 0.42 (0.23) | -ns- | -ns- | -ns- | 0.04 (0.01) | 0.00 (0.00) | 0.08 (0.01) |
|  | 1995 | 0.02 (0.01) | 0.00 (0.00) | 0.08 (0.03) | -ns- | -ns- | -ns- | 0.05 (0.01) | 0.00 (0.00) | 0.09 (0.01) |
| Woodpecker | 1994 | 0.09 (0.02) | 0.00 (0.00) | 0.81 (0.57) | -ns- | -ns- | -ns- | 0.13 (0.03) | 0.00 (0.00) | 0.17 (0.02) |
|  | 1995 | 0.01 (0.00) | 0.00 (0.00) | 0.09 (0.02) | -ns- | -ns- | -ns- | 0.05 (0.01) | 0.00 (0.00) | 0.15 (0.03) |
| Marguerite | 1994 | 0.50 (0.19) | 0.02 (0.01) | 0.98 (0.30) | -ns- | -ns- | -ns- | 0.13 (0.02) | 0.01 (0.00) | 0.56 (0.15) |
|  | 1995 | 0.02 (0.00) | 0.00 (0.00) | 0.14 (0.06) | -ns- | -ns- | -ns- | 0.06 (0.01) | 0.00 (0.00) | 0.21 (0.02) |
| Agassiz | 1994 | 0.58 (0.09) | 0.03 (0.00) | 2.03 (0.49) | -ns- | -ns- | -ns- | 0.68 (0.30) | 0.03 (0.01) | 0.89 (0.20) |
|  | 1995 | 0.13 (0.01) | 0.00 (0.00) | 0.49 (0.08) | -ns- | -ns- | -ns- | 0.41 (0.09) | 0.01 (0.00) | 0.94 (0.20) |
| Mission | 1994 | 0.66 (0.22) | 0.03 (0.01) | 2.17 (0.71) | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
|  | 1995 | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
| Barnston | 1994 | 0.29 (0.03) | 0.01 (0.00) | 0.96 (0.06) | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
|  | 1995 | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
| Main Arm | 1994 | 0.40 (0.07) | 0.02 (0.00) | 1.13 (0.19) | 0.04 (0.01) | 0.00 (0.00) | 0.10 (0.03) | -ns- | -ns- | -ns- |
|  | 1995 | 0.17 (0.06) | 0.01 (0.00) | 0.60 (0.13) | 0.03 (0.01) | 0.00 (0.00) | 0.08 (0.01) | -ns- | -ns- | -ns- |
| North Arm | 1994 | 0.38 (0.04) | 0.01 (0.00) | 1.11 (0.13) | 0.18 (0.07) | 0.01 (0.00) | 0.60 (0.19) | -ns- | -ns- | -ns- |
|  | 1995 | 0.13 (0.00) | 0.00 (0.00) | 0.94 (0.09) | 0.07 (0.02) | 0.00 (0.00) | 0.21 (0.05) | -ns- | -ns- | -ns- |
| N. Thompson | 1994 | 0.14 (0.02) | 0.01 (0.00) | 0.36 (0.05) | -ns- | -ns- | -ns- | 0.36 (0.02) | 0.02 (0.00) | 0.84 (0.07) |
|  | 1995 | 0.05 (0.01) | 0.00 (0.00) | 0.24 (0.05) | -ns- | -ns- | -ns- | 0.17 (0.04) | 0.01 (0.00) | 0.48 (0.21) |
| Thompson | 1994 | 0.11 (0.02) | 0.01 (0.00) | 0.35 (0.06) | -ns- | -ns- | -ns- | 0.38 (0.19) | 0.02 (0.01) | 0.75 (0.27) |
|  | 1995 | 0.03 (0.00) | 0.00 (0.00) | 0.22 (0.01) | -ns- | -ns- | -ns- | 0.10 (0.01) | 0.00 (0.00) | 0.34 (0.03) |

Table 45. WHO PCB TEQs for fish liver from the Fraser basin in1994/5. Values shown as pg TCDD TEQ/g wet weight (mean of 4-5 composites and standard error) using TEFs values for mammals, fish and birds from van den Berg et al. (1998). For the purpose of TEQ calculation, non-detects were set to zero.

|  |  | Peamouth Chub |  |  | Sitarry Founder |  |  | Whitefish |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Reach | Year | Mammal | Fish | Bird | Mammal | Fish | Bird | Mammal | Fish | Bird |
| Nechako | 1994 | 0.80 | 0.03 | 1.09 | -ns- | -ns- | -ns- | 0.06 | 0.00 | 0.09 |
|  | 1995 | 0.41 | 0.01 | 0.38 | -ns- | -ns- | -ns- | 0.00 | 0.00 | 0.00 |
| Hansard | 1994 | 0.61 | 0.03 | 1.91 | -ns- | -ns- | -ns- | 0.00 | 0.00 | 0.00 |
|  | 1995 | 0.93 | 0.02 | 0.61 | -ns- | -ns- | -ns- | 0.00 | 0.00 | 0.00 |
| Woodpecker | 1994 | 0.43 | 0.02 | 1.16 | -ns- | -ns- | -ns- | 0.06 | 0.00 | 0.41 |
|  | 1995 | 0.45 | 0.01 | 0.37 | -ns- | -ns- | -ns- | 0.03 | 0.00 | 0.01 |
| Marguerite | 1994 | 0.59 | 0.03 | 1.36 | -ns- | -ns- | -ns- | 0.14 | 0.01 | 0.69 |
|  | 1995 | 0.40 | 0.01 | 0.32 | -ns- | -ns- | -ns- | 0.04 | 0.00 | 0.05 |
| Agassiz | 1994 | 2.04 | 0.08 | 5.10 | -ns- | -ns- | -ns- | 0.70 | 0.03 | 1.02 |
|  | 1995 | 1.68 | 0.03 | 0.88 | -ns- | -ns- | -ns- | 0.19 | 0.01 | 0.49 |
| Mission | 1994 | 6.00 | 0.26 | 21.25 | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
|  | 1995 | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
| Barnston | 1994 | 5.32 | 0.24 | 12.92 | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
|  | 1995 | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- | -ns- |
| Main Arm | 1994 | 5.28 | 0.22 | 14.80 | 1.72 | 0.07 | 2.69 | -ns- | -ns- | -ns- |
|  | 1995 | 3.62 | 0.07 | 2.17 | 0.91 | 0.02 | 0.60 | -ns- | -ns- | -ns- |
| North Arm | 1994 | 6.77 | 0.29 | 23.18 | 3.13 | 0.13 | 5.85 | -ns- | -ns- | -ns- |
|  | 1995 | 6.79 | 0.11 | 3.16 | 1.26 | 0.02 | 0.71 | -ns- | -ns- | -ns- |
| N. Thompson | 1994 | 0.82 | 0.03 | 1.66 | -ns- | -ns- | -ns- | 0.15 | 0.01 | 0.36 |
|  | 1995 | 0.51 | 0.01 | 0.41 | -ns- | -ns- | -ns- | 0.02 | 0.00 | 0.00 |
| Thompson | 1994 | 1.24 | 0.06 | 3.13 | -ns- | -ns- | -ns- | 0.15 | 0.01 | 0.34 |
|  | 1995 | 0.78 | 0.01 | 0.54 | -ns- | -ns- | -ns- | 0.13 | 0.00 | 0.08 |

Table 46. Historical total PCB levels in fish tissues from the lower Fraser River.

| Region | Year | Species | Tissue | Range <br> (ng/g w.w.) | Ref. |
| :---: | :--- | :---: | :---: | :---: | :--- |
| Lower Fraser $^{1}$ | 1971 | chub | muscle | $<20-527$ | (Albright et al. 1975) |
|  | 1973 | various | muscle | ND -3700 | (Johnston et al. 1975) |
|  | 1980 | various | muscle | $<300-800$ | (Singleton 1983) |
|  | 1988 | chub | muscle | $100-320$ | (Swain and Walton 1989) |
|  | 1988 | chub | liver | $<50-1090$ | (Swain and Walton 1989) |
|  |  |  |  |  |  |
| This study |  |  |  |  |  |
|  | $1994 / 5$ | various | muscle | $0.22-22.0$ |  |
|  | $1994 / 5$ | various | liver | $40.0-245.0$ |  |

${ }^{1}$ Corresponds to sampling reaches of North Arm, Main Arms and Barnston Reaches in the present study.
${ }^{2}$ Corresponds to Agassiz and Mission reaches in the present study.

Table 47. Mean coplanar PCBs in whole composite long-nose sucker (Catastomus catastomus) from sites in the Fraser Valley, as reported by (Rantalainen et al. 1998). Indicated are the sampling sites and the corresponding FRAP reach designations. Concentrations in $\mathrm{pg} / \mathrm{g}$ wet wt. Lipid levels are shown for individual composites.

| Site | Annacis | Burnaby | MacDonald | Rosedale |
| :---: | :---: | :---: | :---: | :---: |
| FRAP Reach | Main Arm | North Arm | North Arm | Agassiz |
| $\mathrm{N}=$ | 3 | 1 | 2 | 1 |
| PCB77 | 148 | 166 | 53.7 | 27.2 |
| PCB126 | 14.7 | 28.3 | 11.5 | 6.2 |
| PCB169 | 0.8 | 1.7 | 0.9 | 0.8 |
|  |  |  |  |  |
| \%lipid | $4.1,5.7,9.9$ | 9.1 | 7.5 |  |

Table 48. Organochlorine (OC) pesticides analyzed in fish tissues from the Fraser basin, 1994/1995.

| "Drins" | Hexachlorocvclohexane | DDT Series | Other Ocs |
| :---: | :---: | :---: | :---: |
| Aldrin <br> Dieldrin <br> Endrin | alpha HCH <br> beta HCH <br> delta HCH <br> gamma HCH | $\begin{aligned} & \text { o,p'-DDT } \\ & \text { p,p'-DDT } \\ & \text { p,p'-DDD } \\ & 0, \mathrm{p}^{\prime}-\mathrm{DDE} \end{aligned}$ | Total Toxaphene Hexachlorobenzene Methoxychlor Mirex |
| Endosuphan |  |  |  |
| alpha-Endosulphan (I) | Heptachlor-related | Chlordanes |  |
| beta-Endosulphan (II) Endosulphan Sulphate | Heptachlor Heptachlor Epoxide | cis-Chlordane Oxychlordane trans-Chlordane trans-Nonachlor |  |

Table 49. Summary of organochlorine pesticide concentrations in peamouth chub muscle tissues from the Fraser basin, 1994 and 1995; concentrations shown are means (standard error) in $\mathrm{ng} / \mathrm{g}$ wet wt., with the mean and range of sample detection limits; ND $=$ not detected.

| Pesticide | Year | Peamouth Chub Muscle (ng/g wet weight) |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Detect Limits | Nechako | Hansard | Woodpecker | Marguerite | Agassiz | Mission | Barnston | Main Arm | North Arm | N. Thompson | Thompson |
| Aldrin | 1994 | 0.07 (0.02-0.24) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.06 (0.01-0.72) | ND | ND | ND | ND | ND | - | - | ND | ND | ND | ND |
| Dieldrin | 1994 | 0.02 (0.01-0.04) | 0.03 (0.00) | 0.08 (0.01) | 0.08 (0.01) | 0.12 (0.00) | 0.17 (0.02) | 0.18 (0.01) | 0.17 (0.01) | 0.19 (0.01) | 0.15 (0.01) | 0.13 (0.01) | 0.15 (0.01) |
|  | 1995 | 0.01 (0.003-0.02) | 0.01 (0.00) | 0.05 (0.01) | 0.01 (0.00) | 0.02 (0.01) | 0.10 (0.01) | . | - | 0.09 (0.01) | 0.03 (0.01) | 0.15 (0.05) | 0.02 (0.00) |
| Endrin | 1994 | 0.06 (0.01-0.18) | ND | ND | ND | 0.04 (0.01) | ND | 0.10 (0.02) | ND | ND | ND | 0.02 (0.00) | 0.02 (0.01) |
|  | 1995 | 0.01 (0.01-0.04) | ND | ND | ND | ND | 0.01 (0.00) | - | - | ND | ND | 0.01 (0.00) | ND |
| alpha-Endosulphan (I) | 1994 | 0.02 (0.01-0.04) | ND | 0.04 (0.01) | ND | 0.05 (0.01) | ND | 0.25 (0.01) | 0.17 (0.02) | 0.21 (0.01) | 1.11 (0.25) | ND | 0.01 (0.00) |
|  | 1995 | 0.01 (0.01-0.04) | ND | ND | ND | ND | 0.14 (0.02) | - | - | 0.18 (0.03) | 0.04 (0.01) | 0.02 (0.01) | 0.01 (0.00) |
| beta-Endosulphan (II) | 1994 | 0.03 (0.01-0.20) | ND | 0.04 (0.01) | ND | ND | ND | 0.05 (0.01) | 0.03 (0.01) | 0.04 (0.01) | 0.23 (0.06) | ND | ND |
|  | 1995 | 0.01 (0.01-0.05) | ND | 0.01 (0.00) | ND | ND | ND | - | - | 0.05 (0.02) | ND | 0.02 (0.01) | ND |
| Endosulphan Sulphate | 1994 | 0.03 (0.01-0.20) | ND | 0.12 (0.02) | ND | ND | ND | 0.18 (0.02) | 0.16 (0.02) | 0.14 (0.01) | 0.87 (0.22) | 0.14 (0.01) | ND |
|  | 1995 | 0.02 (0.01-0.05) | 0.01 (0.00) | 0.07 (0.01) | ND | 0.02 (0.00) | 0.04 (0.00) | - | - | 0.21 (0.06) | 0.04 (0.01) | 0.12 (0.03) | 0.01 (0.00) |
| alpha HCH | 1994 | 0.12 (0.02-0.33) | 0.10 (0.01) | ND | 0.11 (0.01) | 0.12 (0.01) | 0.10 (0.01) | ND | 0.08 (0.01) | 0.15 (0.02) | 0.11 (0.01) | ND | 0.12 (0.02) |
|  | 1995 | 0.05 (0.02-0.12) | 0.07 (0.02) | 0.04 (0.01) | 0.02 (0.01) | 0.04 (0.01) | 0.15 (0.01) | - | - | 0.07 (0.01) | 0.05 (0.01) | 0.15 (0.03) | 0.14 (0.04) |
| beta HCH | 1994 | 0.20 (0.04-0.59) | ND | ND | ND | ND | ND | 2.86 (0.56) | 2.91 (0.44) | 0.32 (0.132) | ND | ND | 2.44 (0.46) |
|  | 1995 | 0.08 (0.03-0.24) | ND | ND | ND | 0.59 (0.33) | 0.10 (0.07) | . | - | ND | ND | 0.72 (0.49) | 2.39 (0.98) |
| delta HCH | 1994 | 0.14 (0.03-0.38) | ND | ND | ND | ND | ND | 0.88 (0.20) | 0.84 (0.13) | ND | ND | ND | 0.46 (0.06) |
|  | 1995 | 0.07 (0.03-1.40) | ND | ND | ND | 0.13 (0.06) | 0.05 (0.02) | - | - | ND | ND | 0.08 (0.03) | ND |
| gamma HCH | 1994 | ${ }^{0.14(0.02-0.38)}$ | ND | 0.23 (0.04) | ND | 0.14 (0.02) | ND | 0.89 (0.17) | 1.30 (0.19) | 0.19 (0.07) | 0.05 (0.00) | ND | 1.49 (0.23) |
|  | 1995 | 0.13 (0.03-0.17) | ND | ND | 0.08 (0.03) | 0.45 (0.23) | 0.18 (0.08) | - | . | 0.16 (0.05) | ND | 0.54 (0.24) | 1.41 (0.43) |
| Hexachlorobenzene | 1994 | 0.05 (0.01-0.22) | 0.10 (0.01) | 0.18 (0.01) | 0.19 (0.03) | 0.50 (0.02) | 0.31 (0.02) | 0.25 (0.01) | 0.22 (0.01) | 0.29 (0.04) | 0.23 (0.01) | 0.29 (0.04) | 0.22 (0.01) |
|  | 1995 | 0.02 (0.01-0.03) | 0.09 (0.00) | 0.21 (0.03) | 0.14 (0.02) | 0.10 (0.02) | 0.38 (0.01) | . | . | 0.30 (0.06) | 0.15 (0.02) | 0.34 (0.04) | 0.19 (0.02) |
| Methoxychlor | 1994 | 0.09 (0.01-0.34) | ND | ND | ND | ND | ND | 0.16 (0.03) | ND | 0.09 (0.01) | 0.12 (0.01) | ND | ND |
|  | 1995 | 0.02 (0.01-0.08) | ND | ND | ND | ND | ND | - | - | ND | 0.01 (0.01) | ND | ND |
| Heptachlor | 1994 | 0.28 (0.02-2.00) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.11 (0.04-0.31) | ND | ND | ND | ND | ND | - | - | ND | ND | ND | ND |
| Heptachlor Epoxide | 1994 | 0.02 (0.01-0.06) | 0.03 (0.00) | 0.07 (0.00) | 0.06 (0.01) | 0.08 (0.00) | 0.08 (0.01) | 0.09 (0.00) | 0.07 (0.01) | 0.08 (0.01) | 0.10 (0.01) | 0.08 (0.01) | 0.08 (0.01) |
|  | 1995 | 0.01 (0.003-0.02) | 0.01 (0.00) | 0.02 (0.00) | 0.00 (0.00) | 0.01 (0.00) | 0.03 (0.00) | - | - | 0.03 (0.01) | 0.01 (0.00) | 0.08 (0.03) | 0.01 (0.00) |
| Mirex | 1994 | 0.27 (0.01-0.64) | 0.02 (0.002) | ND | 0.10 (0.02) | ND | 0.07 (0.01) | ND | ND | 0.14 (0.02) | 0.11 (0.01) | ND | ND |
|  | 1995 | 0.20 (0.02-0.0.27) | ND | ND | ND | ND | ND | - | - | ND | ND | ND | 0.07 (0.01) |
| o,p'-DDT | 1994 | 0.31 (0.02-1.60) | ND | ND | ND | 0.21 (0.02) | ND | 0.07 (0.01) | 0.07 (0.01) | 0.10 (0.01) | ND | 0.06 (0.01) | 0.08 (0.01) |
|  | 1995 | 0.04 (0.01-0.11) | ND | 0.03 (0.01) | 0.04 (0.01) | 0.03 (0.01) | 0.17 (0.01) | - | - | 0.13 (0.02) | 0.12 (0.01) | 0.06 (0.02) | 0.07 (0.01) |
| p,p'-DDT | 1994 | 0.46 (0.03-2.70) | ND | ND | ND | ND | ND | 0.05 (0.01) | 0.02 (0.00) | 0.08 (0.01) | ND | ND | ND |
|  | 1995 | 0.03 (0.01-0.09) | ND | ND | ND | ND | ND | - | - | ND | ND | 0.03 (0.01) | ND |
| p,p'-DDD | 1994 | 0.07 (0.02-0.18) | ND | 0.08 (0.02) | 0.04 (0.01) | 0.28 (0.01) | 0.41 (0.04) | 0.28 (0.01) | 0.24 (0.02) | 0.38 (0.02) | 0.54 (0.03) | 0.25 (0.02) | 0.20 (0.01) |
|  | 1995 | 0.02 (0.01-0.07) | ND | 0.12 (0.06) | 0.03 (0.01) | 0.05 (0.02) | 0.43 (0.02) | - | - | 0.48 (0.06) | 0.39 (0.06) | 0.22 (0.02) | 0.14 (0.01) |
| p,p'-DDE | 1994 | 0.07 (0.01-0.24) | 0.32 (0.04) | 1.44 (0.14) | 0.94 (0.26) | 3.50 (0.17) | 7.30 (1.15) | 3.92 (0.20) | 4.02 (0.31) | 4.39 (0.64) | 2.64 (0.15) | 3.90 (0.33) | 2.55 (0.16) |
|  | 1995 | 0.02 (0.01-0.06) | 0.26 (0.02) | 1.59 (0.64) | 0.59 (0.12) | 0.96 (0.16) | 5.30 (0.43) | - | - | 7.22 (1.32) | 2.98 (0.10) | 3.82 (0.86) | 2.28 (0.12) |
| cis-Chlordane | 1994 | 0.07 (0.03-0.21) | 0.05 (0.01) | 0.05 (0.01) | 0.08 (0.02) | 0.25 (0.01) | 0.27 (0.01) | 0.09 (0.01) | 0.08 (0.01) | 0.21 (0.01) | 0.16 (0.02) | 0.13 (0.02) | 0.09 (0.01) |
|  | 1995 | 0.03 (0.01-0.09) | ND | 0.08 (0.04) | 0.03 (0.01) | 0.03 (0.01) | 0.14 (0.02) | - | - | 0.15 (0.04) | 0.10 (0.01) | 0.11 (0.02) | 0.05 (0.01) |
| Oxychlordane | 1994 | 0.21 (0.04-0.45) | ND | ND | ND | ND | ND | 0.15 (0.02) | 0.12 (0.02) | 0.10 (0.02) | 0.06 (0.01) | ND | 0.23 (0.03) |
|  | 1995 | 0.20 (0.07-0.65) | ND | ND | ND | 0.08 (0.02) | ND | - | - | 0.10 (0.02) | 0.13 (0.03) | 0.37 (0.04) | 0.16 (0.10) |
| trans-Chlordane | 1994 | 0.07 (0.02-0.19) | ND | ND | ND | 0.06 (0.01) | ND | ND | ND | 0.05 (0.01) | 0.05 (0.01) | ND | ND |
|  | 1995 | 0.03 (0.01-0.08) | ND | 0.02 (0.01) | ND | ND | ND | - | - | 0.03 (0.00) | 0.03 (0.01) | 0.03 (0.01) | 0.01 (0.00) |
| trans-Nonachlor | 1994 | 0.07 (0.02-0.22) | ND | 0.17 (0.01) | 0.13 (0.03) | 0.50 (0.03) | 0.60 (0.03) | 0.28 (0.02) | 0.20 (0.008) | 0.47 (0.03) | 0.27 (0.02) | 0.32 (0.04) | 0.15 (0.01) |
|  | 1995 | 0.03 (0.01-0.06) | ND | 0.18 (0.08) | 0.07 (0.02) | 0.09 (0.02) | 0.30 (0.05) | - | - | 0.37 (0.08) | 0.23 (0.02) | 0.32 (0.13) | 0.14 (0.02) |
| Total Toxaphene | 1994 | 0.37 (0.06-1.00) | ND | 1.23 (0.11) | 0.69 (0.22) | 5.58 (0.44) | 1.85 (0.20) | 0.83 (0.04) | 0.78 (0.053) | 1.19 (0.18) | 0.38 (0.03) | 2.15 (0.28) | 1.89 (0.24) |
|  | 1995 | 0.17 (0.05-0.44) | ND | 1.09 (0.60) | 0.18 (0.06) | 0.28 (0.11) | 2.15 (0.33) | - | - | 1.99 (0.65) | 0.36 (0.04) | 2.26 (0.98) | 1.21 (0.28) |

Table 50. Summary of organochlorine pesticide concentrations in starry flounder and whitefish muscle tissues from the Fraser basin, 1994 and 1995; concentrations shown are means (standard error), with the mean and range of sample detection limits; $\mathrm{ND}=$ not detected.

| Pesticide | Year | Starry Flounder ( $\mathrm{ng} / \mathrm{g} \mathrm{ww}$ ) |  |  |
| :---: | :---: | :---: | :---: | :---: |
|  |  | Detect Limits | Main Arm | North Arm |
| Aldrin | 1994 | 0.13 (0.04-0.27) | ND | ND |
|  | 1995 | $0.03(0.03-0.04)$ | ND | ND |
| Dieldrin | 1994 | 0.03 (0.01-0.06) | 0.04 (0.00) | 0.07 (0.01) |
|  | 1995 | 0.01 (0.01-0.01) | 0.01 (0.00) | 0.02 (0.02) |
| Endrin | 1994 | 0.12 (0.06-0.22) | ND | ND |
|  | 1995 | 0.01 (0.01-0.03) | ND | ND |
| alpha-Endosulphan (I) | 1994 | 0.04 (0.03-0.06) | ND | ND |
|  | 1995 | 0.01 (0.01-0.01) | ND | ND |
| beta-Endosulphan (II) | 1994 | 0.04 (0.02-0.06) | ND | ND |
|  | 1995 | 0.01 (0.01-0.02) | ND | ND |
| Endosulphan Sulphate | 1994 | 0.04 (0.02-0.07) | 0.05 (0.01) | 0.11 (0.01) |
|  | 1995 | 0.01 (0.01-0.02) | 0.03 (0.00) | 0.05 (0.03) |
| alpha HCH | 1994 | 0.16 (0.06-0.32) | ND | ND |
|  | 1995 | $0.04(0.03-0.05)$ | 0.02 (0.01) | ND |
| beta HCH | 1994 | 0.24 (0.09-0.46) | 0.27 (0.04) | 0.44 (0.15) |
|  | 1995 | 0.06 (0.04-0.07) | ND | 0.08 (0.05) |
| delta HCH | 1994 | 0.20 (0.07-0.40) | ND | ND |
|  | 1995 | 0.06 (0.05-0.06) | ND | ND |
| gamma HCH | 1994 | 0.21 (0.07-0.40) | ND | 0.23 (0.06) |
|  | 1995 | 0.06 (0.05-0.0.06) | ND | ND |
| Hexachlorobenzene | 1994 | 0.05 (0.02-0.09) | 0.06 (0.01) | 0.09 (0.01) |
|  | 1995 | 0.01 (0.01-0.01) | 0.06 (0.01) | 0.05 (0.01) |
| Methoxychlor | 1994 | 0.24 (0.14-0.40) | ND | ND |
|  | 1995 | 0.02 (0.01-0.04) | ND | ND |
| Heptachlor | 1994 | 0.48 (0.23-0.89) | ND | ND |
|  | 1995 | 0.11 (0.06-0.16) | ND | ND |
| Heptachlor Epoxide | 1994 | 0.04 (0.03-0.06) | ND | ND |
|  | 1995 | 0.01 (0.01-0.01) | ND | 0.01 (0.01) |
| Mirex | 1994 | 0.71 (0.50-1.40) | ND | ND |
|  | 1995 | 0.20 (0.03-0.22) | ND | ND |
| o, ${ }^{\prime}$ '-DDT | 1994 | 0.11 (0.06-0.17) | ND | ND |
|  | 1995 | 0.04 (0.02-0.03) | 0.20 (0.03) | 0.18 (0.04) |
| p,p'-DDT | 1994 | 0.12 (0.05-0.20) | 0.23 (0.02) | 0.34 (0.04) |
|  | 1995 | 0.02 (0.02-0.02) | 0.74 (0.10) | 0.64 (0.12) |
| p, p'-DDD | 1994 | 0.05 (0.02-0.0.08) | 0.07 (0.01) | 0.13 (0.02) |
|  | 1995 | 0.01 (0.01-0.02) | 0.14 (0.02) | 0.19 (0.05) |
| p, p'-DDE | 1994 | 0.05 (0.02-0.10) | 1.40 (0.20) | 1.09 (0.08) |
|  | 1995 | 0.01 (0.01-0.01) | 0.89 (0.12) | 0.78 (0.12) |
| cis-Chlordane | 1994 | 0.12 (0.06-0.21) | ND | ND |
|  | 1995 | 0.02 (0.02-0.03) | 0.03 (0.01) | 0.04 (0.01) |
| Oxychlordane | 1994 | 0.26 (0.10-0.50) | ND | ND |
|  | 1995 | 0.16 (0.13-0.24) | ND | ND |
| trans-Chlordane | 1994 | 0.11 (0.05-0.19) | ND | ND |
|  | 1995 | 0.02 (0.02-0.03) | ND | ND |
| trans-Nonachlor | 1994 | 0.11 (0.05-0.20) | 0.09 (0.01) | 0.10 (0.01) |
|  | 1995 | 0.02 (0.01-0.02) | 0.06 (0.01) | 0.07 (0.02) |
| Total Toxaphene | 1994 | 0.33 (0.21-0.48) | ND | 0.25 (0.04) |
|  | 1995 | 0.19 (0.05-0.24) | ND | 0.15 (0.04) |


|  | Mountain Whitefish Muscle (ng/g wet weight) |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Detect Limits | Nechako | Hansard | Woodpecker | Marguerite | Agassiz | N. Thompson | Thompson |
| 0.07 (0.02-0.36) | ND | ND | ND | ND | ND | ND | ND |
| 07 (0.01-0.35) | ND | ND | ND | ND | 0.08 (0.03 | ND | ND |
| 0.03 (0.01-0.12) | 0.14 (0.01) | 0.06 (0.01) | 0.03 (0.00) | 0.12 (0.01) | 0.21 (0.02) | 0.37 (0.04) | 0.27 (0.01) |
| 0.01 (0.01-0.06) | 0.06 (0.00) | 0.09 (0.02) | 0.08 (0.04) | 0.07 (0.01) | 4.19 (3.94) | 0.20 (0.04) | 0.15 (0.02) |
| 0.06 (0.01-0.18) | ND | ND | 0.08 (0.02) | ND | ND | 0.06 (0.01) | 0.09 (0.01) |
| 0.03 (0.01-0.29) | ND | 0.01 (0.00) | 0.02 (0.01) | 0.02 (0.01) | 0.75 (0.68) | 0.02 (0.01) | 0.05 (0.02) |
| 0.03 (0.01-0.0.09) | 0.05 (0.00 | 0.09 (0.01) | ND | 0.04 (0.01) | 0.14 (0.01) | 27 (0.02 | 0.05 (0.01) |
| 0.02 (0.00-0.06) | 0.08 (0.04) | 0.08 (0.02) | ND | 0.06 (0.01) | 0.14 (0.02) | 0.19 (0.05) | 0.06 (0.01) |
| 0.04 (0.01-0.20) | ND | 0.04 (0.01) | ND | ND | 0.05 (0.01) | 0.12 (0.01) | ND |
| 0.01 (0.01-0.07) | ND | 0.03 (0.01) | ND | ND | 0.02 (0.01) | 0.16 (0.05) | 0.01 (0.00) |
| 0.05 (0.01-0.20) | 0.05 (0.01) | 0.10 (0.01) | ND | 0.03 (0.00) | 0.09 (0.01) | 0.21 (0.01) | 0.11 (0.03) |
| 0.04 (0.01-0.71) | 0.04 (0.01) | 0.07 (0.01) | 0.03 (0.01) | 0.08 (0.00) | 2.25 (2.05) | 0.30 (0.07) | 0.14 (0.03) |
| 0.22 (0.04-0.71) | 0.20 (0.03) | 0.15 (0.01) | ND | 0.23 (0.07) | 0.19 (0.01) | 0.35 (0.02) | 0.35 (0.05) |
| 0.08 (0.01-0.27) | 0.22 (0.04) | 0.11 (0.03) | 0.13 (0.05) | 0.31 (0.15) | 0.69 (0.26) | 0.29 (0.03) | 0.91 (0.34) |
| 0.29 (0.06-1.10) | 0.21 (0.07) | ND | ND | 1.73 (0.39) | 0.70 (0.13) | ND | 0.88 (0.27) |
| 0.13 (0.05-0.42) | ND | ND | 0.49 (0.37) | 0.61 (0.50) | 17.87 (14.78) | 0.30 (0.20) | ND |
| 0.21 (0.04-0.86) | ND | ND | ND | 0.07 (0.01) | ND | ND | 0.33 (0.05) |
| 0.12 (0.03-0.41) | 0.03 (0.00) | ND | 0.14 (0.07) | 0.37 (0.14) | 1.94 (1.06) | 0.12 (0.05) | 0.13 (0.03) |
| 0.21 (0.04-0.86) | ND | ND | 0.34 (0.04) | ND | ND | ND | ND |
| 0.12 (0.01-0.41) | 0.08 (0.02) | 0.07 (0.01) | 0.50 (0.43) | 0.74 (0.63) | 2.23 (1.37) | 0.31 (0.18) | ND |
| 5 (0.02-0.19) | $2(0.0$ | 0.33 (0.02) | 0.19 (0.02) | 29 | 0.33 (0.03) | 0.75 (0.04) | 0.36 (0.02) |
| 0.02 (0.01-0.10) | 0.31 (0.10) | 0.26 (0.02) | 0.29 (0.05) | 0.30 (0.02) | 0.79 (0.25) | 0.65 (0.09) | 0.37 (0.09) |
| 0.11 (0.03-0.25) | ND | ND | 0.02 (0.00) | ND | ND | 0.05 (0.01) | ND |
| 0.04 (0.01-0.25) | ND | ND | ND | ND | 0.26 (0.25) | ND | ND |
| 0.18 (0.05-1.20) | ND | ND | ND | ND | ND | ND | ND |
| 0.11 (0.01-0.48) | 0.03 (0.00) | ND | ND | ND | 0.43 (0.39) | ND | 0.12 (0.04) |
| 0.02 (0.01-0.0.08) | 0.09 (0.01) | 0.04 (0.01) | 0.01 (0.00) | 0.07 (0.00) | 0.08 (0.01) | 0.17 (0.01) | 0.10 (0.04) |
| 0.01 (0.01-0.13) | 0.08 (0.02) | 0.04 (0.01) | 0.03 (0.02) | 0.04 (0.00) | 0.71 (0.63) | 0.08 (0.01) | 0.11 (0.06) |
| 0.24 (0.01-0.43) | ND | 0.04 (0.00) | 0.19 (0.00) | 0.12 (0.01) | 0.15 (0.02) | 0.11 (0.01) | ND |
| 0.22 (0.02-0.27) | 0.13 (0.00) | ND | ND | ND | ND | ND | 0.16 (0.03) |
| 0.20 (0.01-1.30) | 0.12 (0.03) | ND | 0.08 (0.02) | 0.15 (0.02) | 0.97 (0.21) | 0.51 (0.04) | 0.32 (0.03) |
| 0.06 (0.01-0.75) | 0.08 (0.00) | 0.26 (0.06) | 0.04 (0.01) | 0.26 (0.02) | 0.93 (0.14) | 0.41 (0.04) | 0.31 (0.04) |
| 0.28 (0.01-2.20) | 0.08 (0.02) | ND | 0.20 (0.04) | 0.28 (0.02) | 1.35 (0.22) | 1.98 (0.11) | 1.14 (0.13) |
| 0.04 (0.01-0.29) | 0.16 (0.10) | 0.86 (0.20) | 0.10 (0.03) | 0.39 (0.03) | 1.45 (0.12) | 1.38 (0.14) | 2.01 (1.08) |
| 0.05 (0.01-0.19) | 0.07 (0.02) | 0.11 (0.02) | 0.10 (0.03) | 0.13 (0.01) | 1.29 (0.40) | 0.93 (0.08) | 0.49 (0.04) |
| 0.02 (0.01-0.06) | 0.44 (0.42) | 0.20 (0.05) | 0.03 (0.00) | 0.14 (0.02) | 0.85 (0.15) | 0.57 (0.11) | 0.92 (0.59) |
| 0.05 (0.01-0.15) | 1.32 (0.30) | 1.00 (0.09) | 1.12 (0.23) | 1.12 (0.10) | 17.43 (5.12) | 6.73 (0.23) | 4.73 (0.76) |
| 0.02 (0.01-0.06) | 0.48 (0.14) | 1.96 (0.28) | 0.62 (0.09) | 1.53 (0.19) | 12.65 (1.76) | 5.00 (0.52) | 2.89 (0.53) |
| 0.06 (0.02-0.26) | 0.10 (0.03) | 0.13 (0.02) | 0.03 (0.01) | 0.10 (0.01) | 0.41 (0.09) | 0.32 (0.03) | 0.15 (0.01) |
| 0.03 (0.01-0.08) | 0.07 (0.01) | 0.12 (0.03) | 0.03 (0.01) | 0.08 (0.02) | 0.28 (0.08) | 0.22 (0.05) | 0.16 (0.03) |
| 0.31 (0.08-1.30) | ND | ND | 0.21 (0.05) | 0.33 (0.09) | 0.44 (0.14) | ND | ND |
| 0.26 (0.02-1.30) | 0.07 (0.01) | 0.30 (0.20) | 0.30 (0.18) | 0.87 (0.51) | 0.83 (0.34) | 0.27 (0.12) | 0.38 (0.09) |
| 0.06 (0.02-0.24) | ND | ND | ND | ND | 0.10 (0.03) | 0.05 (0.01) | 0.06 (0.01) |
| 0.03 (0.01-0.08) | 0.03 (0.02) | 0.03 (0.01) | ND | ND | 0.11 (0.01) | 0.08 (0.03) | 0.09 (0.05) |
| 0.05 (0.01-0.21) | 0.32 (0.07) | 0.32 (0.04) | 0.14 (0.03) | 0.26 (0.04) | 1.06 (0.20) | 0.75 (0.07) | 0.38 (0.03) |
| 0.03 (0.01-0.08) | 0.12 (0.04) | 0.42 (0.10) | 0.08 (0.02) | 0.17 (0.02) | 0.82 (0.10) | 0.56 (0.06) | 0.26 (0.07) |
| 0.23 (0.10-0.56) | 3.38 (0.53) | 1.91 (0.21) | 3.01 (0.89) | 3.73 (0.36) | 10.30 (2.35) | 11.28 (0.38) | 9.10 (0.56) |
| 0.15 (0.03-0.39) | 1.33 (0.50) | 3.52 (0.62) | 0.33 (0.11) | 1.63 (0.23) | 11.28 (2.04) | 8.16 (1.37) | 5.05 (1.53) |

Table 51. Summary of organochlorine pesticide concentrations in peamouth chub liver tissue from the Fraser basin, 1994 and 1995, concentrations shown are mean with the mean and range of sample detection limits; ND $=$ not detected.

|  |  |  | Peamouth Liver (ng/g wet weight) |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Pesticide | Year | Detect Limits | Nechako | Hansard | Woodpecker | Marguerite | Agassiz | Mission | Barnston | Main Arm | North Arm | N. Thompson | Thompson |
| Aldrin | 1994 | 0.12 (0.02-0.38) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.29 (0.08-1.50) | ND | ND | ND | ND | ND | NS | NS | ND | ND | ND | ND |
| Dieldrin | 1994 | 0.13 (0.01-0.72) | 0.44 | 1.11 | 0.61 | 0.28 | 0.59 | 2.00 | 2.00 | 1.40 | 1.65 | 0.44 | 1.39 |
|  | 1995 | 0.14 (0.05-0.35) | 0.33 | 0.33 | 0.57 | 0.39 | 1.70 | NS | NS | 1.75 | 2.10 | 0.92 | 0.83 |
| Endrin | 1994 | 0.46 (0.12-2.04) | ND | 1.17 | ND | ND | 0.76 | 0.80 | 0.50 | ND | ND | ND | 0.39 |
|  | 1995 | 0.25 (0.09-0.59) | ND | ND | ND | ND | 0.26 | NS | NS | ND | ND | ND | 0.29 |
| alpha-Endosulphan (I) | 1994 | 0.10 (0.02-0.41) | ND | 0.46 | 0.26 | ND | ND | 2.60 | 2.30 | 1.85 | 9.60 | ND | 0.29 |
|  | 1995 | 0.13 (0.06-0.24) | ND | 0.12 | ND | ND | 0.43 | NS | NS | 1.80 | 1.33 | 0.12 | 0.12 |
| beta-Endosulphan (II) | 1994 | 0.13 (0.05-0.38) | ND | ND | ND | ND | ND | 0.37 | 0.13 | ND | ND | ND | ND |
|  | 1995 | 0.16 (0.07-0.28) | ND | 0.07 | ND | ND | ND | NS | NS | 0.47 | 0.34 | ND | 0.15 |
| Endosulphan Sulphate | 1994 | 0.13 (0.06-0.48) | ND | ND | 0.25 | ND | ND | 1.00 | 0.54 | ND | 0.60 | ND | 0.52 |
|  | 1995 | 0.19 (0.08-0.38) | 0.31 | 0.93 | 0.32 | 0.16 | 0.61 | NS | NS | 3.00 | 2.07 | 0.76 | 0.84 |
| alpha HCH | 1994 | 0.21 (0.06-0.74) | 1.33 | 0.73 | 0.61 | 0.31 | 0.48 | 2.30 | 2.60 | 1.45 | 1.43 | 0.51 | 2.05 |
|  | 1995 | 0.28 (0.06-1.80) | 0.93 | 0.56 | 0.29 | ND | 1.68 | NS | NS | ND | 1.30 | 0.86 | 1.45 |
| beta HCH | 1994 | 0.32 (0.10-1.09) | ND | ND | ND | ND | ND | 0.55 | 14.00 | 2.22 | 23.50 | ND | ND |
|  | 1995 | 0.45 (0.10-2.90) | ND | ND | 3.96 | 6.60 | 0.86 | NS | NS | ND | 11.67 | 7.45 | 1.80 |
| delta HCH | 1994 | 0.26 (0.07-0.95) | ND | ND | ND | ND | ND | ND | ND | 2.00 | 8.12 | ND | ND |
|  | 1995 | 0.44 (0.10-2.80) | ND | ND | 0.60 | ND | ND | NS | NS | ND | 1.50 | ND | 0.27 |
| gamma HCH | 1994 | 0.28 (0.07-0.92) | ND | ND | ND | ND | ND | ND | 8.40 | 2.41 | 11.50 | 0.09 | 0.34 |
|  | 1995 | 0.44 (0.10-2.80) | ND | ND | 1.91 | ND | ND | NS | NS | 2.30 | 9.07 | ND | 0.30 |
| Hexachlorobenzene | 1994 | 0.10 (0.04-0.26) | 1.14 | 1.50 | 0.83 | 1.15 | 1.90 | 3.00 | 3.10 | 3.35 | 2.70 | 1.40 | 2.20 |
|  | 1995 | 0.10 (0.01-0.84) | 0.87 | 1.35 | 1.35 | 0.70 | 3.48 | NS | NS | 2.80 | 2.43 | 1.65 | 1.72 |
| Methoxychlor | 1994 | 0.80 (0.14-4.70) | ND | ND | ND | ND | ND | ND | ND | ND | 0.83 | ND | ND |
|  | 1995 | 0.40 (0.14-0.83) | ND | ND | ND | ND | ND | NS | NS | ND | 0.57 | ND | ND |
| Heptachlor | 1994 | 0.55 (0.20-1.70) | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.37 (0.10-2.40) | ND | ND | ND | ND | ND | NS | NS | ND | ND | ND | ND |
| Heptachlor Epoxide | 1994 | 0.13 (0.02-0.61) | 0.40 | 0.73 | 0.48 | 0.15 | 0.52 | 1.20 | 0.95 | 0.82 | 1.02 | 0.24 | 0.62 |
|  | 1995 | 0.12 (0.04-0.29) | 0.29 | 0.22 | 0.38 | 0.21 | 0.59 | NS | NS | 0.49 | 0.65 | 0.47 | 0.28 |
| Mirex | 1994 | 0.77 (0.01-3.40) | 0.08 | 0.28 | 0.14 | 0.23 | 0.14 | ND | ND | ND | ND | 0.26 | 0.32 |
|  | 1995 | 0.15 (0.01-0.70) | ND | 0.17 | 0.30 | ND | 0.23 | NS | NS | 0.45 | 0.72 | 0.18 | 0.31 |
| o,p'-DDT | 1994 | 0.72 (0.08-2.00) | ND | ND | ND | ND | ND | 1.10 | 1.60 | 0.94 | 1.08 | ND | ND |
|  | 1995 | 0.26 (0.04-1.60) | ND | 0.03 | ND | ND | 0.92 | NS | NS | ND | 1.47 | ND | 0.23 |
| p, p'-DDT | 1994 | 1.13 (0.08-3.30) | ND | ND | ND | ND | ND | 1.60 | 2.10 | 1.50 | 1.40 | ND | ND |
|  | 1995 | 0.22 (0.03-1.40) | ND | ND | 0.13 | ND | 0.15 | NS | NS | ND | 0.51 | ND | 0.14 |
| p,p'-DDD | 1994 | 0.19 (0.05-0.52) | 0.21 | 0.76 | ND | ND | 2.05 | 4.00 | 6.00 | 5.10 | 5.90 | 0.86 | 1.70 |
|  | 1995 | 0.15 (0.02-0.96) | 0.25 | 0.77 | 0.41 | 0.58 | 4.53 | NS | NS | 5.80 | 8.30 | 1.72 | 1.40 |
| p,p'-DDE | 1994 | 0.12 (0.04-0.34) | 4.08 | 10.80 | 4.80 | 7.10 | 33.50 | 48.00 | 73.00 | 51.00 | 41.00 | 14.50 | 24.50 |
|  | 1995 | 0.08 (0.01-0.52) | 5.95 | 12.40 | 5.80 | 8.00 | 45.50 | NS | NS | 75.00 | 58.67 | 18.00 | 17.17 |
| cis-Chlordane | 1994 | 0.21 (0.10-0.42) | 0.34 | 0.96 | 0.56 | 1.05 | 1.29 | 1.60 | 1.90 | 1.65 | 1.75 | 1.11 | 1.25 |
|  | 1995 | 0.19 (0.03-1.19) | 0.24 | 0.54 | 0.45 | 0.27 | 1.67 | NS | NS | 1.70 | 1.83 | 0.59 | 0.60 |
| Oxychlordane | 1994 | 0.42 (0.15-0.95) | ND | ND | ND | ND | 0.42 | ND | 1.80 | 1.55 | 1.06 | 0.33 | ND |
|  | 1995 | 1.65 (0.27-9.00) | ND | ND | ND | ND | 0.69 | NS | NS | ND | 0.66 | ND | 0.34 |
| trans-Chlordane | 1994 | 0.21 (0.10-0.43) | ND | 0.15 | ND | ND | 0.36 | 0.50 | 0.60 | 0.39 | 0.72 | 0.25 | 0.04 |
|  | 1995 | 0.17 (0.03-1.10) | ND | 0.11 | 0.12 | ND | 0.44 | NS | NS | ND | 0.78 | 0.17 | 0.14 |
| trans-Nonachlor | 1994 | 0.20 (0.10-0.44) | 0.55 | 1.85 | 0.88 | 2.00 | 2.45 | 3.50 | 4.70 | 4.05 | 3.15 | 1.90 | 2.30 |
|  | 1995 | 0.11 (0.03-0.59) | 0.62 | 1.50 | 0.80 | 0.68 | 3.62 | NS | NS | 5.15 | 4.10 | 1.40 | 0.99 |
| Total Toxaphene | 1994 | 0.48 (0.10-1.15) | 4.73 | 16.50 | 8.50 | 13.00 | 16.00 | 42.00 | 47.00 | 28.50 | 23.00 | 11.00 | 33.50 |
|  | 1995 | 0.40 (0.14-0.98) | 4.40 | 16.00 | 8.70 | 5.60 | 42.67 | NS | NS | 30.00 | 22.67 | 16.00 | 17.00 |

Table 52. Summary of organochlorine concentrations in starry flounder and mountain whitefish liver tissues, 1994 and 1995; concentrations shown are mean with the mean and range of sample detection limits; ND = not detected.

| Pesticide | Year | Starry Flounder Liver ( $\mathrm{ng} / \mathrm{g}$ ww) |  |  | Mountain Whitefish Liver (ng/g wet weight) |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Detect Limits | Main Arm | North Arm | Detect Limits | Nechako | Hansard | Woodpecker | Marguerite | Agassiz | N. Thompson | Thompson |
| Aldrin | 1994 | 0.37 (0.36-0.38) | ND | ND | 0.06 (0.04-0.11) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.23 (0.07-0.39) | ND | ND | 0.37 (0.12-0.59) | ND | ND | ND | ND | ND | ND | ND |
| Dieldrin | 1994 | 0.04 (0.03-0.04) | 0.32 | 0.52 | 0.06 (0.02-0.18) | 0.05 | ND | ND | 0.15 | 0.14 | 0.11 | 0.17 |
|  | 1995 | $0.11(0.10-0.12)$ | 0.46 | 0.55 | 0.06 (0.03-0.09) | ND | 0.09 | ND | 0.09 | 0.17 | ND | 0.12 |
| Endrin | 1994 | 0.22 (0.14-0.30) | ND | ND | 0.18 (0.09-0.42) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.22 (0.20-0.23) | ND | ND | 0.12 (0.06-0.22) | ND | ND | ND | ND | ND | ND | ND |
| alpha-Endosulphan (I) | 1994 | 0.25 (0.16-0.33) | ND | ND | 0.07 (0.02-0.21) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.15 (0.15-0.15) | ND | ND | 0.07 (0.04-0.10) | ND | 0.10 | ND | ND | ND | 0.06 | ND |
| beta-Endosulphan (II) | 1994 | 0.05 (0.04-0.06) | ND | 0.06 | 0.08 (0.03-0.17) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.20 (0.19-0.20) | ND | ND | 0.25 (0.04-1.10) | ND | ND | ND | ND | ND | ND | ND |
| Endosulphan Sulphate | 1994 | 0.05 (0.04-0.06) | ND | 0.06 | 0.11 (0.04-0.47) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.26 (0.25-0.26) | 1.50 | 1.60 | 0.08 (0.05-0.14) | 0.14 | 0.20 | 0.14 | 0.17 | 0.30 | 0.30 | 0.40 |
| alpha HCH | 1994 | 0.34 (0.32-0.36) | ND | ND | 0.12 (0.06-0.20) | 0.57 | 0.36 | 0.36 | 0.28 | 1.10 | 0.56 | 0.50 |
|  | 1995 | 0.11 (0.11-0.11) | 0.25 | 0.28 | 0.29 (0.08-0.53) | 0.64 | ND | 1.02 | ND | 0.81 | 0.65 | 0.38 |
| beta HCH | 1994 | 0.48 (0.42-0.53) | ND | ND | 0.19 (0.09-0.32) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.17 (0.16-0.18) | 0.25 | 0.38 | 0.51 (0.13-1.20) | ND | ND | 6.35 | ND | ND | ND | 2.60 |
| delta HCH | 1994 | 0.39 (0.34-0.43) | ND | ND | 0.14 (0.06-0.24) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.18 (0.17-0.18) | ND | ND | 0.44 (0.14-0.83) | ND | ND | ND | ND | ND | ND | ND |
| gamma HCH | 1994 | 0.39 (0.34-0.43) | ND | ND | 0.15 (0.07-0.23) | ND | ND | ND | ND | 0.31 | 0.11 | 0.03 |
|  | 1995 | 0.18 (0.17-0.18) | ND | 0.70 | 0.44 (0.14-0.83) | ND | ND | ND | ND | ND | ND | ND |
| Hexachlorobenzene | 1994 | 0.12 (0.09-0.15) | 0.56 | 0.54 | 0.08 (0.03-0.17) | 0.26 | 0.30 | 0.34 | 0.66 | 1.10 | 0.39 | 0.36 |
|  | 1995 | 0.02 (0.02-0.02) | 0.62 | 0.46 | 0.06 (0.03-0.11) | 0.42 | 0.36 | 0.29 | 0.33 | 0.88 | 0.42 | 0.61 |
| Methoxychlor | 1994 | 0.33 (0.27-0.39) | ND | 0.39 | 0.39 (0.10-1.10) | ND | ND | ND | ND | ND | 0.09 | ND |
|  | 1995 | 0.38 (0.35-0.40) | ND | ND | 0.17 (0.10-0.29) | ND | ND | ND | ND | ND | ND | ND |
| Heptachlor | 1994 | 1.03 (0.96-1.10) | ND | ND | 0.37 (0.22-0.76) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.14 (0.14-0.14) | ND | ND | 0.67 (0.18-1.10) | ND | ND | ND | ND | ND | ND | ND |
| Heptachlor Epoxide | 1994 | 0.13 (0.13-0.13) | ND | ND | 0.08 (0.02-0.24) | 0.04 | ND | ND | ND | 0.26 | 0.08 | 0.08 |
|  | 1995 | 0.10 (0.09-0.10) | 0.22 | 0.24 | 0.08 (0.03-0.40) | ND | ND | ND | ND | 0.09 | ND | 0.04 |
| Mirex | 1994 | 2.15 (2.10-2.20) | ND | ND | 0.03 (0.01-0.08) | ND | 0.07 | 0.08 | 0.17 | 0.15 | 0.08 | 0.11 |
|  | 1995 | $0.02(0.02-0.02)$ | 0.26 | 0.38 | 0.59 (0.06-0.94) | ND | ND | ND | ND | ND | ND | ND |
| o,p'-DDT | 1994 | 0.15 (0.13-0.17) | 0.33 | 0.24 | 0.93 (0.52-1.80) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.07 (0.07-0.07) | 0.23 | 0.19 | 0.28 (0.07-0.53) | ND | ND | ND | ND | ND | ND | ND |
| p,p'-DDT | 1994 | 0.15 (0.13-0.17) | 1.6 | 1.7 | 1.41 (0.09-3.10) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.06 (0.06-0.06) | 0.96 | 0.72 | 0.26 (0.06-0.53) | ND | ND | ND | ND | ND | ND | ND |
| p,p'-DDD | 1994 | 0.08 (0.07-0.09) |  | 1.6 | 0.22 (0.14-0.37) | ND | ND | ND | ND | 0.92 | ND | ND |
|  | 1995 | 0.03 (0.03-0.03) | 5.90 | 4.80 | 0.16 (0.05-0.32) | ND | 0.20 | ND | ND | 0.53 | 0.25 | 0.53 |
| p,p'-DDE | 1994 | 0.08 (0.07-0.08) | 16 | 8.6 | 0.15 (0.11-0.19) | 0.31 | 0.60 | 0.49 | 1.00 | 11.00 | 2.10 | 2.50 |
|  | 1995 | 0.02 (0.02-0.02) | 11.00 | 6.50 | 0.16 (0.07-0.28) | 0.45 | 0.85 | 0.57 | 0.77 | 4.35 | 1.50 | 3.50 |
| cis-Chlordane | 1994 | 0.37 (0.29-0.44) | ND | 0.49 | 0.13 (0.07-0.23) | ND | ND | ND | ND | 0.22 | ND | ND |
|  | 1995 | 0.05 (0.04-0.06) | 0.66 | 0.44 | 0.18 (0.05-0.28) | ND | ND | ND | ND | 0.13 | ND | ND |
| Oxychlordane | 1994 | 0.62 (0.56-0.67) | ND | ND | 0.26 (0.12-0.43) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.36 (0.34-0.38) | ND | ND | 1.57 (0.04-3.50) | ND | ND | ND | ND | ND | ND | ND |
| trans-Chlordane | 1994 | 0.29 (0.28-0.29) | ND | ND | 0.12 (0.07-0.26) | ND | ND | ND | ND | ND | ND | ND |
|  | 1995 | 0.05 (0.04-0.06) | 0.17 | 0.18 | 0.17 (0.05-0.27) | ND | ND | ND | ND | ND | ND | ND |
| trans-Nonachlor | 1994 | 0.25 (0.23-0.26) | 1.1 | 0.79 | 0.15 (0.09-0.32) | 0.17 | ND | ND | 0.29 | 0.60 | 0.22 | 0.27 |
|  | 1995 | 0.04 (0.03-0.04) | 1.00 | 0.56 | 0.17 (0.06-0.33) | ND | 0.12 | 0.02 | ND | 0.27 | ND | 0.20 |
| Total Toxaphene | 1994 | 0.15 (0.14-0.15) | 2.7 | 2 | 0.57 (0.08-1.40) | 0.45 | 0.77 | 0.21 | ND | 5.30 | ND | 1.23 |
|  | 1995 | 0.51 (0.47-0.55) | 4.80 | 1.50 | 0.21 (0.13-0.35) | ND | 1.50 | ND | ND | 2.40 | 0.59 | 3.00 |

Table 53. PAHs analyzed in fish tissue from the lower Fraser River, 1994, with the number of aromatic rings in the molecule and the molecular weight.

| Low Molecular Weight |  |  | High Molecular Weight |  |  |
| :--- | :---: | :---: | :--- | :---: | :---: |
|  | \# Rings | Mol. Wt. |  | \# Rings | Mol. Wt. |
| Fluorene | 2 | 166 | 3-MeCholanthrene | 4 | 269 |
| Naphthalene | 2 | 128 | $7,12-M e C h o l a n t h r e n e ~$ | 4 | 284 |
| Acenaphthene | 3 | 154 | Benz(a)anthracene | 4 | 228 |
| Acenaphthylene | 3 | 152 | Chrysene | 4 | 228 |
| Anthracene | 3 | 178 | Fluoranthene | 4 | 202 |
| Phenanthrene | 3 | 178 | Pyrene | 4 | 202 |
|  |  |  |  | Benzo(a)pyrene | 5 |

Table 54. Detection frequencies for PAHs in tissues from fish in the lower Fraser River, 1994; shown is the frequency (in \%) and (number of detections/number of samples); PAHs in bold were not detected.

| Compound | Liver | Muscle |
| :--- | :---: | :---: |
| 3-MeCholanthrene | $\mathbf{0 . 0}(\mathbf{0} / 8)$ | $\mathbf{0 . 0}(\mathbf{0} / 34)$ |
| 7,12-MeCholanthrene | $0.0(0 / 8)$ | $91.2(31 / 34)$ |
| Acenaphthene | $100.0(8 / 8)$ | $100.0(34 / 34)$ |
| Acenaphthylene | $100.0(8 / 8)$ | $70.6(24 / 34)$ |
| Anthracene | $100.0(8 / 8)$ | $94.1(32 / 34)$ |
| Benz(a)anthracene | $12.5(1 / 8)$ | $20.6(7 / 34)$ |
| Benzo(a)pyrene | $0.0(0 / 8)$ | $2.9(1 / 34)$ |
| Benzo(c)phenanthrene | $0.0(0 / 8)$ | $2.9(1 / 34)$ |
| Benzo(e)pyrene | $0.0(0 / 8)$ | $2.9(1 / 34)$ |
| Benzo(ghi)perylene | $0.0(0 / 8)$ | $5.9(2 / 34)$ |
| Benzofluoranthenes | $0.0(0 / 8)$ | $11.8(4 / 34)$ |
| Chrysene | $37.5(3 / 8)$ | $47.1(16 / 34)$ |
| Dibenz(ah)anthracene | $0.0(0 / 8)$ | $2.9(1 / 34)$ |
| Dibenzo(a1)pyrene | $\mathbf{0 . 0 ( 0 / 8 )}$ | $\mathbf{0 . 0 ( 0 / 3 4 )}$ |
| Dibenzo(ah)pyrene | $\mathbf{0 . 0}(\mathbf{0 / 8 )}$ | $\mathbf{0 . 0}(0 / 34)$ |
| Dibenzo(ai)pyrene | $\mathbf{0 . 0 ( 0 / 8 )}$ | $\mathbf{0 . 0 ( 0 / 3 4 )}$ |
| Fluoranthene | $100.0(8 / 8)$ | $100.0(34 / 34)$ |
| Fluorene | $100.0(8 / 8)$ | $100.0(34 / 34)$ |
| ldeno(1,2,3,cd)pyrene | $0.0(0 / 8)$ | $2.9(1 / 34)$ |
| Naphthalene | $100.0(8 / 8)$ | $100.0(34 / 34)$ |
| Perylene | $12.5(1 / 8)$ | $2.9(1 / 34)$ |
| Phenanthrene | $100.0(8 / 8)$ | $100.0(34 / 34)$ |
| Pyrene | $100.0(8 / 8)$ | $100.0(34 / 34)$ |

Table 55. PAH concentrations in peamouth chub and starry flounder muscle from the lower Fraser River, 1994; concentrations in ng/g wet weight, means of 5 composite samples, with associated standard error.

| Compound | Peamouth Chub |  |  |  |  | Starry Flounder |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Mission | Barnston | Main Arm | North Arm | Main Arm |  |
|  |  |  |  |  |  |  |
| 3-MeCholanthrene | $<0.54$ | $<0.10$ | $<0.93$ | $<0.54$ | $<0.40$ |  |
| 7,12-MeCholanthrene | $3.90(0.76)$ | $4.52(0.20)$ | $3.51(0.33)$ | $3.26(0.14)$ | $0.91(0.23)$ | $1.12(0.10)$ |
| Acenaphthene | $2.00(0.32)$ | $2.56(0.95)$ | $4.40(1.66)$ | $5.60(1.83)$ | $1.70(0.37)$ | $2.10(0.40)$ |
| Acenaphthylene | $0.18(0.06)$ | $0.13(0.02)$ | $0.25(0.06)$ | $0.26(0.08)$ | $0.33(0.06)$ | $0.25(0.03)$ |
| Anthracene | $0.44(0.07)$ | $0.31(0.11)$ | $0.86(0.29)$ | $0.62(0.10)$ | $0.43(0.08)$ | $0.46(0.07)$ |
| Benz(a)anthracene | $0.20(0.03)$ | $<0.20$ | $<0.20$ | $0.87(0.78)$ | $0.04(0.01)$ | $0.04(0.01)$ |
| Benzo(a)pyrene | $<0.10$ | $<0.30$ | $<0.30$ | $0.53(0.37)$ | $<0.09$ | $<0.10$ |
| Benzo(c)phenanthrene | $<0.04$ | $<0.14$ | $<0.18$ | $0.11(0.03)$ | $<0.03$ | $<0.02$ |
| Benzo(e)pyrene | $<0.10$ | $<0.30$ | $<0.20$ | $0.89(0.78)$ | $<0.08$ | $<0.10$ |
| Benzo(ghi)perylene | $<0.20$ | $<0.60$ | $<0.40$ | $0.76(0.56)$ | $<0.10$ | $<0.20$ |
| Benzofluoranthenes | $0.18(0.03)$ | $<0.30$ | $0.21(0.06)$ | $1.69(1.58)$ | $<0.07$ | $<0.10$ |
| Chrysene | $<0.20$ | $<0.20$ | 0.18 | $1.34(1.17)$ | $0.11(0.03)$ | $0.17(0.03)$ |
| Dibenz(ah)anthracene | $<0.60$ | $<2.0$ | $<1.0$ | $1.14(0.72)$ | $<0.20$ | $<0.50$ |
| Dibenzo(a1)pyrene | $<0.86$ | $<6.4$ | $<0.92$ | $<0.52$ | $<0.20$ | $<1.0$ |
| Dibenzo(ah)pyrene | $<7.4$ | $<4.6$ | $<4.1$ | $<2.30$ | $<0.42$ | $<14$ |
| Dibenzo(ai)pyrene | $<5.4$ | $<0.24$ | $<2.8$ | $<1.50$ | $<0.30$ | $<9.2$ |
| Fluoranthene | $0.60(0.08)$ | $0.76(0.21)$ | $1.94(0.58)$ | $3.58(0.94)$ | $1.16(0.21)$ | $1.30(0.20)$ |
| Fluorene | $1.80(0.20)$ | $1.96(0.57)$ | $3.10(1.23)$ | $3.80(1.32)$ | $1.80(0.20)$ | $1.26(0.31)$ |
| ldeno(1,2,3,cd)pyrene | $<0.30$ | $<0.80$ | $<0.50$ | $0.80(0.55)$ | $<0.10$ | $<0.30$ |
| Naphthalene | $3.12(0.39)$ | $3.20(0.49)$ | $4.00(0.55)$ | $4.00(0.84)$ | $3.00(0.32)$ | $2.50(0.32)$ |
| Perylene | $<0.10$ | $<0.30$ | $<0.30$ | $0.73(0.57)$ | $<0.08$ | $<0.10$ |
| Phenanthrene | $2.60(0.40)$ | $2.70(0.54)$ | $5.70(1.64)$ | $8.80(3.22)$ | $3.80(0.20)$ | $3.20(0.37)$ |
| Pyrene | $0.38(0.04)$ | $0.45(0.13)$ | $1.03(0.27)$ | $2.24(0.83)$ | $0.58(0.08)$ | $0.69(0.09)$ |

Table 56. PAH concentrations in peamouth chub and starry flounder liver from the lower Fraser River, 1994; concentrations in ng/g wet weight, means of 5 composite samples.

|  | Peamouth Chub |  |  |  | Starry Flounder |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Mission | Barnston | Main Arm | North Arm | Main Arm | North Arm |
| 3-MeCholanthrene | <0.76 | <0.42 | $<0.47$ | <0.64 | <1.20 | <1.20 |
| 7,12-MeCholanthrene | <0.93 | <0.47 | <0.88 | <0.72 | <1.80 | <1.20 |
| Acenaphthene | 29 | 29 | 51 | 54 | 24 | 16 |
| Acenaphthylene | 2 | 1 | 2.5 | 3 | 2 | 1 |
| Anthracene | 3 | 3 | 6 | 8 | 3 | 2 |
| Benz(a)anthracene | <0.20 | <0.08 | 0.35 | <0.10 | <0.30 | <0.20 |
| Benzo(a)pyrene | <0.20 | <0.08 | <0.10 | <0.10 | <0.40 | <2.00 |
| Benzo(c)phenanthrene | <0.19 | <0.10 | <0.18 | <0.15 | <0.37 | <0.26 |
| Benzo(e)pyrene | <0.10 | <0.07 | <0.10 | <0.10 | <0.30 | <2.00 |
| Benzo(ghi)perylene | <0.20 | <0.08 | <0.10 | <0.10 | <0.40 | <0.20 |
| Benzofluoranthenes | <0.10 | <0.06 | <0.10 | <0.10 | <0.30 | <2.00 |
| Chrysene | <0.20 | <0.09 | 0.6 | <0.10 | 2 | 1 |
| Dibenz(ah)anthracene | <0.20 | <0.10 | <0.20 | <0.20 | <0.60 | <0.30 |
| Dibenzo(a1)pyrene | <0.28 | <0.13 | <0.19 | <0.20 | <0.49 | <0.28 |
| Dibenzo(ah)pyrene | <0.61 | <0.29 | <0.42 | <0.42 | <1 | <0.61 |
| Dibenzo(ai)pyrene | <0.37 | <0.17 | <0.25 | <0.26 | <0.64 | <0.37 |
| Fluoranthene | 7 | 12 | 17.5 | 30.5 | 10 | 12 |
| Fluorene | 26 | 22 | 40.5 | 45.5 | 20 | 13 |
| Ideno(1,2,3,cd)pyrene | <0.30 | <0.20 | <0.30 | <0.20 | <0.50 | <0.27 |
| Naphthalene | 25 | 25 | 31.5 | 34 | 19 | 13 |
| Perylene | <0.20 | <0.08 | <0.10 | <0.10 | <0.40 | 1.9 |
| Phenanthrene | 34 | 26 | 59.5 | 74.5 | 45 | 32 |
| Pyrene | 3 | 3 | 4.5 | 8.5 | 2 | 6 |

Table 57. PAHs measured in fish tissues from the lower Fraser River, 1988, compared with levels measured in 1994. Values indicate maximum level measured in 1988 with the maximum 1994 value shown in brackets in $\mathrm{ng} / \mathrm{g}$ wet weight. The comparison is restricted to compounds detected in the 1988 survey reported by (Swain and Walton 1989).

|  | $\begin{array}{\|c\|} \hline \text { 1988 D.L } \\ \text { ng/g w.w. } \\ \hline \end{array}$ | Peamouth Liver |  |  | Starry Liver |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Barnston | Main Arm | North Arm | Main Arm | North Arm |
| Acenapthylene | 20 | ND (1) | 24 (3) | 46 (3) | 81 (2) | ND (1) |
| Acenaphthene | 20 | ND (29) | ND (70) | 27 (58) | ND (24) | ND (16) |
| Benzo(a)anthracene | 50 | $93(<0.1)$ | ND (0.9) | ND (<0.1) | ND (<0.3) | ND (0.2) |
| Chrysene | 20 | ND (<0.1) | 77 (1) | ND (<0.1) | ND (2) | ND (1) |
| Fluorene | 20 | ND (22) | 24 (56) | 22 (48) | 48 (20) | ND (13) |
| Napthalene | 20 | 160 (25) | 100 (42) | 140 (34) | 120 (19) | 70 (13) |
| Phenanthrene | 20 | 60 (26) | 50 (84) | 43 (79) | 43 (79) | ND (32) |
| Pyrene | 50 | 70 (3) | 120 (5) | ND (10) | ND (2) | ND (6) |
|  |  | Peamouth Mu:scle |  |  | Starry Muscle |  |
| Fluoranthene | 50 | ND (2) | ND (4) | ND (<1.5) | ND (2) | 110 (2) |
| Phenanthrene | 20 | ND (4) | ND (12) | 100 (3) | ND (4) | 150 (4) |

D.L = detection limit

Table 58. Detection limits for trace metals in tissues at the two analytical laboratories used in the course of this study.

|  | National Laboratory ${ }^{(1)}$ | Quanta Trace $^{(2)}$ |
| :--- | :---: | :---: |
| Arsenic $(\mu \mathrm{g} / \mathrm{g})$ | 0.05 | 0.02 |
| Cadmuim $(\mu \mathrm{g} / \mathrm{g})$ | 0.01 | 0.01 |
| Chromium $(\mu \mathrm{g} / \mathrm{g})$ | 0.2 | 0.02 |
| Cobalt $(\mu \mathrm{g} / \mathrm{g})$ | $0.5(1994) / 0.2(1995)$ | 0.02 |
| Copper $(\mu \mathrm{g} / \mathrm{g})$ | 0.2 | 0.04 |
| Iron $(\mu \mathrm{g} / \mathrm{g})$ | 0.2 | 0.04 |
| Lead $(\mu \mathrm{g} / \mathrm{g})$ | 0.05 | 0.02 |
| Manganese $(\mu \mathrm{g} / \mathrm{g})$ | 0.2 | 0.02 |
| Mercury $(\mu \mathrm{g} / \mathrm{g})$ | 0.01 | 0.05 |
| Nickel $(\mu \mathrm{g} / \mathrm{g})$ | 0.02 | 0.04 |
| Selenium $(\mu \mathrm{g} / \mathrm{g})$ | 0.05 | 0.02 |
| Zinc $(\mu \mathrm{g} / \mathrm{g})$ | 0.2 | 0.1 |

[^1]Table 59. Trace metal concentrations in peamouth chub muscle from areas in the Fraser basin, 1994 and 1995. Concentrations shown are the mean of $4-5$ composite samples of 5 fish each, with the range of measured values. Units are $\mu \mathrm{g} / \mathrm{g}$ wet weight.

Peamouth Muscle

|  | Year | Nechako | Hansard | Woodpecker | Marguerite | Agassiz | Mission | Barnston | Main Arm | North Arm | N. Thompson | Thompson |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Arsenic | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{gathered} <0.05 \\ 0.03(<0.05-0.05) \end{gathered}$ | $\begin{aligned} & 0.03(<0.05-0.05) \\ & 0.04<0.05-0.07) \end{aligned}$ | $\begin{gathered} <0.05 \\ 0.04(<0.05-0.05) \end{gathered}$ | $\begin{aligned} & <0.05 \\ & <0.05 \end{aligned}$ | $\begin{aligned} & 0.11(0.07-0.15) \\ & 0.17(0.13-0.20) \\ & \hline \end{aligned}$ | 0.06 (0.04-0.08) | 0.03 (<0.02-0.10) | $\begin{aligned} & 0.03(<0.02-0.06) \\ & 0.21(0.15-0.27) \end{aligned}$ | $\begin{aligned} & 0.02(<0.02-0.04) \\ & 0.17(0.11-0.21) \end{aligned}$ | $\begin{gathered} <0.05 \\ 0.05(<0.05-0.06) \end{gathered}$ | $\begin{gathered} <0.05 \\ 0.05(<0.05-0.06) \end{gathered}$ |
| Cadmium | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{gathered} <0.01 \\ 0.01(<0.01-0.02) \end{gathered}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $<0.01$ | $<0.01$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{gathered} <0.01 \\ 0.01(<0.01-0.02) \end{gathered}$ | $\begin{gathered} <0.01 \\ 0.01(<0.01-0.03) \end{gathered}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ |
| Chromium | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{array}{r} <0.20 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.20 \\ <0.20 \\ \hline \end{array}$ | $\begin{aligned} & <0.20 \\ & <0.20 \\ & \hline \end{aligned}$ | $\begin{array}{r} <0.20 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.20 \\ <0.20 \\ \hline \end{array}$ | 0.08 (0.07-0.09) | 0.08 (0.07-0.09) | $0.07(0.06-0.09)$ $<0.20$ | $\begin{gathered} 0.08(0.06-0.09) \\ <0.20 \end{gathered}$ | $\begin{array}{r} <0.20 \\ <0.20 \\ \hline \end{array}$ | $\begin{aligned} & <0.20 \\ & <0.20 \\ & \hline \end{aligned}$ |
| Cobalt | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | 0.02 (<0.02-0.03) | 0.02 (<0.02-0.03) | $\begin{aligned} & <0.02 \\ & <0.20 \\ & \hline \end{aligned}$ | $\begin{array}{r} <0.02 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ |
| Copper | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{aligned} & \hline 1.56(1.13-2.45) \\ & 0.28(0.25-0.31) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.96(0.47-2.05) \\ & 0.39(0.34-0.45) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 1.54(0.49-2.43) \\ & 0.51(0.49-0.52) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.62(0.50-0.70) \\ & 0.36(0.24-0.44) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.84(0.49-1.17) \\ & 0.54(0.52-0.57) \\ & \hline \end{aligned}$ | 0.48 (0.41-0.66) | 0.40 (0.35-0.44) | $\begin{gathered} 0.39(0.36-0.41) \\ 0.33(0.29-0.36) \\ \hline \end{gathered}$ | $\begin{aligned} & \hline 0.38(0.33-0.44) \\ & 0.40(0.35-0.44) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 1.45(0.97-2.61) \\ & 0.50(0.39-0.71) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 1.41(0.62-1.72) \\ & 0.49(0.44-0.55) \end{aligned}$ |
| Iron | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & 3.29(2.81-3.75) \\ & 2.81(2.50-3.36) \end{aligned}$ | $\begin{array}{r} 5.34(5.12-5.51) \\ 3.50(2.65-4.68) \\ \hline \end{array}$ | $\begin{aligned} & 5.11(4.46-5.98) \\ & 3.40(2.94-3.79) \\ & \hline \end{aligned}$ | $\begin{aligned} & 4.82(4.36-5.16) \\ & 2.97(2.47-3.55) \\ & \hline \end{aligned}$ | $\begin{aligned} & 5.92(4.83-7.22) \\ & 3.72(3.58-3.90) \\ & \hline \end{aligned}$ | 4.74 (3.50-7.10) | 6.68 (4.84-9.34) | $\begin{aligned} & 4.92(3.76-6.31) \\ & 4.67(3.85-5.44) \\ & \hline \end{aligned}$ | $\begin{aligned} & 5.43(4.08-9.07) \\ & 3.70(3.04-4.42) \\ & \hline \end{aligned}$ | $\begin{aligned} & 6.26(4.62-8.53) \\ & 3.95(3.61-4.62) \\ & \hline \end{aligned}$ | 5.10 (4.23-6.25) <br> $3.46(3.33-3.63)$ |
| Lead | $\begin{aligned} & 19001 \\ & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | <0.02 | <0.02 | $\begin{aligned} & <0.02 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.02 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ |
| Manganese | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{aligned} & \hline 0.34(0.22-0.40) \\ & 0.25(0.21-0.30) \\ & \hline \end{aligned}$ | $\begin{gathered} 0.36(0.34-0.42) \\ 0.27(\leqslant 0.20-0.28) \end{gathered}$ | $\begin{aligned} & 0.28(<0.20-0.42) \\ & 0.25(<0.20-0.30) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.39(0.30-0.48) \\ & 0.31(0.28-0.36) \\ & \hline \end{aligned}$ | $\begin{gathered} 0.34(0.28-0.42) \\ 0.21(<0.20-0.27) \\ \hline \end{gathered}$ | 0.25 (0.21-0.30) | 0.31 (0.23-0.37) | $\begin{aligned} & 0.24(0.20-0.32) \\ & 0.24(0.22-0.27) \\ & \hline \end{aligned}$ | $\begin{gathered} 0.22(0.20-0.27) \\ 0.17(<0.20-0.25) \\ \hline \end{gathered}$ | $\begin{aligned} & 0.32(<0.20-0.46) \\ & 0.17(<0.20-0.28) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.14(<0.20-0.25) \\ & 0.16(<0.20-0.22) \end{aligned}$ |
| Mercury | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.22(0.19-0.26) \\ & 0.30(0.25-0.33) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.19(0.12-0.25) \\ & 0.12(0.11-0.14) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.07(0.05-0.10) \\ & 0.10(0.08-0.12) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.17(0.11-0.20) \\ & 0.09(0.07-0.12) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.24(0.18-0.34) \\ & 0.18(0.18-0.19) \\ & \hline \end{aligned}$ | 0.14 (0.11------26) | 0.13 (0.11-0.15) | $\begin{aligned} & 0.14(0.11-0.17) \\ & 0.23(0.18-0.26) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.11(0.10-0.15) \\ & 0.17(0.16-0.20) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.14(0.12-0.16) \\ & 0.14(0.10-0.18) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.12(0.11-0.14) \\ & 0.14(0.12-0.16) \\ & \hline \end{aligned}$ |
| Nickel | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{aligned} & 0.25(0.15-0.42) \\ & 0.02(0.02-0.02) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.13(0.04-0.37) \\ & 0.01(<0.02-0.03) \end{aligned}$ | $\begin{aligned} & 0.30(<0.20-0.44) \\ & 0.03(<0.02-0.07) \end{aligned}$ | $\begin{aligned} & 0.03(0.02-0.05) \\ & 0.01(<0.02-0.02) \end{aligned}$ | $\begin{aligned} & 0.09(0.02-0.14) \\ & 0.01(<0.02-0.02) \\ & \hline \end{aligned}$ | <0.04 | 0.11 (<0.04-0.06) | $\begin{aligned} & 0.10(<0.04-0.07) \\ & 0.04(<0.02-0.07) \end{aligned}$ | $\begin{aligned} & 0.12(<0.04-0.10) \\ & 0.04(<0.02-0.06) \end{aligned}$ | $\begin{aligned} & 0.34(0.16-0.52) \\ & 0.03(0.02-0.03) \end{aligned}$ | $\begin{gathered} 0.47(0.13-1.05) \\ 0.01(<0.02-0.03) \end{gathered}$ |
| Selenium | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{aligned} & 0.21(0.14-0.31) \\ & 0.17(0.13-0.20) \end{aligned}$ | $\begin{array}{r} 0.37(0.22-0.46) \\ 0.34(0.30-0.39) \\ \hline \end{array}$ | $\begin{array}{r} 0.39(0.33-0.46) \\ 0.44(0.36-0.49) \\ \hline \end{array}$ | $\begin{array}{r} 0.47(0.43-0.51) \\ 0.53(0.46-0.61) \\ \hline \end{array}$ | $\begin{array}{r} 0.41(0.38-0.43) \\ 0.54(0.50-0.59) \\ \hline \end{array}$ | 0.47 (0.30-0.70) | 0.38 (0.30-0.50) | $\begin{aligned} & 0.51(0.20-0.80) \\ & 0.49(0.39-0.58) \end{aligned}$ | $\begin{array}{r} 0.37(0.30-0.50) \\ 0.43(0.30-0.60) \\ \hline \end{array}$ | $\begin{array}{r} 0.42(0.41-0.44) \\ 0.47(0.45-0.49) \\ \hline \end{array}$ | $\begin{aligned} & 0.33(0.29-0.35) \\ & 0.36(0.31-0.40) \end{aligned}$ |
| Zinc | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{aligned} & \hline 7.85(7.15-8.92) \\ & 8.90(8.02-9.81) \\ & \hline \end{aligned}$ | $\begin{gathered} 11.65(10.30-13.60) \\ -10.23(8.25-13.10) \\ \hline \end{gathered}$ | $\begin{array}{r} 10.76(7.64-14.10) \\ 11.13(10.40-12.20) \\ \hline \end{array}$ | $\begin{gathered} 12.80(11.60-13.90) \\ -9.57(8.63-10.70) \\ \hline \end{gathered}$ | $\begin{array}{r} 10.47(8.46-12.30) \\ 10.29(9.39-11.10) \\ \hline \end{array}$ | 9.58 (7.46-------11.60) | 8.93 (8.39-------33) | $\begin{aligned} & \hline 8.91(8.33-9.54) \\ & 8.74(8.38-9.01) \\ & \hline \end{aligned}$ | $\begin{gathered} 7.98(7.35-8.91) \\ 10.27(8.89-11.50) \\ \hline \end{gathered}$ | $\begin{aligned} & 10.16(7.70-13.70) \\ & 10.92(9.96-11.60) \\ & \hline \end{aligned}$ | $\begin{aligned} & 9.27(7.77-10.10) \\ & 9.35(9.03-9.64) \\ & \hline \end{aligned}$ |

Table 60. Trace metal concentrations in peamouth chub liver from areas in the Fraser basin, 1994 and 1995. Concentrations shown are the mean of 1-2 composite samples of up to 30 fish each. Units are $\mu \mathrm{g} / \mathrm{g}$ wet weight.

Peamouth Liver Tissues

|  | Year | Nechako | Hansard | Woodpecker | Marquerite | Agassiz | Mission | Barnston | Main Arm | North Arm | N. Thompson | Thompson |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Arsenic | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{array}{r} \text { n/a } \\ 0.04 \end{array}$ | $\begin{gathered} \mathrm{n} / \mathrm{a} \\ <0.05 \end{gathered}$ | $\begin{aligned} & \text { n/a } \\ & 0.07 \end{aligned}$ | $\begin{aligned} & \text { n/a } \\ & \text { NS } \end{aligned}$ | $\begin{aligned} & \text { n/a } \\ & \text { NS } \end{aligned}$ | <0.5 | <0.5 | $\begin{aligned} & 0.22 \\ & 0.29 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.55 \end{aligned}$ | $\begin{array}{r} \text { n/a } \\ 0.37 \\ \hline \end{array}$ | $\begin{array}{r} \mathrm{n} / \mathrm{a} \\ 0.15 \\ \hline \end{array}$ |
| Cadmium | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.12 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.29 \\ & 0.22 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.21 \\ & 0.14 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.12 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.32 \\ & 0.11 \\ & \hline \end{aligned}$ | 0.29 | 0.34 | $\begin{gathered} 0.24 \\ 0.10 \\ \hline \end{gathered}$ | $\begin{aligned} & \\ & 0.20 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.20 \\ & 0.20 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.16 \\ & 0.16 \\ & \hline \end{aligned}$ |
| Chromium | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{aligned} & <0.2 \\ & \mathrm{NS} \end{aligned}$ | $\begin{array}{r} <0.2 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.2 \\ <200 \\ \hline \end{array}$ | $\begin{gathered} <0.2 \\ \mathrm{NS} \\ \hline \end{gathered}$ | $\begin{array}{r} <0.2 \\ <1.00 \end{array}$ | 0.3 | 0.1 | $\begin{gathered} 0.15 \\ <2.00 \\ \hline \end{gathered}$ | $\begin{aligned} & 0.2 \\ & N S \end{aligned}$ | $\begin{gathered} <0.2 \\ \mathrm{NS} \\ \hline \end{gathered}$ | $\begin{gathered} <0.2 \\ \mathrm{NS} \\ \hline \end{gathered}$ |
| Cobalt | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & <0.5 \\ & \text { NS } \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.43 \end{aligned}$ | $\begin{array}{r} <0.5 \\ <2.00 \end{array}$ | $\begin{gathered} <0.5 \\ \text { NS } \end{gathered}$ | $\begin{array}{r} <0.5 \\ -1.00 \\ \hline \end{array}$ | 0.2 | 0.2 | $\begin{gathered} 0.35 \\ <2.00 \end{gathered}$ | $\begin{gathered} 0.55 \\ \text { NS } \end{gathered}$ | $\begin{aligned} & \hline<0.5 \\ & N \end{aligned}$ | $\begin{gathered} <0.5 \\ \mathrm{NS} \end{gathered}$ |
| Copper | $\begin{array}{r} 1994 \\ -1995 \\ \hline \end{array}$ | $\begin{gathered} 4.09 \\ \text { NS } \end{gathered}$ | $\begin{aligned} & 3.27 \\ & 2.71 \end{aligned}$ | $\begin{array}{r} 3.25 \\ 3.20 \\ \hline \end{array}$ | $\begin{gathered} 2.94 \\ \text { NS } \end{gathered}$ | $\begin{array}{r} 3.52 \\ 2.24 \\ \hline \end{array}$ | 4.30 | 10.50 | $\begin{array}{r} 4.75 \\ 5.70 \\ \hline \end{array}$ | $\begin{gathered} 5.86 \\ \text { NS } \end{gathered}$ | $\begin{gathered} 3.23 \\ \mathrm{NS} \end{gathered}$ | $\begin{aligned} & 3.32 \\ & \text { NS } \end{aligned}$ |
| Iron | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & 139 \\ & \mathrm{NS} \end{aligned}$ | $\begin{gathered} 171 \\ 14700 \\ \hline \end{gathered}$ | $\begin{gathered} 84 \\ 106.00 \\ \hline \end{gathered}$ | $\begin{gathered} 116 \\ \text { NS } \end{gathered}$ | $\begin{gathered} 246 \\ 71.93 \\ \hline \end{gathered}$ | 178 | 153 | $\begin{gathered} 189 \\ \hline \\ \hline 0.050 \end{gathered}$ | $\begin{aligned} & 191 \\ & \text { NS } \end{aligned}$ | $\begin{aligned} & 213 \\ & \text { NS } \end{aligned}$ | $\begin{aligned} & 102 \\ & N S \end{aligned}$ |
| Lead | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{aligned} & <0.5 \\ & 0.20 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.15 \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.04 \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.04 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.5 \\ & N S \\ & \hline \end{aligned}$ | <0.5 | <0.5 | $\begin{aligned} & <0.5 \\ & 0.13 \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.10 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.06 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.5 \\ & 0.06 \\ & \hline \end{aligned}$ |
| Manganese | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{gathered} 1.42 \\ \mathrm{NS} \\ \hline \end{gathered}$ | $\begin{aligned} & 1.74 \\ & 1.19 \end{aligned}$ | $\begin{gathered} 1.85 \\ \hline \\ \hline \end{gathered}$ | $\begin{gathered} 1.51 \\ 1.5 S \\ \hline \end{gathered}$ | $\begin{array}{r} 1.50 \\ 1.10 \\ \hline \end{array}$ | 1.20 | 1.30 | $\begin{gathered} 1.36 \\ -2.00 \\ \hline \end{gathered}$ | $\begin{aligned} & 1.50 \\ & \text { NS } \end{aligned}$ | $\begin{aligned} & 1.54 \\ & \mathrm{NS} \end{aligned}$ | $\begin{array}{r} 1.54 \\ \text { NS } \end{array}$ |
| Mercury | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.19 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.10 \\ & 0.02 \\ & \hline \end{aligned}$ | $\begin{array}{r} 0.04 \\ 0.13 \\ \hline \end{array}$ | $\begin{aligned} & 0.06 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{array}{r} \hline 0.13 \\ <0.10 \\ \hline \end{array}$ | 0.-30---- | ${ }^{0.20}$ | $\begin{gathered} \hline 0.25 \\ <0.20 \\ \hline \end{gathered}$ | $\begin{aligned} & 0.20 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.07 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.12 \\ & \text { NS } \\ & \hline \end{aligned}$ |
| Nickel | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{gathered} <0.2 \\ N S \end{gathered}$ | $\begin{gathered} 0.09 \\ <0.05 \end{gathered}$ | $\begin{gathered} 0.04 \\ <0.50 \\ \hline \end{gathered}$ | $\begin{gathered} 0.03 \\ \text { NS } \end{gathered}$ | $\begin{gathered} 0.03 \\ <0.25 \end{gathered}$ | 0.10 | <0.10 | $\begin{gathered} 0.05 \\ <0.50 \end{gathered}$ | $\begin{aligned} & 0.10 \\ & \text { N } \end{aligned}$ | $\begin{aligned} & 0.09 \\ & \text { N } \end{aligned}$ | $\begin{gathered} 0.05 \\ \text { NS } \end{gathered}$ |
| Selenium | $\begin{array}{r} 1994 \\ 1995 \\ \hline \end{array}$ | $\begin{array}{r} \text { n/a } \\ 1.02 \end{array}$ | $\begin{array}{r} \mathrm{n} / \mathrm{a} \\ 1.53 \\ \hline \end{array}$ | $\begin{array}{r} \mathrm{n} / \mathrm{a} \\ 1.92 \\ \hline \end{array}$ | $\begin{aligned} & \text { n/a } \\ & \text { NS } \end{aligned}$ | $\begin{aligned} & \mathrm{n} / \mathrm{a} \\ & \mathrm{NS} \end{aligned}$ | ${ }^{2 .}-{ }_{-}$ | ${ }^{0.8}$ | $\begin{gathered} 1.9 \\ 1.67 \\ \hline \end{gathered}$ | $\begin{gathered} 0.8 \\ 1.24 \\ \hline \end{gathered}$ | $\begin{array}{r} \mathrm{n} / \mathrm{a} \\ 1.50 \\ \hline \end{array}$ | $\begin{array}{r} \mathrm{n} / \mathrm{a} \\ 1.27 \\ \hline \end{array}$ |
| Zinc | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & 18.9 \\ & \mathrm{NS} \end{aligned}$ | $\begin{gathered} 22.3 \\ 19.00 \\ \hline \end{gathered}$ | $\begin{gathered} 21.9 \\ 22.70 \\ \hline \end{gathered}$ | $\begin{array}{r} 22.5 \\ \text { NS } \end{array}$ | $\begin{gathered} 24.3 \\ 17.13 \\ \hline \end{gathered}$ | 17.6 | 18.7 | $\begin{gathered} 18.5 \\ 19.20 \\ \hline \end{gathered}$ | $\begin{aligned} & 17.6 \\ & \text { NS } \\ & \hline \end{aligned}$ | $\begin{gathered} 23.1 \\ \text { NS } \end{gathered}$ | $\begin{gathered} 19.4 \\ \text { NS } \end{gathered}$ |

Table 61. Trace metal concentrations in starry flounder and mountain whitefish muscle from areas in the Fraser basin, 1994 and 1995. Concentrations shown are the mean of $4-5$ composite samples of 5 fish each, with the range of measured values. Units are $\mu \mathrm{g} / \mathrm{g}$ wet weight.

| Starry Flounder |  |  |  |
| :---: | :---: | :---: | :---: |
| Metal | Year | Main Arm | North Arm |
| Arsenic | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.37(0.26-0.50) \\ & 0.51(0.45-0.57) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.20(0.02-0.32) \\ & 0.55(0.45-0.64) \\ & \hline \end{aligned}$ |
| Cadmium | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\left.\begin{array}{c} <0.01 \\ 0.01 \end{array}<0.01-0.02\right)$ | $\begin{gathered} <0.01 \\ 0.01(<0.01-0.01) \end{gathered}$ |
| Chromium | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{array}{cc} \hline 0.07(0.06-0.10) \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{cc} \hline 0.08(0.06-0.10) \\ <0.20 \\ \hline \end{array}$ |
| Cobalt | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.02 \\ & <0.20 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.02 \\ & <0.20 \\ & \hline \end{aligned}$ |
| Copper | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.25(0.22-0.30) \\ & 0.29(0.27-0.32) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.27(0.23-0.35) \\ & 0.26(0.23-0.30) \\ & \hline \end{aligned}$ |
| Iron | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{array}{r} 3.77(2.43-4.82) \\ 3.27(2.22-4.13) \\ \hline \end{array}$ | $\begin{aligned} & 3.50(2.75-4.57) \\ & 2.29(1.84-3.29) \\ & \hline \end{aligned}$ |
| Lead | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.02 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.02 \\ & <0.05 \\ & \hline \end{aligned}$ |
| Manganese | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{array}{r} \hline 0.28(0.20-0.44) \\ 0.25(<0.20-0.48) \\ \hline \end{array}$ | $\begin{array}{r} 0.31(0.25-0.41) \\ 0.25(<0.20-0.32) \\ \hline \end{array}$ |
| Mercury | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.09(0.06-0.10) \\ & 0.07(0.05-0.08) \\ & \hline \end{aligned}$ | $\begin{array}{ll} 0.08(0.05-0.11) \\ 0.07 & (0.06-0.08) \\ \hline \end{array}$ |
| Nickel | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.09(<0.04-0.20) \\ & 0.08(0.04-0.13) \end{aligned}$ | $\begin{gathered} 0.07(<0.04-0.10) \\ 0.05(0.03-0.09) \\ \hline \end{gathered}$ |
| Selenium | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.48(0.40-0.60) \\ & 0.41(0.36-0.47) \\ & \hline \end{aligned}$ | $0.27(0.20-0.40)$ $0.41(0.35-0.49)$ |
| Zinc | $\begin{aligned} & 1994 \\ & 1995 \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 8.69 \text { (8.07-9.80) } \\ & 8.28(6.85-8.82) \\ & \hline \end{aligned}$ | $\begin{array}{r} 7.60(6.74-8.49) \\ 9.88(9.27-10.30) \\ \hline \end{array}$ |

Whitefish Muscle

| Nechako | Hansard | Woodpecker | Marguerite | Agassiz | N. Thompson | Thompson |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $\begin{gathered} <0.05 \\ 0.06(0.05-0.08) \end{gathered}$ | $\begin{gathered} <0.05 \\ 0.03(<0.05-0.05) \\ \hline \end{gathered}$ | $\begin{gathered} <0.05 \\ 0.07(0.06-0.10) \end{gathered}$ | $\begin{array}{r} 0.10(0.07-0.15) \\ 0.07(<0.05-0.11) \\ \hline \end{array}$ | $\begin{aligned} & 0.04(<0.05-0.08) \\ & 0.06(0.05-0.07) \\ & \hline \end{aligned}$ | $\begin{array}{r} <0.05 \\ <0.05 \\ \hline \end{array}$ | $\begin{aligned} & 0.07(0.05-0.08) \\ & 0.07(0.05-0.07) \\ & 0.07 \end{aligned}$ |
| $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{aligned} & <0.01 \\ & <0.01 \end{aligned}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ | $\begin{array}{r} <0.01 \\ <0.01 \\ \hline \end{array}$ |
| $\begin{gathered} 0.16(<0.20-0.29) \\ <0.20 \end{gathered}$ | $\begin{gathered} 0.14(<0.20-0.28) \\ <0.20 \\ \hline \end{gathered}$ | $\begin{gathered} 0.20(<0.20-0.35) \\ <0.20 \\ \hline \end{gathered}$ | $\begin{aligned} & <0.20 \\ & <0.20 \\ & \hline \end{aligned}$ | $\begin{gathered} 0.18(<0.20-0.27) \\ <0.20 \\ \hline \end{gathered}$ | $\begin{gathered} 0.26(<0.20-0.46) \\ <0.20 \\ \hline \end{gathered}$ | $\begin{gathered} 0.18(<0.20-0.24) \\ <0.20 \\ \hline \end{gathered}$ |
| $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ | $\begin{array}{r} <0.50 \\ <0.20 \\ \hline \end{array}$ |
| $\begin{aligned} & 1.10(0.58-1.51) \\ & 0.32(0.23-0.39) \end{aligned}$ | $\begin{aligned} & \hline 1.08(0.57-1.76) \\ & 0.45(0.41-0.50) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 1.48(0.73-2.14) \\ & 0.49(0.39-0.55) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 1.26(0.54-1.88) \\ & 0.46(0.43-0.50) \\ & \hline \end{aligned}$ | $\begin{aligned} & 1.16(0.62-1.97) \\ & 0.56(0.52-0.60) \end{aligned}$ | $\begin{aligned} & \hline 1.22(0.65-2.11) \\ & 0.45(0.38-0.53) \end{aligned}$ | $\begin{aligned} & \hline 1.45(0.59-1.79) \\ & 0.45(0.36-0.55) \\ & \hline \end{aligned}$ |
| $\begin{aligned} & \hline 6.03(4.46-8.30) \\ & 4.20(3.43-4.89) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 7.09(5.97-8.52) \\ & 5.50(5.16-5.82) \\ & \hline \end{aligned}$ | $\begin{aligned} & 6.26(4.96-9.11) \\ & 4.85(4.46-5.25) \\ & \hline \end{aligned}$ | $\begin{array}{ll}  & 5.03 \\ 4.56 & (4.38-5.41) \\ \hline \end{array}$ | $\begin{array}{r} \hline 7.95(6.08-10.30) \\ 4.48(4.21-4.90) \\ \hline \end{array}$ | $\begin{aligned} & 7.36(6.67-8.00) \\ & 5.07(4.50-5.83) \\ & \hline \end{aligned}$ | $\begin{aligned} & 5.05(4.29-6.49) \\ & 3.64(3.36-4.13) \\ & \hline \end{aligned}$ |
| $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ | $\begin{aligned} & <0.05 \\ & <0.05 \\ & \hline \end{aligned}$ |
| $\begin{gathered} 0.32(0.27-0.45) \\ 0.16(<0.20-0.25) \end{gathered}$ | $\begin{array}{r} 0.35(0.25-0.46) \\ 0.13(<0.20-0.23) \\ \hline \end{array}$ | $\begin{gathered} 0.24(0.20-0.30) \\ <0.20 \\ \hline \end{gathered}$ | $\begin{array}{r} \hline 0.32(0.21-0.40) \\ 0.18(<0.20-0.23) \\ \hline \end{array}$ | $\begin{gathered} 0.34(<0.20-0.45) \\ <0.20 \\ \hline \end{gathered}$ | $\begin{aligned} & 0.16(<0.20-0.40) \\ & 0.13(<0.20-0.21) \\ & \hline \end{aligned}$ | $\begin{gathered} 0.13(<0.20-0.24) \\ <0.20 \\ \hline \end{gathered}$ |
| $\begin{aligned} & 0.09(0.07-0.13) \\ & 0.10(0.08-0.16) \end{aligned}$ | $\begin{aligned} & 0.07(0.07-0.10) \\ & 0.06(0.05-0.07) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.06(0.06-0.07) \\ & 0.05(0.04-0.06) \\ & \hline \end{aligned}$ | $\begin{aligned} & \hline 0.02(0.01-0.03) \\ & 0.03(0.03-0.03) \\ & \hline \end{aligned}$ | $\begin{aligned} & 0.05(0.04-0.06) \\ & 0.04(0.04-0.05) \\ & \hline \end{aligned}$ | $\begin{array}{ll} 0.03 & (0.02-0.04) \\ 0.07 & (0.06-0.09) \end{array}$ | $\begin{aligned} & 0.04(0.04-0.05) \\ & 0.04(0.04-0.05) \\ & \hline \end{aligned}$ |
| $\begin{gathered} 0.21 \begin{array}{c} (0.12-0.29) \\ <0.02 \end{array} \\ \hline \end{gathered}$ | $\begin{array}{r} 0.20(0.05-0.44) \\ 0.01(<0.02-0.02) \\ \hline \end{array}$ | $\begin{gathered} 0.48(0.10-1.63) \\ <0.02 \\ \hline \end{gathered}$ | $\begin{gathered} 0.26 \begin{array}{c} (0.12-0.42) \\ <0.02 \end{array} \\ \hline \end{gathered}$ | $\begin{array}{r} 0.26(0.08-0.44) \\ 0.05(<0.02-0.14) \\ \hline \end{array}$ | $\begin{array}{r} 0.32(0.06-0.94) \\ 0.01(<0.02-0.02) \\ \hline \end{array}$ | $\begin{array}{r} 0.28(0.13-0.36) \\ 0.03(<0.02-0.07) \end{array}$ |
| $\begin{aligned} & 0.15(0.12-0.15) \\ & 0.17(0.14-0.20) \end{aligned}$ | $\begin{aligned} & \hline 0.45(0.42-0.49) \\ & 0.48(0.45-0.54) \\ & \hline \end{aligned}$ | 0.36 0.50 $0.0 .33-0.39)$ (0.0.65) | $0.46(0.40-0.57)$ $0.49(0.34-0.58)$ | 0.41 (0.34-0.53) $0.48(0.42-0.52)$ | $0.37(0.35-0.39)$ 0.45 (0.39-0.57) | $0.33(0.30-0.37)$ $0.43(0.39-0.46)$ |
| $\begin{aligned} & \hline 6.82(6.07-8.69) \\ & 6.35(5.48-7.16) \\ & \hline \end{aligned}$ | $\begin{aligned} & 5.40(4.76-5.85) \\ & 6.29(5.51-7.34) \\ & \hline \end{aligned}$ | $5.39(4.90-5.80)$ $6.11(4.74-6.65)$ | $5.52(4.09-7.40)$ $5.36(4.90-5.69)$ | $7.08(5.93-8.13)$ 7.36 (6.81-8.71) | 5.55 (4.94-6.38) $5.15(4.30-5.91)$ | $\begin{aligned} & 5.87(5.37-6.62) \\ & 6.64(6.05-7.15) \\ & \hline \end{aligned}$ |

Table 62. Trace metal concentrations in starry flounder and mountain whitefish liver from areas in the Fraser basin, 1994 and 1995. Concentrations shown are the mean of 1-2 composite samples of up to 30 fish each. Units are $\mu \mathrm{g} / \mathrm{gwet}$ weight.

| Starry Flounder Liver |  |  |  |
| :---: | :---: | :---: | :---: |
|  | Year | Main_Arm | North_Arm |
| Arsenic | 1994 | 0.3 | 0.15 |
|  | 1995 | NS | NS |
| Cadmium | 1994 | 2.33 | 2.69 |
|  | 1995 | 1.64 | 1.74 |
| Chromium | 1994 | 0.5 | 0.99 |
|  | 1995 | $<0.50$ | 0.44 |
| Cobalt | 1994 | 1.1 | 5.6 |
|  | 1995 | $\leq 0.50$ | $\leq 0.40$ |
| Copper | 1994 | 16.10 | 14.80 |
|  | 1995 | 14.80 | 16.20 |
| Iron | 1994 | 164 | 142 |
|  | 1995 | 154.00 | 133.00 |
| Lead | 1994 | <0.7 | 0.32 |
|  | 1995 | NS | NS |
| Manganese | 1994 | 1.8 | 2.0 |
|  | 1995 | 1.75 | 1.64 |
| Mercury | 1994 | <0.3 | <0.3 |
|  | 1995 | 0.08 | $<0.04$ |
| Nickel | 1994 | 0.3 | 0.7 |
|  | 1995 | $\leq 0.13$ | $<0.10$ |
| Selenium | 1994 | 3.2 | 1.8 |
|  | 1995 | NS | NS |
| Zinc | 1994 | 38.2 | 39.1 |
|  | 1995 | 38.50 | 39.90 |

Whitefish Liver

| Nechako | Hansard | Woodpecker | Marguerite | Agassiz | $N$. Thompson | Thompson |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| NS | $\leq 0.05$ | NS | $\leq 0.05$ | $\leq 0.05$ | $\leq 0.05$ | NS |
| 0.12 | 0.17 | 0.56 | 0.07 | 0.06 | 0.09 | 0.04 |
| 0.20 | 0.21 | 0.33 | NS | NS | 0.14 | 0.07 |
| <0.2 | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 | <0.2 |
| $<0.40$ | $\leq 2.00$ | $<0.50$ | NS | NS | $\leq 0.40$ | <0.40 |
| <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 |
| 0.31 | <2.00 | $<0.50$ | NS | NS | <0.40 | 0.46 |
| 2.09 | 2.38 | 2.36 | 3.13 | 2.83 | 1.89 | 2.43 |
| 2.66 | 2.40 | 2.92 | NS | NS | 1.60 | 2.22 |
| 92 | 64 | 73 | 71 | 51 | 41 | 69 |
| 104.45 | 57.95 | 85.45 | NS | NS | 43.60 | 54.50 |
| <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 | <0.5 |
| NS | 0.12 | NS | 0.05 | 0.03 | 0.09 | NS |
| 2.0 | 1.5 | 1.9 | 1.6 | 1.8 | 1.6 | 1.6 |
| 2.02 | <2.00 | 2.06 | NS | NS | 1.96 | 1.81 |
| 0.25 | 0.19 | 0.15 | 0.03 | 0.05 | 0.07 | 0.29 |
| 0.07 | 0.13 | <0.05 | NS | NS | 0.06 | <0.04 |
| 0.02 | 0.06 | 0.03 | 0.05 | <0.02 | 0.05 | 0.02 |
| $<0.10$ | <0.50 | $<0.13$ | NS | NS | <0.10 | $\leq 0.10$ |
| n/a | n/a | n/a | n/a | n/a | n/a | n/a |
| NS | 2.79 | NS | 3.79 | 2.75 | 1.91 | NS |
| 23.5 | 25.7 | 24.1 | 27.4 | 23.6 | 22.6 | 24.8 |
| 26.55 | 25.10 | 27.00 | NS | NS | 21.80 | 25.20 |

Table 63. Trace metals in peamouth chub and starry flounder muscle and liver from the Fraser River in 1988 (Swain and Walton 1989). Concentrations in $\mu \mathrm{g} / \mathrm{g}$ wet tissue weight.

|  |  | Muscle |  |  | FRAP 94/95 |
| :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | Barnston Island | North Arm | Main Arm |  |
| As | PC | 0.04-0.09 | 0.04-0.10 | 0.09-0.14 | <0.02-0.10 |
|  | SF |  | 0.04-0.57 | 0.18-0.40 | 0.02-0.64 |
| Cd | PC | <0.005 | <0.005 | <0.005 | <0.01-0.02 |
|  | SF |  | <0.005 | 0.03 | <0.01-0.02 |
| Cr | PC | 0.01-0.03 | 0.10-0.30 | 0.01-0.09 | 0.06-0.09 |
|  | SF |  | 0.16-2.50 | 0.55-2.03 | 0.06-0.10 |
| Cu | PC | 0.26-0.30 | 0.35-0.62 | 0.30-0.45 | 0.29-0.44 |
|  | SF |  | 0.26-1.08 | 0.70-1.17 | 0.22-0.35 |
| Fe | PC | 2.89-4.74 | 3.13-7.54 | 3.58-8.23 | 3.04-9.34 |
|  | SF |  | 5.33-156 | 56.9-129 | 1.84-4.82 |
| Pb | PC | 0.02-0.03 | 0.01-0.10 | ND - 0.01 | <0.02 |
|  | SF |  | 0.03-0.08 | 0.02-0.03 | <0.02 |
| Mn | PC | 0.17-0.29 | 0.19-0.42 | 0.13-0.31 | <0.02-0.37 |
|  | SF |  | 0.23-4.82 | 4.38-9.59 | <0.02-0.48 |
| Hg | PC | 0.10-0.12 | 0.21-0.22 | 0.20-0.28 | 0.10-0.26 |
|  | SF |  | 0.02-0.13 | 0.02-0.05 | 0.05-0.11 |
| Ni | PC | 0.04-0.09 | 0.05-1.08 | 0.03-0.06 | <0.02-0.10 |
|  | SF |  | 0.10-2.67 | 0.29-1.22 | <0.04-0.20 |
| Zn | PC | 4.75-5.62 | 5.70-7.56 | 5.04-5.95 | 7.35-11.50 |
|  | SF |  | 8.89-17.90 | 16.77-26.9 | 6.74-10.30 |


| Liver |  |  | Main Arm |
| :---: | :---: | :---: | :---: |
| Barnston <br> Island | North Arm | FRAP 94/95 |  |
| 0.12 | 0.42 | 0.26 | $0.22-0.55$ |
|  | 0.45 | 0.16 | $0.15-0.30$ |
| 0.13 | 0.10 | 0.13 | $<0.10-0.34$ |
|  | 0.05 | 0.19 | $1.64-2.69$ |
| 0.44 | ND -0.37 | $0.05-0.09$ | $0.10-0.20$ |
|  | 0.07 | 0.21 | $0.44-0.99$ |
| 2.67 | $1.67-6.56$ | $2.38-4.49$ | $4.75-10.50$ |
|  | 9.67 | 4.87 | $14.80-16.20$ |
| 158 | $49.4-120$ | $88-111$ | $153-205$ |
|  | 166 | 165 | $133-164$ |
| 0.07 | $\mathrm{ND}-0.08$ | ND | $0.10-0.13$ |
|  | ND | ND | 0.32 |
| 1.28 | $0.74-3.78$ | $0.70-0.85$ | $1.30-1.50$ |
|  | 0.60 | 0.76 | $1.64-2.00$ |
| 0.24 | $0.07-0.17$ | $0.08-0.10$ | $0.20-0.25$ |
|  | 0.18 | 0.09 | $<0.04-0.08$ |
| 0.12 | $\mathrm{ND}-0.24$ | $\mathrm{ND}-0.06$ | $0.05-0.10$ |
|  | 0.10 | 0.11 | $<0.10-0.70$ |
| 18.10 | $12-14.5$ | $11.3-14.6$ | $17.6-19.2$ |
|  | 31 | 23.7 | $38.2-39.9$ |

Table 64. Metal concentrations measured in mountain whitefish muscle and whitefish from the Fraser River, 1994 and 1995 and as reported in whitefish collected from uncontaminated BC lakes in 1982-1987 (Rieberger 1992). Concentrations in $\mu \mathrm{g} / \mathrm{g}$ wet wt.

|  | Muscle |  |  |  | Liver |  |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Frap 1994 <br> range | Frap 1995 <br> range | BC lakes <br> average | FRAP 1994 <br> range | FRAP 1995 <br> range | BC lakes <br> average |
| $\mathbf{C d}$ | $<0.01$ | $<0.01$ | $\mathbf{0 . 2 4}$ | $0.07-0.33$ | $0.04-0.56$ | $\mathbf{0 . 2 5}$ |
| $\mathbf{C u}$ | $0.54-2.14$ | $0.01-0.16$ | $\mathbf{0 . 3 5}$ | $1.89-3.13$ | $1.60-2.92$ | $\mathbf{3 . 6 2}$ |
| $\mathbf{F e}$ | $4.29-10.30$ | $3.36-5.83$ | $\mathbf{7 . 1 7}$ | $41-92$ | $43.6-104.5$ | $\mathbf{9 5 . 7}$ |
| $\mathbf{P b}$ | $<0.05$ | $<0.05$ | $\mathbf{0 . 2 7}$ | $<0.5$ | $0.03-0.12$ | $\mathbf{0 . 3 8}$ |
| $\mathbf{M n}$ | $<0.20-0.46$ | $<0.20-0.25$ | $\mathbf{0 . 3 2}$ | $1.5-2.0$ | $<2.0-2.06$ | $\mathbf{1 . 7 7}$ |
| $\mathbf{H g}$ | $0.01-0.13$ | $0.03-0.16$ | $\mathbf{0 . 1 1}$ | $0.03-0.29$ | $<0.04-0.13$ | $\mathbf{0 . 1 2}$ |
| $\mathbf{N i}$ | $0.05-0.94(1.63)$ | $<0.02-0.14$ | $\mathbf{1 . 2 1}$ | $<0.02-0.06$ | $<0.10$ | $\mathbf{1 . 2 4}$ |
| $\mathbf{Z n}$ | $4.09-8.69$ | $4.30-8.71$ | $\mathbf{4 . 1 6}$ | $22.6-27.1$ | $21.8-27.0$ | $\mathbf{2 3 . 3}$ |

Table 65. Relative concentrations of trace metals in muscle as a percentage of measured concentrations in liver. Values were calculated as average muscle concentration/liver concentration*100, where concentrations in both tissues were above detection. For the table below, equal concentrations will be $100 \%$. Shown are the means over both years with the range of calculated values.

| Arsenic Chromium | Peamouth |  | Whitefish |  | Flounder |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Mean (Min/Max) | n | Mean (Min/Max) | n | Mean (Min/Max) | n |
|  | 42.6 (13-75) | 7 |  |  | 67.3 (1-133) | 2 |
|  | 48.3 (27-80) | 4 |  |  | 11.0 (8-14) | 2 |
| Copper | 22.5 (4-47) | 15 | 38.6 (12-65) | 12 | 1.7 (2-2) | 4 |
| Iron | 3.4 (2-6) | 15 | 9.3 (4-18) | 12 | 2.2 (2-3) | 4 |
| Manganese | 19.8 (9-26) | 13 | 13.8 (7-23) | 9 | 15.1 (14-16) | 4 |
| Mercury | 180.0 (47-600) | 13 | 70.6 (14-143) | 10 | 100 | 1 |
| Nickel | 366.5 (100-940) | 8 | 1,265.0 (400-2,400) | 6 | 20.0 (10-30) | 2 |
| Selenium | 29.5 (17-47) | 11 | 17.8 (13-24) | 4 | 15.0 (15-15) | 2 |
| Zinc | 49.2(41-60) | 15 | 24.4 (20-30) | 12 | 22.1 (19-25) | 4 |

Table 66. EROD and ECOD activities (pmoles/mg protein/min) in fish livers 1994 to 1996; sample number $(\mathrm{N})$, means, and standard error (SE).

| Reach | Sex | EROD |  |  | ECOD |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $N$ | Mean | SE | $N$ | Mean | SE |
| Peamouth Chub -- 1994 |  |  |  |  |  |  |  |
| Nechako | F | 10 | 0.71 | 0.33 | 10 | 3.81 | 0.91 |
|  | M | 13 | 2.86 | 0.99 | 13 | 10.19 | 1.82 |
|  | Total | 23 | 1.93 | 0.61 | 23 | 7.42 | 1.27 |
| Hansard | F | 10 | 1.57 | 1.00 | 10 | 2.91 | 1.34 |
|  | M | 9 | 3.54 | 1.76 | 9 | 7.46 | 4.11 |
|  | Total | 19 | 2.50 | 0.98 | 19 | 5.07 | 2.08 |
| Woodpecker | F | 11 | 7.88 | 1.79 | 11 | 11.71 | 2.26 |
|  | M | 9 | 15.38 | 4.88 | 9 | 22.36 | 6.02 |
|  | U | 1 | 29.60 | . | 1 | 50.20 | . |
|  | Total | 21 | 12.13 | 2.52 | 21 | 18.10 | 3.39 |
| Marguerite | F | 10 | 10.57 | 4.09 | 10 | 5.14 | 1.24 |
|  | M | 10 | 11.81 | 1.93 | 10 | 9.08 | 1.38 |
|  | U | 1 | 19.00 |  | 1 | 11.50 |  |
|  | Total | 21 | 11.56 | 2.13 | 21 | 7.32 | 0.99 |
| Agassiz | F | 9 | 1.33 | 0.51 | 9 | 1.86 | 0.44 |
|  | M | 9 | 5.69 | 1.30 | 9 | 3.24 | 0.62 |
|  | Total | 18 | 3.51 | 0.86 | 18 | 2.55 | 0.41 |
| Mission | F | 3 | 15.67 | 4.79 | 3 | 14.47 | 6.97 |
|  | M | 3 | 22.20 | 10.70 | 3 | 21.77 | 10.60 |
|  | Total | 6 | 18.94 | 5.44 | 6 | 18.12 | 5.90 |
| Barnston | F | 8 | 19.22 | 6.36 | 8 | 18.45 | 5.92 |
|  | M | 7 | 19.55 | 4.03 | 7 | 19.87 | 2.17 |
|  | Total | 15 | 19.37 | 3.75 | 15 | 19.11 | 3.22 |
| MainArm | F | 6 | 38.12 | 14.76 | 6 | 34.12 | 11.21 |
|  | M | 6 | 30.92 | 5.60 | 6 | 28.12 | 4.48 |
|  | Total | 12 | 34.52 | 7.60 | 12 | 31.12 | 5.82 |
| N.Thompson | F | 9 | 1.75 | 0.59 | 9 | 2.66 | 0.79 |
|  | M | 8 | 4.33 | 2.34 | 8 | 4.28 | 1.13 |
|  | U | 1 | 6.13 |  | 1 | 6.55 | . |
|  | Total | 18 | 3.14 | 1.10 | 18 | 3.60 | 0.67 |
| Thompson | F | 9 | 5.27 | 0.70 | 9 | 8.75 | 1.29 |
|  | M | 11 | 15.42 | 5.77 | 11 | 16.73 | 3.90 |
|  | U | 2 | 9.70 | 9.70 | 2 | 14.64 | 14.27 |
|  | Total | 22 | 10.75 | 3.08 | 22 | 13.28 | 2.34 |

Table 66. EROD and ECOD activities (pmoles/mg protein/min) in fish livers 1994 to 1996; sample number $(\mathrm{N})$, means, and standard error (SE). (continued)

| Reach | Sex | EROD |  |  | ECOD |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $N$ | Mean | SE | $N$ | Mean | SE |
| Peamouth Chub -- 1995 |  |  |  |  |  |  |  |
| Nechako | F | 13 | 3.52 | 0.37 | 0 | . |  |
|  | M | 5 | 3.91 | 0.75 | 0 | . |  |
|  | U | 1 | 2.62 |  | 0 | . |  |
|  | Total | 19 | 3.57 | 0.32 | 0 | . | . |
| Hansard | F | 10 | 2.85 | 0.70 | 0 |  |  |
|  | M | 7 | 3.75 | 0.56 | 0 | . |  |
|  | Total | 17 | 3.22 | 0.47 | 0 | . | . |
| Woodpecker | F | 7 | 3.80 | 0.72 | 0 |  |  |
|  | M | 5 | 4.88 | 0.36 | 0 | . | . |
|  | Total | 12 | 4.25 | 0.46 | 0 | . | . |
| Marguerite | F | 7 | 4.09 | 0.55 | 0 | . | . |
|  | M | 2 | 5.82 | 2.29 | 0 | . | . |
|  | Total | 9 | 4.47 | 0.62 | 0 | . | . |
| Agassiz | F | 8 | 7.98 | 1.02 | 0 | . |  |
|  | M | 3 | 6.39 | 1.36 | 0 | . | . |
|  | Total | 11 | 7.54 | 0.82 | 0 | . | . |
| MainArm | F | 11 | 9.11 | 1.72 | 0 | . | . |
|  | M | 9 | 10.01 | 1.64 | 0 | . | . |
|  | Total | 20 | 9.52 | 1.17 | 0 | . | . |
| NorthArm | F | 13 | 7.89 | 2.59 | 0 | . | . |
|  | M | 5 | 7.97 | 3.77 | 0 | . | . |
|  | U | 2 | 15.15 | 7.87 | 0 | . | . |
|  | Total | 20 | 8.63 | 2.02 | 0 | . | . |
| N.Thompson | F | 11 | 3.40 | 0.54 | 0 | . |  |
|  | M | 8 | 3.58 | 0.47 | 0 | . |  |
|  | Total | 19 | 3.48 | 0.36 | 0 | . | . |
| Thompson | F | 9 | 5.16 | 1.23 | 0 | . |  |
|  | M | 9 | 8.00 | 1.51 | 0 | . |  |
|  | Total | 18 | 6.58 | 1.00 | 0 | . | . |
| Peamouth Chub -- 1996 |  |  |  |  |  |  |  |
| Hansard | F | 10 | 3.25 | 0.92 | 0 | . | . |
|  | M | 5 | 3.04 | 0.52 | 0 | . | . |
|  | Total | 15 | 3.18 | 0.62 | 0 | . | . |
| Woodpecker | F | 12 | 2.96 | 0.45 | 0 | . | . |
|  | M | 3 | 4.51 | 0.38 | 0 | . | . |
|  | U | 1 | 2.94 |  | 0 | . | . |
|  | Total | 16 | 3.25 | 0.37 | 0 | . | . |
| MainArm | F | 8 | 6.01 | 1.02 | 0 | - | . |
|  | M | 1 | 6.51 | . | 0 | . | . |
|  | U | 8 | 9.94 | 3.13 | 0 | . | - |
|  | Total | 17 | 7.89 | 1.57 | 0 | . | . |
| MainArm Sep | F | 10 | 6.61 | 1.39 | 0 | . | . |
|  | M | 10 | 6.44 | 0.96 | 0 | . | . |
|  | U | 1 | 11.28 | . | 0 | . | . |
|  | Total | 22 | 6.64 | 0.78 | 0 | . | . |

Table 66. EROD and ECOD activities (pmoles/mg protein/min) in fish livers 1994 to 1996; sample number $(\mathrm{N})$, means, and standard error (SE). (continued)

| Reach | Sex | EROD |  |  | ECOD |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $N$ | Mean | SE | $N$ | Mean | SE |
| Mountain Whitefish -- 1994 |  |  |  |  |  |  |  |
| Nechako | F | 10 | 0.00 | 0.00 | 10 | 0.12 | 0.10 |
|  | M | 10 | 3.19 | 1.36 | 10 | 0.32 | 0.14 |
|  | Total | 20 | 1.59 | 0.76 | 20 | 0.22 | 0.08 |
| Hansard | F | 8 | 0.00 | 0.00 | 8 | 0.00 | 0.00 |
|  | M | 9 | 14.57 | 5.84 | 9 | 1.13 | 0.47 |
|  | U | 3 | 13.72 | 5.12 | 3 | 1.95 | 1.29 |
|  | Total | 20 | 8.62 | 3.08 | 20 | 0.80 | 0.31 |
| Woodpecker | F | 10 | 0.45 | 0.45 | 10 | 0.00 | 0.00 |
|  | M | 10 | 250.15 | 52.19 | 10 | 31.17 | 13.80 |
|  | Total | 20 | 125.30 | 38.28 | 20 | 15.58 | 7.61 |
| Marguerite | F | 12 | 71.46 | 44.14 | 12 | 5.59 | 3.41 |
|  | M | 6 | 146.33 | 41.91 | 6 | 10.82 | 3.20 |
|  | Total | 18 | 96.42 | 32.95 | 18 | 7.33 | 2.52 |
| Agassiz | F | 9 | 44.83 | 24.71 | 9 | 3.81 | 2.08 |
|  | M | 6 | 80.56 | 23.72 | 6 | 7.03 | 2.52 |
|  | Total | 15 | 59.12 | 17.65 | 15 | 5.10 | 1.60 |
| N.Thompson | F | 10 | 9.61 | 4.13 | 10 | 0.56 | 0.23 |
|  | M | 9 | 13.14 | 2.90 | 9 | 0.81 | 0.27 |
|  | U | 1 | 29.22 |  | 1 | 2.35 |  |
|  | Total | 20 | 12.18 | 2.57 | 20 | 0.76 | 0.19 |
| Thompson | F | 15 | 17.86 | 5.65 | 15 | 4.11 | 1.53 |
|  | M | 8 | 29.12 | 7.70 | 8 | 3.73 | 0.87 |
|  | Total | 23 | 21.77 | 4.59 | 23 | 3.98 | 1.03 |
| Mountain Whitefish -- 1995 |  |  |  |  |  |  |  |
| Nechako | F | 10 | 2.61 | 0.49 | 0 | . | . |
|  | M | 10 | 3.49 | 0.59 | 0 | . |  |
|  | Total | 20 | 3.05 | 0.39 | 0 | . | . |
| Hansard | F | 10 | 3.35 | 1.28 | 0 | . | . |
|  | M | 9 | 4.90 | 0.69 | 0 | . | . |
|  | Total | 19 | 4.08 | 0.75 | 0 | . | . |
| Woodpecker | F | 5 | 2.48 | 1.64 | 0 |  |  |
|  | M | 5 | 5.36 | 1.36 | 0 | . | . |
|  | Total | 10 | 3.92 | 1.11 | 0 | . |  |
| Marguerite | F | 5 | 11.70 | 4.29 | 0 | . | . |
|  | M | 1 | 11.25 | . | 0 | . | . |
|  | Total | 6 | 11.62 | 3.51 | 0 |  |  |
| Agassiz | F | 10 | 2.72 | 0.93 | 0 | . | . |
|  | M | 10 | 11.90 | 2.21 | 0 | . | . |
|  | Total | 20 | 7.31 | 1.57 | 0 | . | . |
| N.Thompson | F | 10 | 1.34 | 0.71 | 0 | . | . |
|  | M | 9 | 3.72 | 0.47 | 0 | . | . |
|  | Total | 19 | 2.47 | 0.51 | 0 | . | . |
| Thompson | F | 9 | 4.00 | 1.66 | 0 | . | . |
|  | M | 7 | 11.43 | 2.18 | 0 | . | . |
|  | U | 2 | 7.28 | 0.41 | 0 | . | . |
|  | Total | 18 | 7.26 | 1.42 | 0 | . | . |

Table 66. EROD and ECOD activities (pmoles/mg protein/min) in fish livers 1994 to 1996; sample number (N), means, and standard error (SE). (continued)

| Reach | Sex | EROD |  |  | ECOD |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $N$ | Mean | SE | $N$ | Mean | SE |
| Mountain Whitefish -- 1996 |  |  |  |  |  |  |  |
| Hansard | F | 10 | 1.96 | 0.86 | 0 | . |  |
|  | M | 10 | 4.67 | 1.10 | 0 |  | . |
|  | Total | 20 | 3.31 | 0.75 | 0 |  |  |
| Woodpecker | F | 10 | 1.64 | 1.12 | 0 | . |  |
|  | M | 8 | 8.17 | 2.32 | 0 | . |  |
|  | Total | 18 | 4.54 | 1.40 | 0 |  |  |
| Starry Flounder --1994 |  |  |  |  |  |  |  |
| MainArm | U | 11 | 9.88 | 2.72 | 11 | 2.90 | 0.77 |
| Starry Flounder -- 1995 |  |  |  |  |  |  |  |
| MainArm | U | 10 | 13.77 | 2.27 | 0 |  |  |
| NorthArm | U | 10 | 33.86 | 4.67 | 0 |  |  |

Table 67. EROD activity (pmoles/mg protein/min) -- comparison of blind duplicate analyses.

| Reach | Species | No. | Duplicate 1 | Duplicate 2 | Mean | RSD |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1995 |  |  |  |  |  |  |
| Nechako | PC | 8 | 0.75 | 3.77 | 2.26 | 94.63 |
|  |  | 11 | 2.41 | 1.97 | 2.19 | 13.97 |
|  |  | 15 | 5.69 | 3.14 | 4.42 | 40.79 |
| Nechako | MW | 2 | 2.27 | 1.09 | 1.68 | 49.43 |
|  |  | 3 | 1.30 | 7.27 | 4.28 | 98.62 |
| Hansard | PC | 5 | 2.60 | 2.21 | 2.41 | 11.39 |
|  |  | 18 | 1.88 | 1.47 | 1.68 | 17.51 |
|  |  | 20 | 1.13 | 1.13 | 1.13 | 0.06 |
|  |  | 23 | 3.86 | 3.83 | 3.84 | 0.53 |
| Hansard | MW | 13 | 3.51 | 2.98 | 3.24 | 11.48 |
|  |  | 20 | 1.02 | 0.78 | 0.90 | 18.61 |
|  |  | 22 | 1.10 | 0.35 | 0.72 | 73.14 |
| Woodpecker | PC | 4 | 2.19 | 1.27 | 1.73 | 37.65 |
|  |  | 10 | 1.27 | 1.89 | 1.58 | 27.85 |
|  |  | 23 | 3.47 | 3.88 | 3.67 | 7.86 |
|  |  | 38 | 5.73 | 5.85 | 5.79 | 1.48 |
| Woodpecker | MW | 7 | 0.76 | 0.66 | 0.71 | 9.61 |
|  |  | 12 | 0.25 | 0.45 | 0.35 | 40.47 |
|  |  | 14 | 3.20 | 5.32 | 4.26 | 35.14 |
|  |  | 16 | 4.17 | 13.85 | 9.01 | 75.91 |
| Marguerite | PC | 2 | 2.30 | 4.24 | 3.27 | 41.92 |
|  |  | 7 | 1.83 | 2.29 | 2.06 | 15.56 |
|  |  | 26 | 6.77 | 9.46 | 8.12 | 23.43 |
|  |  | 29 | 4.79 | 4.79 | 4.79 | 0.02 |
|  |  | 30 | 8.44 | 4.65 | 6.55 | 40.91 |
| N. Thompson | PC | 12 | 1.79 | 2.03 | 1.91 | 9.20 |
|  |  | 13 | 1.57 | 1.59 | 1.58 | 0.92 |
|  |  | 14 | 4.41 | 5.24 | 4.83 | 12.06 |
|  |  | 23 | 5.99 | 7.80 | 6.90 | 18.61 |
| N. Thompson | MW | 2 | 1.74 | 2.34 | 2.04 | 20.96 |
|  |  | 5 | 0.80 | 0.61 | 0.70 | 20.01 |
|  |  | 10 | 1.65 | 0.64 | 1.15 | 62.45 |
| Thompson | PC | 1 | 3.42 | 3.78 | 3.60 | 7.13 |
|  |  | 5 | 4.26 | 4.05 | 4.15 | 3.44 |
| Thompson | MW | 1 | 0.91 | 1.12 | 1.02 | 14.91 |
|  |  | 16 | 6.15 | 6.51 | 6.33 | 4.05 |
|  |  | 23 | 19.65 | 17.68 | 18.66 | 7.44 |
|  |  | 27 | 2.25 | 2.49 | 2.37 | 7.26 |

Table 67. EROD activity (pmoles/mg protein/min) -- comparison of blind duplicate analyses. (continued)

| Reach | Species | No. | Duplicate 1 | Duplicate 2 | Mean | RSD |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1995 (cont) |  |  |  |  |  |  |
| Agassiz | PC | 3 | 6.62 | 5.76 | 5.68 | 10.77 |
|  |  | 5 | 6.77 | 11.12 | 8.94 | 34.37 |
|  |  | 7 | 11.13 | 1.42 | 6.27 | 109.49 |
| Agassiz | MW | 2 | 2.41 | 1.60 | 2.00 | 28.90 |
|  |  | 3 | 1.68 | 1.03 | 1.35 | 34.15 |
|  |  | 8 | 3.14 | 3.08 | 3.11 | 1.41 |
| Main Arm | SF | 13 | 10.60 | 8.96 | 9.78 | 11.85 |
|  |  | 54 | 10.66 | 11.16 | 10.91 | 3.22 |
|  |  | 58 | 10.75 | 17.66 | 14.21 | 34.39 |
| North Arm | PC | 5 | 5.76 | 4.52 | 5.14 | 17.09 |
|  |  | 19 | 5.58 | 3.17 | 4.37 | 38.91 |
|  |  | 33 | 25.09 | 20.95 | 23.02 | 12.72 |
|  |  | 34 | 0.81 | 1.65 | 1.23 | 48.42 |
|  |  | 35 | 26.91 | 18.60 | 22.75 | 25.84 |
|  |  | 38 | 6.12 | 7.06 | 6.59 | 10.05 |
|  |  | 40 | 5.05 | 9.51 | 7.28 | 43.32 |
| 1996 |  |  |  |  |  |  |
| Hansard | PC | 7 | 2.38 | 5.40 | 3.89 | 54.90 |
|  |  | 16 | 3.19 | 0.93 | 2.06 | 77.58 |
| Hansard | MW | 2 | 0.00 | 0.43 | 0.22 | 141.42 |
|  |  | 17 | 4.31 | 5.48 | 4.90 | 16.90 |
|  |  | 20 | 7.05 | 4.35 | 5.70 | 33.49 |
| Woodpecker | PC | 8 | 1.63 | 1.15 | 1.39 | 24.42 |
|  |  | 20 | 2.77 | 1.54 | 2.16 | 40.36 |
|  |  | 21 | 2.24 | 1.47 | 1.86 | 29.35 |
|  |  | 40 | 4.42 | 3.74 | 4.08 | 11.79 |
| Woodpecker | MW | 12 | 0.03 | 1.52 | 0.78 | 135.95 |
|  |  | 14 | 15.02 | 10.92 | 12.97 | 22.35 |
|  |  | 19 | 4.26 | 1.28 | 2.77 | 76.07 |
|  |  | 21 | 13.52 | 13.62 | 13.57 | 0.52 |
| Main Arm | PC | 15 | 3.00 | 6.50 | 4.75 | 52.10 |
|  |  | 33 | 5.63 | 7.60 | 6.62 | 21.06 |
|  |  | 39 | 9.68 | 5.01 | 7.35 | 44.96 |
|  |  | 78 | 0.65 | 6.56 | 3.61 | 115.92 |
|  |  | 81 | 7.13 | 2.75 | 4.94 | 62.69 |
|  |  | 82 | 4.42 | 5.88 | 5.15 | 20.05 |
|  |  | 89 | 2.15 | 4.83 | 3.49 | 54.30 |
| PC = peamouth chub |  |  | MW = mountain whitefish |  |  |  |
| SF = starry flounder |  |  | RSD $=$ relative | andard deviation |  |  |

Table 68. EROD activity (pmoles/mg protein/min) - comparison of the 1994 and 1995/6 methods using peamouth chub samples collected from the Mission reach in 1994.

| Fish \# | 1994 Method |
| :--- | :---: |
| PC187 | 18.8 |
| PC188 | 13.4 |
| PC190 | 24.9 |
| PC191 | 3.0 |
| PC193 | 40.1 |
| PC194 | 23.5 |
| MEAN | $\mathbf{1 8 . 9 5}$ |
|  | 1995/6 Method |
| PC213 | 5.87 |
| PC214 | 0.99 |
| PC215 | 2.57 |
| PC217 | 2.85 |
| PC218 | 0.55 |
| PC219 | 1.67 |
| MEAN | $\mathbf{2 . 4 2}$ |

Table 69. Pearson correlation coefficients for log EROD activity and GSI for peamouth chub and mountain whitefish 1994 to 1996.

Peamouth Chub

| REACH | YEAR | SEX | $r$ | $p$ | $N$ | SEX | $r$ | $p$ | $N$ |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | 1994 | M | -0.05 | 0.87 | 13 | F | -0.42 | 0.23 | 10 |
| Hansard |  |  | -0.16 | 0.68 | 9 |  | 0.26 | 0.47 | 10 |
| Woodpecker |  |  | -0.42 | 0.26 | 9 |  | 0.00 | 0.99 | 11 |
| Marguerite |  |  | -0.25 | 0.49 | 10 |  | -0.94 | $\mathbf{0 . 0 0}$ | $\mathbf{1 0}$ |
| Agassiz |  |  | -0.15 | 0.69 | 9 |  | 0.12 | 0.76 | 9 |
| Mission |  |  | -0.83 | 0.38 | 3 |  | -0.67 | 0.53 | 3 |
| Barnston |  |  | 0.15 | 0.75 | 7 |  | -0.59 | 0.12 | 8 |
| MainArm |  |  | -0.24 | 0.64 | 6 |  | -0.65 | 0.16 | 6 |
| N.Thompson |  |  | -0.07 | 0.87 | 8 |  | -0.34 | 0.37 | 9 |
| Thompson |  |  | 0.36 | 0.27 | 11 |  | -0.73 | $\mathbf{0 . 0 3}$ | $\mathbf{9}$ |
| Nechako | 1995 | M | 0.17 | 0.79 | 5 | F | -0.41 | 0.16 | 13 |
| Hansard |  |  | -0.49 | 0.27 | 7 |  | -0.44 | 0.21 | 10 |
| Woodpecker |  |  | $\mathbf{0 . 9 6}$ | $\mathbf{0 . 0 4}$ | $\mathbf{5}$ |  | -0.97 | 0.00 | 7 |
| Marguerite |  |  | - | - | 0 |  | -0.02 | 0.96 | 7 |
| Agassiz |  |  | 0.30 | 0.81 | 3 |  | -0.52 | 0.19 | 8 |
| MainArm |  |  | -0.14 | 0.73 | 9 |  | 0.25 | 0.47 | 11 |
| NorthArm |  |  | $-\mathbf{0 . 9 6}$ | $\mathbf{0 . 0 1}$ | $\mathbf{5}$ |  | -0.15 | 0.62 | 13 |
| N.Thompson |  |  | -0.41 | 0.31 | 8 |  | -0.80 | $\mathbf{0 . 0 0}$ | $\mathbf{1 1}$ |
| Thompson |  |  | $-\mathbf{0 . 6 8}$ | $\mathbf{0 . 0 4}$ | $\mathbf{9}$ |  | -0.08 | 0.84 | 9 |
| Hansard | 1996 | M | -0.24 | 0.70 | 5 | F | -0.48 | 0.16 | 10 |
| Woodpecker |  |  | 0.95 | 0.20 | 3 |  | 0.01 | 0.98 | 12 |
| MainArm |  |  | 0.61 | 0.14 | 8 |  | -0.58 | 0.13 | 8 |
| MainArm Sep | 1996 | U | -0.25 | 0.49 | 10 | F | -0.50 | 0.14 | 10 |

Mountain Whitefish

| REACH | YEAR | SEX |  |  | SEX | $r$ | $p$ | $N$ |  |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | 1994 | M | 0.51 | 0.13 | 10 | F | EROD=0 |  | 10 |
| Hansard |  |  | -0.52 | 0.15 | 9 |  | EROD $=0$ | $\mathbf{8}$ |  |
| Woodpecker |  |  | $\mathbf{0 . 7 7}$ | $\mathbf{0 . 0 1}$ | $\mathbf{1 0}$ |  | 0.35 | 0.33 | 10 |
| Marguerite |  |  | -0.27 | 0.61 | 6 |  | $\mathbf{- 0 . 9 1}$ | $\mathbf{0 . 0 0}$ | $\mathbf{1 2}$ |
| Agassiz |  |  | 0.00 | 1.00 | 6 |  | $\mathbf{- 0 . 9 2}$ | $\mathbf{0 . 0 0}$ | $\mathbf{9}$ |
| N.Thompson |  |  | -0.24 | 0.53 | 9 |  | $\mathbf{- 0 . 9 4}$ | $\mathbf{0 . 0 0}$ | $\mathbf{1 0}$ |
| Thompson |  |  | -0.19 | 0.66 | 8 |  | $\mathbf{- 0 . 8 6}$ | $\mathbf{0 . 0 0}$ | $\mathbf{1 5}$ |
| Nechako | 1995 | M | -0.28 | 0.44 | 10 | F | $\mathbf{- 0 . 8 6}$ | $\mathbf{0 . 0 0}$ | $\mathbf{1 0}$ |
| Hansard |  |  | 0.22 | 0.58 | 9 |  | $\mathbf{- 0 . 7 7}$ | $\mathbf{0 . 0 1}$ | $\mathbf{1 0}$ |
| Woodpecker |  |  | 0.18 | 0.77 | 5 |  | $\mathbf{- 0 . 9 1}$ | $\mathbf{0 . 0 3}$ | $\mathbf{5}$ |
| Marguerite |  |  | - | - | 0 |  | 0.24 | 0.70 | 5 |
| Agassiz |  |  | -0.13 | 0.72 | 10 |  | $\mathbf{- 0 . 7 1}$ | $\mathbf{0 . 0 2}$ | $\mathbf{1 0}$ |
| N.Thompson |  |  | 0.47 | 0.20 | 9 |  | -0.30 | 0.40 | 10 |
| Thompson |  |  | 0.14 | 0.77 | 7 |  | $\mathbf{- 0 . 8 0}$ | $\mathbf{0 . 0 1}$ | $\mathbf{9}$ |
| Hansard | 1996 | M | 0.49 | 0.15 | 10 | F | $\mathbf{- 0 . 8 1}$ | $\mathbf{0 . 0 0}$ | $\mathbf{1 0}$ |
| Woodpecker |  |  | -0.55 | $\mathbf{0 . 1 6}$ | $\mathbf{8}$ |  | $\mathbf{- 0 . 8 3}$ | $\mathbf{0 . 0 0}$ | $\mathbf{1 0}$ |

Table 70. Pearson correlation coefficients for log EROD activity with HAI, condition index, and hepatosomatic index for mountain whitefish and peamouth chub males and starry flounder immature fish 1994 to 1996.

| Species | Year | Stat | HAI | Condition Index | HSI |
| :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | $r$ | -0.32 | -0.20 | -0.20 |
|  |  | $p$ | 0.00 | 0.07 | 0.07 |
|  |  | $N$ | 85 | 84 | 85 |
| Peamouth Chub | 1995 | $r$ | 0.07 | -0.19 | 0.13 |
|  |  | $p$ | 0.64 | 0.17 | 0.35 |
|  |  | $N$ | 53 | 53 | 53 |
| Peamouth Chub | 1996 | $r$ | 0.30 | -0.42 | 0.20 |
|  |  | $p$ | 0.20 | 0.08 | 0.42 |
|  |  | $N$ | 19 | 19 | 19 |
| Mountain Whitefish | 1994 | $r$ | -0.45 | -0.04 | 0.17 |
|  |  | $p$ | 0.00 | 0.75 | 0.00 |
|  |  | $N$ | 58 | 57 | 57 |
| Mountain Whitefish | 1995 | $r$ | -0.11 | 0.25 | 0.42 |
|  |  | $p$ | 0.46 | 0.08 | 0.00 |
|  |  | $N$ | 51 | 51 | 51 |
| Mountain Whitefish | 1996 | $r$ | -0.11 | -0.08 | -0.20 |
|  |  | $p$ | 0.65 | 0.75 | 0.43 |
|  |  | $N$ | 18 | 18 | 18 |
| Starry Flounder | 1994 | $r$ | 0.30 | 0.77 | 0.25 |
|  |  | $p$ | 0.37 | 0.01 | 0.46 |
|  |  | $N$ | 11 | 10 | 11 |
| Starry Flounder | 1995 | $r$ | -0.02 | -0.22 | 0.77 |
|  |  | $p$ | 0.92 | 0.34 | 0.00 |
|  |  | $N$ | 20 | 20 | 20 |

Table 71. Pearson correlation coefficients for EROD activity and contaminants for peamouth chub and mountain whitefish 1994 to 1995.

| Species | Year |  | Liver PCBs TEQs | Liver Dioxin <br> TEQs | Liver <br> Total <br> TEQs | Muscle PCBs TEQs | Muscle Dioxin TEQs | Muscle Total TEQs | Liver Total DDTs | Muscle Total DDTs | Liver <br> Toxa- <br> phene | Muscle <br> Toxa- <br> phene | Bed Sediment Total PAHs | Bed Sediment Retene |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | $r$ | 0.66 | 0.42 | 0.70 | 0.47 | -0.20 | 0.37 | 0.64 | 0.27 | 0.70 | 0.03 | 0.72 | No |
|  |  | $p$ | 0.04 | 0.30 | 0.05 | 0.18 | 0.63 | 0.37 | 0.04 | 0.45 | 0.02 | 0.93 | 0.07 | Sample |
|  |  | $N$ | 10 | 8 | 8 | 10 | 8 | 8 | 10 | 10 | 10 | 10 | 7 |  |
| Peamouth Chub | 1995 | $r$ | 0.68 | No | No | 0.77 | No | No | 0.76 | 0.63 | 0.53 | 0.17 | 0.57 | 0.70 |
|  |  | $p$ | 0.04 | Sample | Sample | 0.02 | Sample | Sample | 0.02 | 0.07 | 0.15 | 0.67 | 0.11 | 0.03 |
|  |  | $N$ | 9 |  |  | 9 |  |  | 9 | 9 | 9 | 9 | 9 | 9 |
| Mountain Whitefish | 1994 | $r$ | 0.37 | 0.89 | 0.89 | 0.12 | -0.29 | -0.30 | 0.23 | 0.11 | 0.17 | -0.06 | 0.22 | No |
|  |  | $p$ | 0.42 | 0.04 | 0.05 | 0.79 | 0.64 | 0.62 | 0.62 | 0.82 | 0.71 | 0.91 | 0.64 | Sample |
|  |  | $N$ | 7 | 5 | 5 | 7 | 5 | 5 | 7 | 7 | 7 | 7 | 7 |  |
| Mountain Whitefish | 1995 | $r$ | 0.92 | No | No | 0.50 | No | No | 0.66 | 0.48 | 0.56 | 0.30 | 0.55 | 0.30 |
|  |  | $p$ | 0.00 | Sample | Sample | 0.26 | Sample | Sample | 0.11 | 0.28 | 0.19 | 0.52 | 0.20 | 0.52 |
|  |  | $N$ | 7 |  |  | 7 |  |  | 7 | 7 | 7 | 7 | 7 | 7 |

Table 72. Somatic variables for peamouth chub - sample size ( $n$ ), mean, and standard error (SE).

| $\begin{gathered} \text { む̀ } \\ \text { ® } \end{gathered}$ | $\begin{aligned} & \text { y } \\ & \text { © } \\ & \text { © } \\ & \text { © } \end{aligned}$ | Reach |  | $\underset{~}{8}$ |  |  |  |  |  |  | ले | ら | ¢ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1994 | PC | Nechako | n | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 |
|  |  |  | mean | 7.8 | 239.2 | 168.3 | 148.0 | 2.40 | 5.24 | 5.37 | 3.03 | 1.55 | 1.05 |
|  |  |  | SE | 0.3 | 2.9 | 7.5 | 6.1 | 0.16 | 0.28 | 0.80 | 0.34 | 0.04 | 0.01 |
| 1994 | PC | Hansard | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 9.7 | 221.7 | 124.5 | 108.0 | 2.06 | 4.07 | 4.93 | 3.78 | 1.76 | 0.95 |
|  |  |  | SE | 0.4 | 3.4 | 6.5 | 5.2 | 0.19 | 0.32 | 0.69 | 0.44 | 0.08 | 0.01 |
| 1994 | PC | Woodpecker | n | 58 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 59 |
|  |  |  | mean | 5.9 | 202.6 | 89.7 | 79.9 | 1.22 | 2.72 | 1.59 | 1.62 | 1.47 | 0.92 |
|  |  |  | SE | 0.3 | 2.5 | 4.1 | 3.5 | 0.08 | 0.15 | 0.28 | 0.23 | 0.06 | 0.01 |
| 1994 | PC | Marguerite | n | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 6.7 | 197.8 | 81.6 | 72.8 | 1.28 | 2.34 | 2.09 | 2.07 | 1.70 | 0.91 |
|  |  |  | SE | 0.3 | 2.2 | 3.7 | 3.0 | 0.08 | 0.08 | 0.49 | 0.42 | 0.05 | 0.01 |
| 1994 | PC | Agassiz | n | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 59 | 60 |
|  |  |  | mean | 5.7 | 211.2 | 95.7 | 85.2 | 1.31 | 2.59 | 2.79 | 2.80 | 1.53 | 0.89 |
|  |  |  | SE | 0.3 | 2.3 | 3.5 | 2.9 | 0.07 | 0.11 | 0.43 | 0.36 | 0.05 | 0.01 |
| 1994 | PC | Mission | n | 56 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 |
|  |  |  | mean | 4.5 | 201.2 | 92.7 | 81.8 | 1.19 | 2.61 | 1.13 | 1.25 | 1.43 | 0.98 |
|  |  |  | SE | 0.3 | 2.2 | 3.7 | 3.1 | 0.07 | 0.10 | 0.15 | 0.13 | 0.05 | 0.01 |
| 1994 | PC | Barnston | n | 59 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 |
|  |  |  | mean | 5.8 | 212.5 | 108.5 | 96.0 | 1.69 | 4.28 | 1.49 | 1.41 | 1.71 | 0.98 |
|  |  |  | SE | 0.3 | 2.3 | 4.0 | 3.4 | 0.11 | 0.31 | 0.17 | 0.11 | 0.06 | 0.01 |
| 1994 | PC | MainArm | n | 53 | 62 | 62 | 61 | 62 | 61 | 62 | 61 | 61 | 61 |
|  |  |  | mean | 5.7 | 218.4 | 120.2 | 104.8 | 1.88 | 4.37 | 1.60 | 1.49 | 1.77 | 0.99 |
|  |  |  | SE | 0.4 | 2.3 | 4.2 | 3.6 | 0.10 | 0.23 | 0.13 | 0.09 | 0.07 | 0.01 |
| 1994 | PC | NorthArm | n | 50 | 62 | 62 | 62 | 61 | 62 | 62 | 62 | 61 | 62 |
|  |  |  | mean | 5.2 | 213.4 | 112.7 | 98.1 | 1.87 | 4.50 | 1.36 | 1.31 | 1.90 | 1.00 |
|  |  |  | SE | 0.3 | 2.0 | 3.5 | 3.0 | 0.09 | 0.20 | 0.13 | 0.10 | 0.06 | 0.01 |
| 1994 | PC | N.Thompson | n | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 8.9 | 231.5 | 138.4 | 122.0 | 2.18 | 4.11 | 3.82 | 2.62 | 1.75 | 0.96 |
|  |  |  | SE | 0.4 | 2.6 | 5.0 | 4.2 | 0.11 | 0.18 | 0.60 | 0.37 | 0.06 | 0.01 |
| 1994 | PC | Thompson | n | 60 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 59 |
|  |  |  | mean | 7.8 | 221.2 | 134.9 | 114.5 | 1.67 | 3.78 | 2.71 | 2.06 | 1.47 | 1.02 |
|  |  |  | SE | 0.5 | 2.8 | 5.3 | 4.5 | 0.08 | 0.19 | 0.46 | 0.29 | 0.05 | 0.01 |

Table 72. Somatic variables for peamouth chub - sample size ( $n$ ), mean, and standard error (SE). (continued)

| $\begin{aligned} & \text { ষ̀ } \\ & \text { ঠ̀ } \end{aligned}$ |  | Reach |  | $\stackrel{\otimes}{8}$ |  |  |  |  | $\begin{aligned} & \stackrel{H}{0} \\ & : ⿹ \\ & \vdots \\ & \vdots \\ & 0 \\ & 0 \end{aligned}$ | $\begin{aligned} & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \end{aligned}$ | $\begin{aligned} & \text { ¢े } \end{aligned}$ | ら | ¢ 0 0 0 0 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1995 | PC | Nechako | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 6.5 | 214.9 | 122.0 | 106.9 | 2.01 | 4.85 | 2.21 | 1.43 | 1.76 | 1.00 |
|  |  |  | SE | 0.4 | 3.7 | 8.0 | 6.6 | 0.18 | 0.38 | 0.44 | 0.21 | 0.06 | 0.01 |
| 1995 | PC | Hansard | n | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 |
|  |  |  | mean | 10.7 | 225.3 | 130.0 | 114.8 | 2.48 | 4.52 | 2.40 | 1.90 | 2.10 | 0.98 |
|  |  |  | SE | 0.4 | 2.4 | 4.7 | 3.9 | 0.15 | 0.32 | 0.31 | 0.18 | 0.06 | 0.01 |
| 1995 | PC | Woodpecker | n | 60 | 60 | 60 | 60 | 60 | 60 | 59 | 59 | 60 | 60 |
|  |  |  | mean | 6.8 | 208.9 | 102.8 | 91.3 | 1.67 | 3.88 | 1.66 | 1.40 | 1.76 | 0.97 |
|  |  |  | SE | 0.2 | 2.5 | 4.6 | 3.9 | 0.11 | 0.17 | 0.35 | 0.23 | 0.05 | 0.01 |
| 1995 | PC | Marguerite | n | 30 | 30 | 30 | 30 | 30 | 29 | 30 | 30 | 30 | 30 |
|  |  |  | mean | 6.4 | 197.1 | 86.2 | 76.2 | 1.43 | 3.52 | 1.81 | 1.42 | 1.73 | 0.95 |
|  |  |  | SE | 0.4 | 4.2 | 7.7 | 6.3 | 0.24 | 0.32 | 0.80 | 0.51 | 0.14 | 0.01 |
| 1995 | PC | Agassiz | n | 61 | 61 | 60 | 61 | 61 | 61 | 61 | 61 | 61 | 61 |
|  |  |  | mean | 4.7 | 209.8 | 117.1 | 95.3 | 4.76 | 5.41 | 3.64 | 3.25 | 4.94 | 1.01 |
|  |  |  | SE | 0.2 | 2.2 | 4.2 | 3.2 | 0.21 | 0.22 | 0.53 | 0.37 | 0.11 | 0.01 |
| 1995 | PC | MainArm | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 5.0 | 206.9 | 96.3 | 84.1 | 1.71 | 3.95 | 0.82 | 0.87 | 2.03 | 0.93 |
|  |  |  | SE | 0.3 | 2.5 | 3.7 | 3.1 | 0.08 | 0.20 | 0.11 | 0.09 | 0.06 | 0.01 |
| 1995 | PC | NorthArm | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 5.0 | 213.5 | 105.9 | 92.3 | 2.07 | 3.52 | 1.13 | 1.14 | 2.25 | 0.94 |
|  |  |  | SE | 0.2 | 2.1 | 3.1 | 2.7 | 0.09 | 0.11 | 0.11 | 0.09 | 0.07 | 0.01 |
| 1995 | PC | N.Thompson | n | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 7.3 | 222.3 | 133.5 | 113.8 | 2.27 | 4.44 | 3.40 | 2.15 | 2.00 | 0.98 |
|  |  |  | SE | 0.4 | 3.3 | 6.7 | 5.6 | 0.15 | 0.23 | 0.69 | 0.36 | 0.07 | 0.01 |
| 1995 | PC | Thompson | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 6.3 | 219.1 | 131.4 | 113.5 | 2.95 | 5.23 | 1.92 | 1.37 | 2.68 | 1.05 |
|  |  |  | SE | 0.3 | 2.6 | 5.1 | 4.3 | 0.08 | 0.19 | 0.46 | 0.23 | 0.06 | 0.01 |
| 1996 | PC | Hansard | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 10.3 | 220.5 | 122.8 | 108.1 | 2.38 | 3.21 | 2.64 | 2.18 | 2.13 | 0.98 |
|  |  |  | SE | 0.4 | 2.4 | 4.6 | 3.8 | 0.14 | 0.15 | 0.35 | 0.24 | 0.06 | 0.01 |
| 1996 | PC | Woodpecker | n | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 |
|  |  |  | mean | 7.3 | 210.7 | 99.3 | 88.1 | 1.62 | 3.33 | 1.06 | 1.04 | 1.81 | 0.93 |
|  |  |  | SE | 0.3 | 2.0 | 3.4 | 2.9 | 0.08 | 0.18 | 0.20 | 0.16 | 0.05 | 0.01 |
| 1996 | PC | MainArm | n | 60 | 59 | 60 | 58 | 60 | 60 | 60 | 58 | 58 | 57 |
|  |  |  | mean | 4.9 | 206.5 | 103.3 | 86.9 | 2.75 | 4.02 | 1.00 | 1.06 | 3.09 | 0.97 |
|  |  |  | SE | 0.3 | 2.1 | 3.7 | 3.1 | 0.13 | 0.20 | 0.11 | 0.11 | 0.10 | 0.01 |
| 1996 | PC | MainArm Sep | n | 52 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 |
|  |  |  | mean | 4.2 | 201.1 | 89.2 | 76.8 | 1.69 | 3.32 | 1.38 | 1.62 | 2.19 | 0.93 |
|  |  |  | SE | 0.2 | 1.8 | 2.7 | 2.2 | 0.08 | 0.13 | 0.19 | 0.19 | 0.09 | 0.01 |

Table 73. Somatic variables for mountain whitefish - sample ( $n$ ), mean, and standard error (SE).

| $\begin{aligned} & \text { む̀ } \\ & \hline \end{aligned}$ | $\begin{aligned} & \text { e } \\ & \dot{0} \\ & \text { © } \\ & \dot{\omega} \end{aligned}$ | Reach | 0 <br> 0 <br>  <br>  | $\begin{gathered} 8 \\ \hline \end{gathered}$ | $\begin{aligned} & \text { s } \\ & \text { O } \\ & \text { d } \\ & \text { 튼 } \\ & \text { B E } \end{aligned}$ |  | $\begin{aligned} & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \end{aligned}$ | $$ | $\begin{aligned} & \stackrel{\rightharpoonup}{5} \\ & 0 \\ & 3 \\ & 3 \\ & \vdots \\ & 0 \\ & 0 \end{aligned}$ | $\begin{aligned} & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \end{aligned}$ | ָ̄ | ¢ | $\begin{aligned} & \text { y } \\ & 0.3 \\ & 0.0 \\ & 0 \\ & 0 \end{aligned}$ | 0  <br> $\stackrel{3}{3}$  <br> $\frac{0}{0}$  <br> 0  <br> 0 3 <br>   |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1994 | MW | Nechako | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 5.8 | 269.4 | 215.2 | 182.3 | 1.62 | 4.27 | 18.41 | 9.72 | 0.88 | 0.92 | 84.8 |
|  |  |  | SE | 0.3 | 2.8 | 7.3 | 5.8 | 0.12 | 0.36 | 1.37 | 0.66 | 0.06 | 0.01 | 1.1 |
| 1994 | MW | Hansard | n | 59 | 60 | 60 | 59 | 59 | 54 | 60 | 59 | 58 | 59 | 60 |
|  |  |  | mean | 5.4 | 252.4 | 174.1 | 151.4 | 1.31 | 2.46 | 12.70 | 7.11 | 0.80 | 0.91 | 81.2 |
|  |  |  | SE | 0.4 | 3.7 | 8.9 | 7.1 | 0.14 | 0.18 | 1.58 | 0.72 | 0.06 | 0.01 | 1.0 |
| 1994 | MW | Woodpecker | n | 62 | 63 | 64 | 64 | 64 | 64 | 64 | 64 | 64 | 63 | 63 |
|  |  |  | mean | 5.4 | 278.1 | 253.3 | 215.0 | 2.02 | 3.73 | 27.23 | 11.69 | 0.90 | 0.96 | 88.0 |
|  |  |  | SE | 0.3 | 4.0 | 11.6 | 9.1 | 0.20 | 0.24 | 2.82 | 0.96 | 0.06 | 0.01 | 1.1 |
| 1994 | MW | Marguerite | n | 30 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 |
|  |  |  | mean | 4.0 | 267.5 | 199.8 | 182.0 | 1.59 | 5.26 | 1.48 | 0.69 | 0.88 | 0.91 | 78.6 |
|  |  |  | SE | 0.3 | 5.4 | 14.4 | 13.0 | 0.12 | 0.49 | 0.39 | 0.16 | 0.03 | 0.01 | 1.1 |
| 1994 | MW | Agassiz | n | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 |
|  |  |  | mean | 3.2 | 266.2 | 206.3 | 172.4 | 2.13 | 4.87 | 20.18 | 9.20 | 1.11 | 0.88 | 80.7 |
|  |  |  | SE | 0.4 | 6.4 | 18.8 | 13.3 | 0.35 | 0.58 | 5.06 | 1.99 | 0.10 | 0.01 | 1.3 |
| 1994 | MW | N.Thompson | n | 58 | 60 | 60 | 59 | 60 | 60 | 60 | 59 | 59 | 59 | 60 |
|  |  |  | mean | 4.6 | 281.7 | 266.1 | 222.6 | 1.76 | 3.11 | 34.47 | 14.38 | 0.77 | 0.96 | 89.8 |
|  |  |  | SE | 0.2 | 3.2 | 11.2 | 7.9 | 0.15 | 0.13 | 3.49 | 1.24 | 0.05 | 0.01 | 1.4 |
| 1994 | MW | Thompson | n | 78 | 78 | 78 | 78 | 78 | 78 | 78 | 78 | 78 | 78 | 78 |
|  |  |  | mean | 3.3 | 247.9 | 186.4 | 165.3 | 1.31 | 2.66 | 7.97 | 3.05 | 0.73 | 1.05 | 91.7 |
|  |  |  | SE | 0.2 | 3.5 | 8.9 | 6.8 | 0.13 | 0.17 | 1.94 | 0.69 | 0.04 | 0.01 | 0.7 |
| 1995 | MW | Nechako | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 5.5 | 263.8 | 195.9 | 171.5 | 1.35 | 4.62 | 8.20 | 4.69 | 0.79 | 0.92 | 82.5 |
|  |  |  | SE | 0.4 | 3.1 | 6.9 | 5.8 | 0.07 | 0.36 | 0.55 | 0.27 | 0.03 | 0.01 | 0.9 |
| 1995 | MW | Hansard | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 8.9 | 257.2 | 189.7 | 166.7 | 1.73 | 3.50 | 11.32 | 6.37 | 1.00 | 0.95 | 85.4 |
|  |  |  | SE | 0.3 | 3.0 | 8.0 | 6.7 | 0.14 | 0.16 | 1.04 | 0.42 | 0.06 | 0.01 | 1.1 |
| 1995 | MW | Woodpecker | n | 60 | 60 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 59 |
|  |  |  | mean | 4.0 | 263.2 | 215.5 | 187.4 | 1.34 | 3.14 | 18.52 | 9.38 | 0.68 | 1.01 | 91.7 |
|  |  |  | SE | 0.3 | 3.0 | 7.8 | 6.1 | 0.12 | 0.19 | 1.40 | 0.53 | 0.05 | 0.01 | 1.2 |
| 1995 | MW | Marguerite | n | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 |
|  |  |  | mean | 4.2 | 273.1 | 233.1 | 205.0 | 1.96 | 5.77 | 11.61 | 3.69 | 0.79 | 0.94 | 82.5 |
|  |  |  | SE | 0.7 | 10.5 | 35.8 | 27.3 | 0.69 | 1.52 | 5.32 | 1.39 | 0.13 | 0.01 | 1.9 |
| 1995 | MW | Agassiz | n | 45 | 45 | 45 | 45 | 45 | 45 | 45 | 45 | 45 | 45 | 45 |
|  |  |  | mean | 4.6 | 286.7 | 319.2 | 255.2 | 5.56 | 10.59 | 32.84 | 12.10 | 2.11 | 1.03 | 101.2 |
|  |  |  | SE | 0.3 | 5.1 | 18.1 | 13.9 | 0.45 | 0.68 | 3.17 | 0.94 | 0.10 | 0.01 | 1.3 |
| 1995 | MW | N.Thompson | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 6.0 | 271.3 | 233.9 | 197.5 | 2.12 | 3.95 | 22.32 | 11.08 | 1.06 | 0.96 | 89.5 |
|  |  |  | SE | 0.3 | 2.9 | 9.0 | 7.5 | 0.16 | 0.27 | 1.35 | 0.50 | 0.06 | 0.01 | 1.0 |
| 1995 | MW | Thompson | n | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  |  | mean | 3.2 | 271.0 | 228.9 | 199.9 | 1.87 | 3.49 | 14.48 | 6.16 | 0.89 | 0.99 | 87.9 |
|  |  |  | SE | 0.3 | 3.2 | 8.9 | 6.8 | 0.15 | 0.18 | 1.88 | 0.73 | 0.05 | 0.01 | 0.9 |

Table 73. Somatic variables for mountain whitefish - sample (n), mean, and standard error (SE). (continued)

|  |  | Reach | 0 0 0 0 0 | $\begin{gathered} \mathbb{D} \\ \hline \end{gathered}$ |  |  | $\begin{aligned} & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 0 \end{aligned}$ |  | $$ | $\begin{aligned} & \text { B } \\ & \text { D } \\ & \text { D } \\ & 0 \\ & 0 \\ & 0 \\ & 0 \end{aligned}$ | C్ర | $\overline{\text { M }}$ | $\begin{aligned} & \text { y } \\ & \text { N } \\ & 0 \\ & 0 \end{aligned}$ | 0 $\stackrel{3}{5}$ 0 0 0 0 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1996 | MW | Hansard | n | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 |
|  |  |  | mean | 8.8 | 269.8 | 215.9 | 185.1 | 1.98 | 4.55 | 14.93 | 7.43 | 1.04 | 0.93 | 84.4 |
|  |  |  | SE | 0.4 | 2.9 | 8.5 | 6.5 | 0.19 | 0.24 | 1.61 | 0.65 | 0.08 | 0.01 | 1.2 |
| 1996 | MW | Woodpecker | n | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 |
|  |  |  | mean | 5.0 | 273.8 | 217.4 | 189.9 | 1.78 | 3.93 | 13.91 | 7.48 | 0.95 | 0.90 | 80.7 |
|  |  |  | SE | 0.5 | 5.1 | 12.1 | 10.5 | 0.18 | 0.41 | 1.80 | 0.93 | 0.08 | 0.01 | 1.4 |

Table 74. Somatic variables for starry flounder - sample size ( $n$ ), mean, and standard error (SE).

| $\begin{gathered} \text { む̀ } \\ \text { 入 } \end{gathered}$ | $\begin{aligned} & \infty \\ & \text { ó } \\ & \text { © } \\ & \dot{0} \\ & \hline \end{aligned}$ | Reach |  | $\stackrel{8}{\mathrm{~g}}$ | E 0 $\vdots$ ㄴ ㄴ L |  |  |  |  |  | ¢े | ¢ | \% |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1994 | SF | MainArm | n | 62 | 61 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 61 |
|  |  |  | mean | 2.1 | 176.4 | 58.6 | 52.4 | 0.55 | 2.05 | 0.10 | 0.19 | 1.06 | 0.94 |
|  |  |  | SE | 0.0 | 1.3 | 1.3 | 1.2 | 0.02 | 0.07 | 0.01 | 0.02 | 0.03 | 0.01 |
| 1994 | SF | NorthArm | n | 59 | 62 | 62 | 62 | 62 | 62 | 61 | 61 | 62 | 62 |
|  |  |  | mean | 2.2 | 184.2 | 68.1 | 61.3 | 0.73 | 2.49 | 0.06 | 0.10 | 1.19 | 0.97 |
|  |  |  | SE | 0.1 | 1.3 | 1.7 | 1.5 | 0.03 | 0.10 | 0.01 | 0.01 | 0.04 | 0.01 |
| 1995 | SF | MainArm | n | 65 | 65 | 65 | 64 | 65 | 64 | 65 | 64 | 64 | 64 |
|  |  |  | mean | 2.0 | 180.0 | 67.6 | 61.2 | 0.74 | 2.73 | 0.16 | 0.25 | 1.22 | 1.00 |
|  |  |  | SE | 0.1 | 3.4 | 3.7 | 3.4 | 0.04 | 0.16 | 0.02 | 0.04 | 0.03 | 0.01 |
| 1995 | SF | NorthArm | n | 60 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 59 |
|  |  |  | mean | 1.7 | 178.4 | 63.5 | 57.2 | 0.78 | 3.17 | 0.14 | 0.25 | 1.35 | 0.99 |
|  |  |  | SE | 0.1 | 2.0 | 2.2 | 2.0 | 0.04 | 0.19 | 0.02 | 0.04 | 0.03 | 0.01 |

Table 75. Lipid levels (\%) in peamouth chub, mountain whitefish, and starry flounder tissue composites; sample number ( N ), means, and standard error (SE).

| Reach | Year | Liver |  |  | Muscle |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $N$ | Mean | SE | $N$ | Mean | SE |
| Peamouth Chub | 1994 |  |  |  |  |  |  |
| Nechako |  | 2 | 15.00 | 2.00 | 4 | 1.36 | 0.11 |
| Hansard |  | 2 | 12.50 | 0.50 | 4 | 2.10 | 0.14 |
| Woodpecker |  | 1 | 9.10 | . | 4 | 1.78 | 0.20 |
| Marguerite |  | 1 | 3.75 | . | 4 | 1.68 | 0.11 |
| Agassiz |  | 2 | 7.20 | 0.00 | 4 | 1.30 | 0.14 |
| Mission |  | 1 | 28.00 | . | 5 | 1.68 | 0.22 |
| Barnston |  | 1 | 22.00 | . | 5 | 1.76 | 0.17 |
| MainArm |  | 2 | 21.50 | 0.50 | 5 | 1.96 | 0.17 |
| NorthArm |  | 2 | 24.00 | 0.00 | 5 | 1.72 | 0.16 |
| N.Thompson |  | 2 | 4.95 | 0.45 | 4 | 1.40 | 0.12 |
| Thompson |  | 2 | 22.00 | 2.00 | 4 | 3.03 | 0.22 |
| Nechako | 1995 | 2 | 13.50 | 0.50 | 4 | 1.18 | 0.09 |
| Hansard |  | 2 | 15.00 | 2.00 | 4 | 1.78 | 0.13 |
| Woodpecker |  | 2 | 14.00 | 1.00 | 4 | 1.66 | 0.17 |
| Marguerite |  | 1 | 6.80 | . | 4 | 0.49 | 0.17 |
| Agassiz |  | 3 | 24.33 | 0.67 | 4 | 2.48 | 0.05 |
| MainArm |  | 2 | 23.00 | 4.00 | 4 | 1.30 | 0.18 |
| NorthArm |  | 3 | 26.33 | 1.20 | 4 | 1.55 | 0.17 |
| N. Thompson |  | 2 | 10.18 | 0.83 | 4 | 2.01 | 0.29 |
| Thompson |  | 3 | 19.33 | 0.67 | 4 | 2.16 | 0.13 |
| Hansard | 1996 | 2 | 18.72 | 1.91 | 4 | 1.11 | 0.15 |
| Woodpecker |  | 2 | 8.34 | 0.46 | 4 | 0.64 | 0.10 |
| Main Arm |  | 2 | 23.17 | 2.10 | 4 | 1.25 | 0.06 |
| MainArm Sep |  | 2 | 11.19 | 0.13 | 4 | 1.35 | 0.10 |
| Mountain Whitefish | 1994 |  |  |  |  |  |  |
| Nechako |  | 2 | 2.85 | 0.25 | 4 | 2.61 | 0.37 |
| Hansard |  | 1 | 3.00 | . | 4 | 2.48 | 0.33 |
| Woodpecker |  | 2 | 3.03 | 0.03 | 4 | 2.55 | 0.18 |
| Marguerite |  | 1 | 3.30 | . | 4 | 1.78 | 0.34 |
| Agassiz |  | 1 | 3.20 | . | 4 | 1.43 | 0.06 |
| N.Thompson |  | 2 | 2.40 | 0.20 | 4 | 4.45 | 0.31 |
| Thompson |  | 2 | 2.95 | 0.35 | 4 | 3.58 | 0.19 |
| Nechako | 1995 | 2 | 3.25 | 0.05 | 4 | 2.80 | 0.50 |
| Hansard |  | 2 | 3.00 | 0.10 | 4 | 2.46 | 0.23 |
| Woodpecker |  | 2 | 3.25 | 0.25 | 4 | 2.75 | 0.26 |
| Marguerite |  | 1 | 3.80 | . | 3 | 2.27 | 0.29 |
| Agassiz |  | 2 | 4.35 | 0.25 | 4 | 5.88 | 0.67 |
| N.Thompson |  | 2 | 3.18 | 0.23 | 4 | 3.88 | 0.63 |
| Thompson |  | 2 | 3.15 | 0.25 | 4 | 2.90 | 0.15 |
| Hansard | 1996 | 2 | 2.24 | 0.09 | 4 | 2.19 | 0.10 |
| Woodpecker |  | 1 | 2.87 | . | 4 | 1.49 | 0.25 |
| Starry FLounder | 1994 |  |  |  |  |  |  |
| MainArm |  | 1 | 4.90 | . | 5 | 0.20 | 0.03 |
| NorthArm |  | 1 | 5.20 | - | 5 | 0.34 | 0.14 |
| MainArm | 1995 | 1 | 5.55 | . | 4 | 0.37 | 0.03 |
| NorthArm |  | 1 | 5.30 | . | 4 | 0.36 | 0.09 |

Table 76. Fecundity estimates and egg diameter of peamouth chub collected in 1996 from Hansard, Woodpecker, and the Main Arm; sample number (N), means, and standard error (SE).

|  | Fish | Relative | Egg Diameter |  |  | Length | Weight |  |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Reach | No | Fecundity | Mean | SE | $\boldsymbol{N}$ | Age | (mm) | (g) |

Table 77. Fecundity estimates and egg diameter of mountain whitefish collected in 1996 from Hansard, Woodpecker, and the Main Arm.

| Reach | $\begin{gathered} \text { Fish } \\ \text { No } \end{gathered}$ | Relative Fecundity | Egg Diameter |  |  | Age | $\begin{gathered} \text { Length } \\ (\mathrm{mm}) \end{gathered}$ | Weight <br> (g) |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  | Mean | SE | $N$ |  |  |  |
| Hansard | 1 | 2942 | 2266 | 11.4 | 51 | 4 | 259 | 197.4 |
|  | 2 | 3695 | 2331 | 15.9 | 51 | 9 | 275 | 253.0 |
|  | 3 | 2771 | 2311 | 14.8 | 51 | 9 | 286 | 229.5 |
|  | 12 | 3505 | 2495 | 18.1 | 50 | 9 | 267 | 234.3 |
|  | 14 | 4109 | 2551 | 21.6 | 51 | 14 | 301 | 284.3 |
|  | 15 | 3290 | 2384 | 17.7 | 51 | 6 | 274 | 239.1 |
|  | 16 | 5244 | 2224 | 27.4 | 61 | 11 | 305 | 342.0 |
|  | 17 | 2011 | 2349 | 13.0 | 50 | 8 | 283 | 239.6 |
|  | 21 | 3737 | 2285 | 14.7 | 50 | 9 | 274 | 244.4 |
|  | 22 | 2890 | 2206 | 13.1 | 56 | 9 | 295 | 143.6 |
|  | 28 | 2230 | 2096 | 14.8 | 86 | 8 | 245 | 157.1 |
|  | 29 | 4268 | 2483 | 17.9 | 50 | 13 | 283 | 280.8 |
|  | 30 | 4016 | 2424 | 19.9 | 50 | 11 | 266 | 171.3 |
|  | 31 | 4424 | 2503 | 17.3 | 51 | 9 | 279 | 274.3 |
|  | 35 | 2054 | 2414 | 15.2 | 54 | 6 | 242 | 176.9 |
|  | 40 | 2566 | 2258 | 23.3 | 53 | 12 | 280 | 224.6 |
|  | 41 | 2618 | 2254 | 21.7 | 52 | 10 | 262 | 183.0 |
|  | 43 | 2049 | 2033 | 23.7 | 70 | 10 | 255 | 165.3 |
|  | 44 | 3566 | 2329 | 12.8 | 50 | 11 | 280 | 242.1 |
|  | 46 | 2685 | 2066 | 14.6 | 65 | 10 | 263 | 194.0 |
|  | 48 | 1784 | 2403 | 14.0 | 50 | 11 | 252 | 172.6 |
|  | 49 | 4379 | 2344 | 16.0 | 50 | 10 | 285 | 285.7 |
|  | 50 | 6407 | 2483 | 16.4 | 50 | 12 | 318 | 399.9 |
| Woodpecker | 3 | 3954 | 2441 | 16.9 | 50 | 4 | 275 | 249.3 |
|  | 4 | 2445 | 2607 | 11.4 | 50 | 4 | 262 | 192.5 |
|  | 5 | 2107 | 2592 | 17.6 | 50 | 2 | 255 | 185.3 |
|  | 6 | 2885 | 2543 | 12.2 | 50 | 4 | 258 | 191.0 |
|  | 11 | 3674 | 2482 | 12.0 | 50 | 5 | 272 | 250.2 |
|  | 12 | 3649 | 2512 | 19.5 | 50 | 4 | 266 | 216.0 |
|  | 13 | 2490 | 2565 | 16.2 | 50 | 4 | 257 | 196.7 |
|  | 16 | 2452 | 2422 | 12.9 | 50 | 3 | 251 | 180.8 |
|  | 17 | 2153 | 2507 | 14.5 | 50 | 3 | 265 | 200.0 |
|  | 24 | 1789 | 2475 | 13.5 | 50 | 4 | 247 | 151.2 |
|  | 27 | 2328 | 2478 | 16.7 | 50 | 3 | 251 | 170.0 |
|  | 29 | 1928 | 2553 | 17.5 | 50 | 8 | 276 | 215.2 |
|  | 35 | 3841 | 2645 | 13.3 | 50 | 18 | 338 | 412.7 |
|  | 38 | 2108 | 2599 | 17.7 | 50 | 3 | 254 | 183.8 |

Table 78. Sex ratios for peamouth chub, mountain whitefish, and starry flounder -- 1994-1996.

Peamouth Chub

| Year | Reach | $\boldsymbol{F}$ | $\boldsymbol{M}$ | $\boldsymbol{U}$ | $\boldsymbol{F} / \boldsymbol{M}$ <br> Ratio |
| :---: | :--- | :---: | :---: | :---: | :---: |
| 1994 | Nechako | 25 | 34 | 3 | 0.7 |
| 1995 |  | 34 | 18 | 8 | 1.9 |
| 1994 | Hansard | 28 | 32 | 0 | 0.9 |
| 1995 |  | 25 | 30 | 1 | 0.8 |
| 1996 |  | 27 | 31 | 2 | 0.9 |
| 1994 | Woodpecker | 25 | 29 | 6 | 0.9 |
| 1995 |  | 40 | 18 | 2 | 2.2 |
| 1996 |  | 43 | 10 | 2 | 4.3 |
| 1994 | Marguerite | 23 | 22 | 15 | 1.0 |
| 1995 |  | 20 | 10 | 0 | 2.0 |
| 1994 | Agassiz | 27 | 31 | 2 | 0.9 |
| 1995 |  | 35 | 26 | 0 | 1.3 |
| 1994 | Mission | 26 | 31 | 0 | 0.8 |
| 1994 | Barnston | 36 | 25 | 1 | 1.4 |
| 1994 | MainArm | 46 | 9 | 7 | 5.1 |
| 1995 |  | 27 | 32 | 1 | 0.8 |
| 1996 |  | 31 | 3 | 26 | 10.3 |
| 1996 | MainArm Sep | 29 | 22 | 2 | 1.3 |
| 1994 | NorthArm | 42 | 7 | 13 | 6.0 |
| 1995 |  | 46 | 8 | 6 | 5.8 |
| 1994 | N.Thompson | 27 | 25 | 8 | 1.1 |
| 1995 |  | 35 | 18 | 7 | 1.9 |
| 1994 | Thompson | 22 | 35 | 3 | 0.6 |
| 1995 |  | 31 | 27 | 2 | 1.1 |
| Fema |  |  | 4 |  |  |

Mountain Whitefish

| Year | Reach | $\boldsymbol{F}$ | $\boldsymbol{M}$ | $\boldsymbol{U}$ | $F / \boldsymbol{M}$ <br> Ratio |
| :---: | :--- | :---: | :---: | :---: | :---: |
| 1994 | Nechako | 27 | 32 | 1 | 0.8 |
| 1995 |  | 27 | 33 | 0 | 0.8 |
| 1994 | Hansard | 27 | 27 | 6 | 1.0 |
| 1995 |  | 34 | 26 | 0 | 1.3 |
| 1996 |  | 31 | 21 | 0 | 1.5 |
| 1994 | Woodpecker | 27 | 37 | 0 | 0.7 |
| 1995 |  | 21 | 39 | 0 | 0.5 |
| 1996 |  | 21 | 17 | 0 | 1.2 |
| 1994 | Marguerite | 25 | 6 | 0 | 4.2 |
| 1995 |  | 9 | 4 | 1 | 2.3 |
| 1994 | Agassiz | 19 | 6 | 0 | 3.2 |
| 1995 |  | 31 | 14 | 0 | 2.2 |
| 1994 | N.Thompson | 30 | 28 | 2 | 1.1 |
| 1995 |  | 37 | 23 | 0 | 1.6 |
| 1994 | Thompson | 48 | 28 | 2 | 1.7 |
| 1995 |  | 33 | 25 | 2 | 1.3 |

Starry Flounder

| Year | Reach | $\boldsymbol{F}$ | $\boldsymbol{M}$ | $\boldsymbol{U}$ | F/M <br> Ratio |
| :---: | :--- | :---: | :---: | :---: | :---: |
| 1994 | MainArm | 0 | 1 | 61 |  |
| 1995 |  | 29 | 27 | 9 | 1.1 |
| 1994 | NorthArm | 1 | 0 | 61 |  |
| 1995 |  | 31 | 19 | 10 | 1.6 |

$\mathrm{F}=$ Female $\quad \mathrm{M}=$ Male $\quad \mathrm{U}=$ Immature

Table 79. Incidences of intersex in male peamouth chub and starry flounder collected from the Fraser River estuary in 1995 and 1996.

| Year | Reach | Intersex Incidence <br> in Males |
| :--- | :--- | :---: |
| Peamouth Chub |  |  |
| 1995 | MainArm | 0 of 32 |
| 1996 | MainArm Sep | 0 of 22 |
| 1995 | NorthArm | 0 of 8 |
| Starry Flounder |  |  |
| 1995 | MainArm | 4 of 27 |
| 1995 | NorthArm | 3 of 19 |

Table 80. Gut indices (\% of body weight) for peamouth chub -- sample size ( n ), mean, and standard error (SE).

| Reach | Stat | Gut Index (\%) | Reach | Stat | $\begin{gathered} \hline \text { Gut Index } \\ \text { (\%) } \\ \hline \end{gathered}$ | Reach | Stat | $\begin{gathered} \hline \text { Gut Index } \\ (\%) \\ \hline \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $\begin{aligned} & \hline 1994 \\ & \text { Nechako } \end{aligned}$ | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} 62 \\ 3.54 \\ 0.12 \\ \hline \end{gathered}$ | $\begin{array}{\|l\|} \hline 1995 \\ \text { Nechako } \end{array}$ | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} 60 \\ 4.41 \\ 0.14 \\ \hline \end{gathered}$ | 1996 |  |  |
| Hansard | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 59 \\ 3.67 \\ 0.14 \\ \hline \end{gathered}$ | Hansard | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 56 \\ 3.92 \\ 0.22 \\ \hline \end{gathered}$ | Hansard | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SF} \end{gathered}$ | $\begin{gathered} \hline 60 \\ 2.99 \\ 0.09 \\ \hline \end{gathered}$ |
| Woodpecker | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 60 \\ 3.46 \\ 0.12 \\ \hline \end{gathered}$ | Woodpecker | $\begin{gathered} \mathrm{n} \\ \text { mean } \end{gathered}$ | $\begin{gathered} \hline 60 \\ 4.29 \\ 0.09 \\ \hline \end{gathered}$ | Woodpecker | $\begin{gathered} \mathrm{n} \\ \text { mean } \end{gathered}$ | $\begin{gathered} \hline 55 \\ 3.81 \\ 0.2 \\ \hline \end{gathered}$ |
| Marguerite | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 60 \\ 3.31 \\ 0.08 \\ \hline \end{gathered}$ | Marguerite | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 29 \\ 4.7 \\ 0.24 \\ \hline \end{gathered}$ |  |  |  |
| Agassiz | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \end{gathered}$ | $\begin{gathered} \hline 60 \\ 3.05 \\ 0.09 \end{gathered}$ | Agassiz | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \end{gathered}$ | $\begin{gathered} \hline 61 \\ 5.76 \\ 0.19 \end{gathered}$ |  |  |  |
| Mission | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \end{gathered}$ | $\begin{gathered} \hline 57 \\ 3.26 \\ 0.1 \\ \hline \end{gathered}$ |  |  |  |  |  |  |
| Barnston | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \end{gathered}$ | $\begin{gathered} \hline 62 \\ 4.35 \\ 0.23 \\ \hline \end{gathered}$ |  |  |  |  |  |  |
| MainArm | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 61 \\ 4.14 \\ 0.13 \\ \hline \end{gathered}$ | MainArm | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 59 \\ 4.77 \\ 0.18 \\ \hline \end{gathered}$ | MainArm | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \end{gathered}$ | $\begin{gathered} \hline 58 \\ 4.56 \\ 0.14 \\ \hline \end{gathered}$ |
| NorthArm | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SF} \end{gathered}$ | $\begin{gathered} \hline 62 \\ 4.6 \\ 0.17 \\ \hline \end{gathered}$ | NorthArm | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \end{gathered}$ | $\begin{gathered} \hline 60 \\ 3.94 \\ 0.14 \\ \hline \end{gathered}$ | MainArm Sep | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \end{gathered}$ | $\begin{gathered} \hline 53 \\ 4.34 \\ 0.13 \\ \hline \end{gathered}$ |
| N.Thompson | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 60 \\ 3.4 \\ 0.11 \\ \hline \end{gathered}$ | N.Thompson | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SE} \\ \hline \end{gathered}$ | $\begin{gathered} \hline 60 \\ 4.07 \\ 0.18 \\ \hline \end{gathered}$ |  |  |  |
| Thompson | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SF} \end{gathered}$ SE | $\begin{gathered} \hline 60 \\ 3.31 \\ 0.1 \\ \hline \end{gathered}$ | Thompson | $\begin{gathered} \mathrm{n} \\ \text { mean } \\ \mathrm{SF} \end{gathered}$ | $\begin{gathered} \hline 59 \\ 4.67 \\ 0.12 \\ \hline \end{gathered}$ |  |  |  |

Table 81. Gut indices (\% of body weight) for mountain whitefish - sample size ( $n$ ), mean, and standard error (SE).


Table 82. Gut indices (\% of body weight) for starry flounder - sample size (n), mean, and standard error (SE).

| Reach | Stat | Gut Index <br> (\%) | Reach | Stat | Gut Index <br> (\%) |
| :--- | :---: | :---: | :--- | :---: | :---: |
| $\mathbf{1 9 9 4}$ | n | 62 | MainArm | n | 64 |
| MainArm | mean | 3.95 |  | mean | 4.75 |
|  | SE | 0.11 |  | SE | 0.22 |
| NorthArm | n | 62 | NorthArm | n | 60 |
|  | mean | 4.04 |  | mean | 5.48 |
|  | SE | 0.12 |  | SE | 0.23 |

Table 83. Gut contents of peamouth chub collected from the Fraser River estuary in 1994.

| PEAMOUTH CHUB | \%Occ:urrence |  |  |  | Mean Weicıht/Fish (mg) |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | North <br> Arm | $\begin{aligned} & \text { Main } \\ & \text { Arm } \end{aligned}$ | Barn- <br> ston | Mis- <br> sion | North <br> Arm | Main Arm | Barnston | Mission |
| N | 62 | 62 | 59 | 41 | 62 | 62 | 59 | 41 |
| EMPTY | 30.65 | 20.97 | 45.76 | 65.85 |  |  |  |  |
| NON-FOOD ITEMS |  |  |  |  |  |  |  |  |
| Mud and sand | 48.39 | 62.90 | 16.95 | 7.32 | 148.15 | 253.53 | 60.64 | 12.46 |
| Terrestrial vegetation | 8.06 | 27.42 | 18.64 | 14.63 | 0.48 | 0.27 | 3.98 | 3.88 |
| TERRESTRIAL ITEMS |  |  |  |  |  |  |  |  |
| Insecta | 1.61 | 3.23 | 10.17 | 12.20 | 0.37 | 0.35 | 2.83 | 1.71 |
| Aranea | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.02 | 0.00 | 0.00 |
| UNKNOWN ORIGIN |  |  |  |  |  |  |  |  |
| Arthropoda remains | 12.90 | 9.68 | 5.08 | 0.00 | 0.06 | 0.31 | 0.31 | 0.00 |
| Fish bones | 0.00 | 0.00 | 1.69 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Fish scales | 0.00 | 0.00 | 1.69 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 |
| Fish eggs | 0.00 | 0.00 | 0.00 | 2.44 | 0.00 | 0.00 | 0.00 | 0.07 |
| FRESHWATER BENTHOS |  |  |  |  |  |  |  |  |
| Chironomidae pupae | 0.00 | 4.84 | 1.69 | 4.88 | 0.00 | 0.11 | 0.02 | 0.17 |
| Chironomidae unid | 4.84 | 20.97 | 1.69 | 0.00 | 0.11 | 0.06 | 0.05 | 0.00 |
| Chironomini unid | 1.61 | 9.68 | 1.69 | 4.88 | 0.03 | 0.18 | 0.02 | 0.07 |
| Chironomus sp | 9.68 | 20.97 | 3.39 | 0.00 | 0.16 | 4.76 | 0.07 | 0.00 |
| Cryptochironomus sp | 0.00 | 3.23 | 0.00 | 0.00 | 0.00 | 0.06 | 0.00 | 0.00 |
| Dicrotendipes sp | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.03 | 0.00 | 0.00 |
| Phaenopsectra sp | 0.00 | 3.23 | 0.00 | 0.00 | 0.00 | 0.05 | 0.00 | 0.00 |
| Polypedilum (Pentapedilum) sp | 1.61 | 1.61 | 0.00 | 0.00 | 0.02 | 0.03 | 0.00 | 0.00 |
| Polypedilum (Polypedilum) sp | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.02 | 0.00 | 0.00 |
| Orthocladiinae | 0.00 | 9.68 | 3.39 | 2.44 | 0.00 | 0.52 | 0.05 | 0.02 |
| Paracladius sp? | 0.00 | 19.35 | 0.00 | 0.00 | 0.00 | 0.29 | 0.00 | 0.00 |
| Diamesinae | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Monodiamesa sp | 3.23 | 4.84 | 6.78 | 2.44 | 0.06 | 0.19 | 0.27 | 0.02 |
| Tanypodinae | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.02 | 0.00 | 0.00 |
| Procladius sp | 0.00 | 11.29 | 1.69 | 0.00 | 0.00 | 0.58 | 0.03 | 0.00 |
| Ceratopogonidae pupae | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.03 | 0.00 | 0.00 |
| Palpomyia sp | 0.00 | 4.84 | 0.00 | 0.00 | 0.00 | 0.10 | 0.00 | 0.00 |
| Hydropsychidae | 0.00 | 1.61 | 11.86 | 4.88 | 0.00 | 0.00 | 1.76 | 0.07 |

Table 83. Gut contents of peamouth chub collected from the Fraser River estuary in 1994. (continued)

| PEAMOUTH CHUB | \%Occ:urrence |  |  |  | Mean Weight/Fish (mg) |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | North <br> Arm | Main <br> Arm | Barnston | Mission | North <br> Arm | Main Arm | Barnston | Mission |
| Plecoptera | 0.00 | 0.00 | 0.00 | 2.44 | 0.00 | 0.00 | 0.00 | 0.02 |
| Candona sp | 0.00 | 3.23 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Mollusca | 1.61 | 0.00 | 0.00 | 0.00 | 1.05 | 0.00 | 0.00 | 0.00 |
| Bivalvia | 4.84 | 1.61 | 0.00 | 0.00 | 0.03 | 0.00 | 0.00 | 0.00 |
| Pisidium (compressum) | 6.45 | 8.06 | 3.39 | 0.00 | 1.74 | 1.76 | 1.53 | 0.00 |
| Gastropoda | 0.00 | 0.00 | 1.69 | 0.00 | 0.00 | 0.00 | 32.88 | 0.00 |
| Lymnaea sp | 0.00 | 0.00 | 1.69 | 0.00 | 0.00 | 0.00 | 0.08 | 0.00 |
| Oligochaeta egg | 1.61 | 0.00 | 0.00 | 2.44 | 0.00 | 0.00 | 0.00 | 0.02 |
| Green algae (Oedogonium sp?) | 16.13 | 19.35 | 30.51 | 2.44 | 107.73 | 245.15 | 365.10 | 1.02 |
| MARINE BENTHOS |  |  |  |  |  |  |  |  |
| Polychaeta | 0.00 | 3.23 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Gammaridea | 1.61 | 1.61 | 0.00 | 0.00 | 0.03 | 0.00 | 0.00 | 0.00 |
| Ampeliscidae | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.13 | 0.00 | 0.00 |
| Corophium sp | 3.23 | 3.23 | 0.00 | 0.00 | 0.00 | 0.03 | 0.00 | 0.00 |
| Mysidacea | 1.61 | 0.00 | 0.00 | 0.00 | 0.56 | 0.00 | 0.00 | 0.00 |
| Neomysis mercedis | 1.61 | 0.00 | 0.00 | 0.00 | 0.11 | 0.00 | 0.00 | 0.00 |
| Cirripedia | 1.61 | 0.00 | 3.39 | 0.00 | 0.00 | 0.00 | 3.07 | 0.00 |
| Calanoida | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Caligoida | 0.00 | 0.00 | 5.08 | 0.00 | 0.00 | 0.00 | 3.42 | 0.00 |
| Mollusca | 1.61 | 0.00 | 1.69 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Mya sp? | 1.61 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| Mytilus edulis | 14.52 | 0.00 | 8.47 | 0.00 | 73.52 | 0.00 | 7.95 | 0.00 |
| Tellinidae | 3.23 | 0.00 | 1.69 | 0.00 | 1.42 | 0.00 | 0.24 | 0.00 |
| Algae | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| (Brown?) | 0.00 | 0.00 | 1.69 | 0.00 | 0.00 | 0.00 | 0.95 | 0.00 |
| Foliose (red?) | 0.00 | 1.61 | 0.00 | 0.00 | 0.00 | 0.26 | 0.00 | 0.00 |
| Green (Enteromorpha?) | 4.84 | 6.45 | 5.08 | 12.20 | 0.23 | 1.73 | 8.64 | 17.27 |

Table 84. Gut contents of starry flounder collected from the Fraser River estuary in 1994.

| STARRY FLOUNDER <br> ( $n=62$ for all samples) | \%Occurrence <br> North Arm Main Arm |  | ```Mean Weight/Fish (mg)```North Arm Main Arm |  |
| :---: | :---: | :---: | :---: | :---: |
| EMPTY | 20.97 | 14.52 |  |  |
| NON-FOOD ITEMS |  |  |  |  |
| Mud and sand | 69.35 | 83.87 | 268.45 | 183.73 |
| Terrestrial vegetation | 4.84 | 4.84 | 0.29 | 0.10 |
| UNKNOWN ORIGIN |  |  |  |  |
| Arthropoda rem | 4.84 | 1.61 | 0.00 | 0.00 |
| Nematoda | 1.61 | 1.61 | 0.00 | 0.29 |
| FRESHWATER BENTHOS |  |  |  |  |
| Chironomidae pupae | 8.06 | 0.00 | 0.16 | 0.00 |
| Chironomini unid | 22.58 | 0.00 | 0.63 | 0.00 |
| Chironomus sp | 0.00 | 1.61 | 0.00 | 0.03 |
| Orthocladiinae | 1.61 | 0.00 | 0.02 | 0.00 |
| Monodiamesa sp | 29.03 | 1.61 | 1.16 | 0.02 |
| Palpomyia sp | 1.61 | 0.00 | 0.03 | 0.00 |
| Tubificidae | 1.61 | 0.00 | 0.05 | 0.00 |
| Algae Nostoc sp | 0.00 | 1.61 | 0.00 | 0.03 |
| MARINE BENTHOS |  |  |  |  |
| Polychaeta | 50.00 | 80.65 | 0.03 | 0.40 |
| Ampharetidae | 0.00 | 4.84 | 0.00 | 1.82 |
| Amage sp | 25.81 | 6.45 | 89.95 | 1.63 |
| Glyceroidea? | 1.61 | 0.00 | 0.98 | 0.00 |
| Nereidae | 1.61 | 0.00 | 0.02 | 0.00 |
| Gammaridea | 20.97 | 1.61 | 1.47 | 0.00 |
| Corophium sp | 37.10 | 41.94 | 2.89 | 4.06 |
| Ramellogammarus ramellus | 11.29 | 6.45 | 8.19 | 0.21 |
| Grandifoxus grandis | 0.00 | 1.61 | 0.00 | 0.32 |
| Mysidacea | 0.00 | 1.61 | 0.00 | 0.24 |
| Neomysis mercedis | 0.00 | 1.61 | 0.00 | 0.29 |
| Cirripedia | 1.61 | 0.00 | 2.18 | 0.00 |

Table 84. Gut contents of starry flounder collected from the Fraser River estuary in 1994. (continued)

| STARRY FLOUNDER ( $\mathrm{n}=62$ for all samples) | \%Occurrence |  | Mean Wt/Fish (mg) |  |
| :---: | :---: | :---: | :---: | :---: |
|  | North A | Main Arm | North Arm | Main Arm |
| Decapoda frag | 1.61 | 1.61 | 0.00 | 0.00 |
| Crangon alba | 0.00 | 3.23 | 0.00 | 0.63 |
| Mollusca (marine) | 0.00 | 3.23 | 0.00 | 0.00 |
| Bivalvia | 0.00 | 9.68 | 0.00 | 2.71 |
| Macoma balthica | 8.06 | 11.29 | 14.44 | 27.87 |
| Mya sp? | 0.00 | 17.74 | 0.00 | 17.69 |
| Mytilus edulis | 9.68 | 0.00 | 1.68 | 0.00 |
| Tellinidae | 4.84 | 8.06 | 4.11 | 0.53 |
| Tellina sp | 3.23 | 6.45 | 2.26 | 0.00 |
| Green Algae (Enteromorpha?) | 0.00 | 4.84 | 0.00 | 0.68 |

Table 85. Potential causes of abnormalities described in the necropsy-based health assessment.

| Organ/Abnormality | Cause | Reference |
| :---: | :---: | :---: |
| Fin erosion | Toxic chemicals, low dissolved oxygen, high microbial populations | (Sindermann 1979) |
|  | Open wounds exposed to pathogens | (Goede and Barton 1990) |
|  | Chronic exposure to bleached kraft pulp mill effluent, PCBs, zinc, cadmium, and lead | (Lindesjoo and Thulin 1990) |
|  |  | (Lehtinen 1989) |
|  |  | (Couillard et al. 1988) |
|  | Chronic exposure to PCBs, zinc, cadmium, and lead | (Sindermann 1979) |
| Skin lesions | Pollution, bacterial infection, net damage, predator attack, protozoan infection | (Sindermann 1979) |
|  | Any tissue insult or microbial or parasitic lesion | (Roberts 1978) |
| Gill abnormalities | External irritants | (Roberts 1978) |
|  | Damaged gill tissue provides substrate for bacteria and protozoans | (Goede and Barton 1990) |
|  | Clubbing resulting from poor water quality. | (Goede 1993) |
| Bile colour | Short term indicator of feeding activity and nutritional status | (Love 1980) |
|  | Dark green indicative of not feeding for a week or longer; straw yellow indicates feeding within past few hours | (Goede and Barton 1990) |
| Fatty liver | Natural lipid storage gives a light brown or yellow colour | (Roberts 1978) |
|  | Metabolic disorder can cause excessive accumulation of lipids | (Adams et al. 1993) |
| Liver with cysts | Parasites or mycobacteria give hepatomas or cysts | (Goede and Barton 1990) |
| Enlarged spleen | Bacterial infection or disease causes enlargement | (Goede and Barton 1990) |
| Inflamed hindgut | Irritants or toxicants cause inflammation | (Goede and Barton 1990) |
| Swollen or mottled kidney | Bacterial or protozoan colonization causes mottling, swelling, or granulation | (Goede and Barton 1990) <br> (Roberts 1978) |
| Urolithic kidney | High levels of carbon dioxide in water or unsuitable levels of calcium or magesium in diets cause urolithiasis | (Roberts 1978) |

Table 86. Health assessment index and individual component variables for peamouth chub - sample size ( $n$ ), mean, and standard error (SE).

| Year | Reach |  | $\begin{aligned} & \text { E } \\ & \frac{5}{5} \\ & \hline \end{aligned}$ | $\underset{i n}{n}$ | $\stackrel{0}{\omega}$ | $\because$ <br> 0 <br> 0 <br> 0 <br> 0 | $\stackrel{n}{\substack{0}}$ | $\stackrel{\cong}{m}$ | ذ | ~ | $\begin{aligned} & \text { む} \\ & \frac{\otimes}{⿺} \\ & \text { e } \end{aligned}$ |  |  |  | ¢ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1994 | Nechako | n | 62 | 62 | 62 | 62 | 62 | 59 | 62 | 62 | 62 | 62 | 62 | 62 | 62 |
|  |  | mean | 3.4 | 0.8 | 0.0 | 0.0 | 1.9 | 1.8 | 23.7 | 3.3 | 3.4 | 1.9 | 30.0 | 10.8 | 76.0 |
|  |  | SE | 0.9 | 0.3 | 0.0 | 0.0 | 0.9 | 0.1 | 1.6 | 0.1 | 1.2 | 0.6 | 0.0 | 1.3 | 3.2 |
|  | Hansard | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.7 | 0.0 | 0.0 | 0.0 | 6.0 | 1.8 | 12.5 | 2.4 | 1.5 | 8.0 | 30.0 | 1.8 | 60.5 |
|  |  | SE | 0.3 | 0.0 | 0.0 | 0.0 | 1.6 | 0.1 | 1.9 | 0.1 | 0.9 | 0.9 | 0.0 | 0.5 | 3.1 |
|  | Woodpecker | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.7 | 0.0 | 0.0 | 0.0 | 0.0 | 1.0 | 26.5 | 2.6 | 0.0 | 7.7 | 29.5 | 2.7 | 67.0 |
|  |  | SE | 0.3 | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 1.3 | 0.1 | 0.0 | 1.1 | 0.5 | 0.6 | 2.0 |
|  | Marguerite | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 59 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.0 | 0.3 | 0.0 | 0.0 | 1.0 | 0.9 | 21.5 | 1.9 | 0.0 | 0.7 | 30.0 | 3.7 | 57.2 |
|  |  | SE | 0.0 | 0.3 | 0.0 | 0.0 | 0.7 | 0.1 | 1.8 | 0.1 | 0.0 | 0.4 | 0.0 | 0.6 | 2.1 |
|  | Agassiz | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.0 | 0.0 | 0.0 | 0.0 | 3.5 | 1.0 | 12.5 | 1.5 | 1.0 | 5.5 | 30.0 | 5.5 | 58.0 |
|  |  | SE | 0.0 | 0.0 | 0.0 | 0.0 | 1.3 | 0.1 | 1.9 | 0.1 | 0.7 | 0.9 | 0.0 | 0.6 | 2.2 |
|  | Mission | n | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 | 57 |
|  |  | mean | 0.0 | 0.0 | 0.0 | 0.0 | 2.6 | 1.9 | 2.6 | 3.4 | 2.1 | 0.9 | 30.0 | 1.1 | 39.3 |
|  |  | SE | 0.0 | 0.0 | 0.0 | 0.0 | 1.1 | 0.1 | 1.1 | 0.1 | 1.0 | 0.4 | 0.0 | 0.4 | 2.1 |
|  | Barnston | n | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 |
|  |  | mean | 0.5 | 0.5 | 0.0 | 0.0 | 1.0 | 1.2 | 6.3 | 2.6 | 1.9 | 1.5 | 30.0 | 2.9 | 44.5 |
|  |  | SE | 0.3 | 0.4 | 0.0 | 0.0 | 0.7 | 0.1 | 1.6 | 0.2 | 0.9 | 0.5 | 0.0 | 0.6 | 2.0 |
|  | MainArm | n | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 |
|  |  | mean | 0.6 | 0.8 | 0.0 | 0.0 | 1.0 | 1.7 | 13.5 | 3.4 | 1.5 | 5.2 | 30.0 | 1.5 | 54.0 |
|  |  | SE | 0.3 | 0.4 | 0.0 | 0.0 | 0.7 | 0.1 | 1.9 | 0.1 | 0.8 | 0.8 | 0.0 | 0.5 | 2.5 |
|  | NorthArm | n | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 61 | 62 | 62 | 62 | 62 | 62 |
|  |  | mean | 1.1 | 0.5 | 0.0 | 0.0 | 1.0 | 1.9 | 1.0 | 3.4 | 1.0 | 8.7 | 30.0 | 0.2 | 43.4 |
|  |  | SE | 0.5 | 0.3 | 0.0 | 0.0 | 0.7 | 0.1 | 0.7 | 0.1 | 0.7 | 0.8 | 0.0 | 0.2 | 1.4 |
|  | N.Thompson | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.3 | 0.7 | 0.0 | 0.0 | 0.0 | 0.9 | 6.0 | 1.7 | 7.5 | 0.3 | 30.0 | 5.7 | 50.5 |
|  |  | SE | 0.2 | 0.5 | 0.0 | 0.0 | 0.0 | 0.1 | 1.6 | 0.1 | 1.7 | 0.2 | 0.0 | 1.1 | 2.8 |
|  | Thompson | n | 60 | 60 | 60 | 60 | 60 | 58 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.8 | 0.0 | 0.0 | 0.0 | 1.5 | 0.5 | 3.5 | 3.4 | 1.0 | 2.3 | 30.0 | 6.2 | 45.3 |
|  |  | SE | 0.4 | 0.0 | 0.0 | 0.0 | 0.9 | 0.1 | 1.3 | 0.1 | 0.7 | 0.7 | 0.0 | 1.1 | 2.5 |

Table 86. Health assessment index and individual component variables for peamouth chub - sample size (n), mean, and standard error (SE). (Continued)

| Year | Reach |  | $\begin{aligned} & \text { E } \\ & \frac{5}{5} \\ & \hline \end{aligned}$ | $\underset{i n}{n}$ | $\stackrel{0}{\omega}$ | $\because$ 0 0 0 0 | $\stackrel{n}{0}$ | $\stackrel{\cong}{m}$ | ذ | ~ | $\begin{aligned} & \text { む} \\ & \frac{\otimes}{⿺} \\ & \text { e } \end{aligned}$ |  |  | \% | ¢ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1995 | Nechako | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.8 | 0.3 | 0.0 | 0.3 | 1.0 | 1.5 | 6.0 | 2.7 | 0.0 | 1.7 | 30.0 | 1.5 | 41.7 |
|  |  | SE | 0.4 | 0.3 | 0.0 | 0.3 | 0.7 | 0.1 | 1.6 | 0.1 | 0.0 | 0.6 | 0.0 | 0.5 | 2.4 |
|  | Hansard | n | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 | 56 |
|  |  | mean | 0.2 | 0.0 | 0.0 | 0.0 | 2.7 | 1.1 | 3.2 | 2.7 | 1.6 | 0.7 | 30.0 | 0.7 | 39.1 |
|  |  | SE | 0.2 | 0.0 | 0.0 | 0.0 | 1.2 | 0.1 | 1.3 | 0.1 | 0.9 | 0.3 | 0.0 | 0.3 | 2.2 |
|  | Woodpecker | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.5 | 0.0 | 0.0 | 0.0 | 1.0 | 0.9 | 12.0 | 3.6 | 1.5 | 3.5 | 30.0 | 1.8 | 50.3 |
|  |  | SE | 0.4 | 0.0 | 0.0 | 0.0 | 0.7 | 0.1 | 1.9 | 0.1 | 0.9 | 0.7 | 0.0 | 0.5 | 2.6 |
|  | Marguerite | n | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 30 | 30 |
|  |  | mean | 0.3 | 0.0 | 0.0 | 0.0 | 0.0 | 0.8 | 11.0 | 2.2 | 1.0 | 6.7 | 30.0 | 0.3 | 49.3 |
|  |  | SE | 0.3 | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 2.7 | 0.2 | 1.0 | 1.1 | 0.0 | 0.3 | 2.9 |
|  | Agassiz | n | 61 | 61 | 61 | 61 | 61 | 60 | 61 | 61 | 61 | 61 | 61 | 61 | 61 |
|  |  | mean | 0.2 | 0.0 | 0.0 | 0.0 | 0.5 | 0.2 | 0.0 | 3.9 | 0.5 | 9.8 | 30.0 | 1.0 | 42.0 |
|  |  | SE | 0.2 | 0.0 | 0.0 | 0.0 | 0.5 | 0.1 | 0.0 | 0.1 | 0.5 | 0.5 | 0.0 | 0.4 | 1.0 |
|  | MainArm | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 1.3 | 0.3 | 0.0 | 0.0 | 0.5 | 1.0 | 6.0 | 3.4 | 0.0 | 8.5 | 30.0 | 2.7 | 49.3 |
|  |  | SE | 0.6 | 0.3 | 0.0 | 0.0 | 0.5 | 0.1 | 1.6 | 0.1 | 0.0 | 0.9 | 0.0 | 0.8 | 2.2 |
|  | NorthArm | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.8 | 0.0 | 0.0 | 0.0 | 0.0 | 1.3 | 5.0 | 3.7 | 4.5 | 1.7 | 30.0 | 1.0 | 43.0 |
|  |  | SE | 0.4 | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 1.5 | 0.1 | 1.4 | 0.5 | 0.0 | 0.4 | 2.3 |
|  | N.Thompson | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.6 | 0.0 | 3.0 | 0.0 | 0.2 | 30.0 | 10.0 | 40.2 |
|  |  | SE | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 0.0 | 0.2 | 0.0 | 0.2 | 0.0 | 1.4 | 1.4 |
|  | Thompson | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.0 | 0.2 | 0.0 | 0.0 | 0.0 | 0.5 | 0.0 | 3.5 | 0.0 | 0.5 | 30.0 | 0.8 | 31.5 |
|  |  | SE | 0.0 | 0.2 | 0.0 | 0.0 | 0.0 | 0.1 | 0.0 | 0.1 | 0.0 | 0.3 | 0.0 | 0.4 | 0.5 |

Table 86．Health assessment index and individual component variables for peamouth chub－sample size（n）， mean，and standard error（SE）．（Continued）

| Year | Reach |  | $\begin{aligned} & \sqrt[5]{5} \\ & \end{aligned}$ | $\underset{i n}{n}$ | $\stackrel{0}{山}$ | O O 0 0 | $\stackrel{n}{0}$ | $\stackrel{y}{\omega}$ | $\stackrel{\text { ® }}{ \pm}$ | ＊ | $\begin{aligned} & \text { む} \\ & \frac{\otimes}{⿺} \\ & \text { e } \end{aligned}$ |  |  | \％ | ¢ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1996（1） | Hansard | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.0 | 0.0 | 0.0 | 0.0 | 1.0 | 1.4 | 0.0 | 3.7 | 1.0 | 0.2 | 30.0 | 0.5 | 32.7 |
|  |  | SE | 0.0 | 0.0 | 0.0 | 0.0 | 0.7 | 0.1 | 0.0 | 0.1 | 0.7 | 0.2 | 0.0 | 0.3 | 1.1 |
|  | Woodpecker | n | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 | 55 |
|  |  | mean | 0.0 | 0.5 | 0.0 | 0.0 | 1.1 | 1.1 | 18.5 | 3.3 | 0.0 | 0.2 | 30.0 | 1.5 | 51.8 |
|  |  | SE | 0.0 | 0.5 | 0.0 | 0.0 | 0.8 | 0.1 | 2.0 | 0.1 | 0.0 | 0.2 | 0.0 | 0.5 | 2.0 |
|  | MainArm | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.2 | 0.5 | 0.5 | 0.0 | 3.0 | 0.9 | 0.5 | 3.6 | 0.5 | 1.5 | 30.0 | 0.8 | 37.5 |
|  |  | SE | 0.2 | 0.4 | 0.5 | 0.0 | 1.2 | 0.1 | 0.5 | 0.1 | 0.5 | 0.5 | 0.0 | 0.4 | 1.9 |
|  | MainArm Sep | n | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 | 53 |
|  |  | mean | 0.0 | 0.4 | 0.0 | 0.0 | 0.0 | 0.7 | 4.0 | 3.4 | 0.0 | 4.0 | 30.0 | 1.1 | 39.4 |
|  |  | SE | 0.0 | 0.4 | 0.0 | 0.0 | 0.0 | 0.1 | 1.4 | 0.1 | 0.0 | 0.7 | 0.0 | 0.4 | 1.8 |
| 1996（2） | Hansard | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 1.3 | 0.0 | 0.0 | 0.0 | 1.5 | 1.6 | 0.0 | 3.5 | 0.5 | 0.3 | 30.0 | 0.7 | 34.3 |
|  |  | SE | 0.8 | 0.0 | 0.0 | 0.0 | 0.9 | 0.1 | 0.0 | 0.1 | 0.5 | 0.2 | 0.0 | 0.3 | 1.5 |
|  | Woodpecker | n | 55 | 55 | 55 | 55 | 55 | 53 | 54 | 54 | 55 | 53 | 53 | 54 | 53 |
|  |  | mean | 0.2 | 0.2 | 0.0 | 0.0 | 1.6 | 1.1 | 15.0 | 3.0 | 0.0 | 0.0 | 30.0 | 2.0 | 49.2 |
|  |  | SE | 0.2 | 0.2 | 0.0 | 0.0 | 0.9 | 0.1 | 2.1 | 0.1 | 0.0 | 0.0 | 0.0 | 0.6 | 2.1 |
|  | MainArm | n | 52 | 52 | 52 | 52 | 52 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 52 |
|  |  | mean | 0.4 | 0.8 | 0.6 | 0.2 | 6.3 | 1.0 | 0.5 | 3.4 | 1.5 | 1.2 | 30.0 | 1.2 | 42.7 |
|  |  | SE | 0.3 | 0.4 | 0.6 | 0.2 | 1.7 | 0.1 | 0.5 | 0.1 | 0.9 | 0.5 | 0.0 | 0.4 | 2.4 |
|  | MainArm Sep | n | 53 | 53 | 53 | 53 | 53 | 51 | 53 | 53 | 53 | 53 | 53 | 53 | 53 |
|  |  | mean | 0.0 | 0.8 | 0.0 | 0.2 | 1.1 | 0.6 | 5.1 | 3.6 | 1.1 | 4.7 | 30.0 | 1.5 | 44.5 |
|  |  | SE | 0.0 | 0.5 | 0.0 | 0.2 | 0.8 | 0.1 | 1.6 | 0.1 | 0.8 | 1.0 | 0.0 | 0.5 | 2.4 |

Table 87．Health assessment index and individual component variables for mountain whitefish－sample size （ $n$ ），mean，and standard error（SE）．

| Year | Reach | $\begin{aligned} & \text { U } \\ & \stackrel{y}{W} \\ & \stackrel{y}{*} \\ & \hline \end{aligned}$ | $\begin{aligned} & \text { 咅 } \\ & \hline \end{aligned}$ | $\underset{i n}{n}$ | $\begin{array}{r} 0 \\ \stackrel{\omega}{\omega} \\ \hline \end{array}$ | $\begin{aligned} & 0 \\ & 00 \\ & 0 \\ & 0 \end{aligned}$ | $$ | $⿳ 亠 丷 厂 犬$ | $\begin{aligned} & \pm \\ & \hline \end{aligned}$ | $\stackrel{\widetilde{\sim}}{\underline{\sim}}$ | $\begin{aligned} & \text { ¿ } \\ & \frac{0}{む} \\ & \text { © } \end{aligned}$ | $\begin{aligned} & \text { N } \\ & \text { D } \\ & \text { 表 } \end{aligned}$ | $\begin{aligned} & \stackrel{\rightharpoonup}{\mathbf{0}} \\ & \stackrel{\rightharpoonup}{i} \\ & \stackrel{y}{3} \end{aligned}$ |  | ¢ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1994 | Nechako | n | 60 | 60 | 60 | 60 | 60 | 57 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.5 | 0.5 | 0.0 | 0.0 | 0.5 | 2.4 | 6.0 | 1.5 | 2.0 | 1.8 | 18.0 | 18.0 | 47.3 |
|  |  | SE | 0.3 | 0.4 | 0.0 | 0.0 | 0.5 | 0.1 | 1.6 | 0.1 | 1.0 | 0.6 | 1.9 | 1.2 | 3.6 |
|  | Hansard | n | 59 | 60 | 60 | 60 | 60 | 58 | 60 | 60 | 60 | 60 | 60 | 60 | 59 |
|  |  | mean | 3.6 | 0.3 | 0.0 | 0.0 | 4.0 | 2.5 | 6.5 | 1.7 | 2.0 | 4.3 | 7.0 | 3.3 | 30.5 |
|  |  | SE | 0.7 | 0.2 | 0.0 | 0.0 | 1.3 | 0.1 | 1.6 | 0.1 | 1.0 | 0.8 | 1.7 | 0.7 | 4.0 |
|  | Woodpecker | n | 64 | 64 | 64 | 64 | 64 | 64 | 64 | 64 | 64 | 64 | 64 | 64 | 64 |
|  |  | mean | 0.9 | 0.6 | 0.5 | 0.0 | 0.0 | 2.5 | 5.6 | 1.1 | 1.4 | 5.2 | 1.4 | 0.5 | 16.1 |
|  |  | SE | 0.4 | 0.4 | 0.5 | 0.0 | 0.0 | 0.1 | 1.5 | 0.1 | 0.8 | 0.8 | 0.8 | 0.3 | 2.4 |
|  | Marguerite | n | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 | 31 |
|  |  | mean | 0.0 | 0.6 | 0.0 | 0.0 | 0.0 | 0.9 | 18.4 | 1.5 | 0.0 | 1.3 | 11.6 | 3.5 | 35.5 |
|  |  | SE | 0.0 | 0.6 | 0.0 | 0.0 | 0.0 | 0.2 | 2.7 | 0.2 | 0.0 | 0.8 | 2.7 | 1.0 | 4.5 |
|  | Agassiz | n | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 | 25 |
|  |  | mean | 0.0 | 0.0 | 0.0 | 0.0 | 1.2 | 0.6 | 12.0 | 1.0 | 1.2 | 4.0 | 3.6 | 1.2 | 23.2 |
|  |  | SE | 0.0 | 0.0 | 0.0 | 0.0 | 1.2 | 0.2 | 3.0 | 0.1 | 1.2 | 1.2 | 2.0 | 0.7 | 4.6 |
|  | N．Thompson | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.7 | 0.0 | 0.0 | 0.0 | 1.0 | 2.4 | 4.0 | 2.1 | 0.5 | 2.2 | 0.0 | 0.2 | 8.5 |
|  |  | SE | 0.3 | 0.0 | 0.0 | 0.0 | 0.7 | 0.1 | 1.3 | 0.1 | 0.5 | 0.6 | 0.0 | 0.2 | 1.7 |
|  | Thompson | n | 78 | 78 | 78 | 78 | 78 | 78 | 78 | 78 | 77 | 78 | 78 | 78 | 77 |
|  |  | mean | 0.9 | 0.4 | 0.0 | 0.0 | 1.2 | 1.5 | 4.6 | 3.2 | 2.3 | 0.6 | 7.7 | 3.6 | 21.0 |
|  |  | SE | 0.4 | 0.2 | 0.0 | 0.0 | 0.7 | 0.1 | 1.2 | 0.1 | 0.9 | 0.3 | 1.5 | 0.6 | 2.8 |
| 1995 | Nechako | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 1.5 | 0.8 | 0.0 | 0.0 | 0.5 | 1.2 | 1.5 | 2.2 | 2.5 | 1.3 | 1.0 | 10.0 | 19.2 |
|  |  | SE | 0.5 | 0.4 | 0.0 | 0.0 | 0.5 | 0.2 | 0.9 | 0.2 | 1.1 | 0.4 | 0.7 | 1.1 | 2.6 |
|  | Hansard | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 1.0 | 0.0 | 0.0 | 0.2 | 0.0 | 1.9 | 1.5 | 2.3 | 1.5 | 0.2 | 0.5 | 1.0 | 5.8 |
|  |  | SE | 0.4 | 0.0 | 0.0 | 0.2 | 0.0 | 0.1 | 0.9 | 0.1 | 0.9 | 0.2 | 0.5 | 0.4 | 1.5 |
|  | Woodpecker | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.0 | 0.2 | 0.0 | 0.0 | 1.5 | 2.1 | 2.0 | 1.2 | 0.0 | 3.2 | 0.0 | 0.3 | 7.2 |
|  |  | SE | 0.0 | 0.2 | 0.0 | 0.0 | 0.9 | 0.1 | 1.0 | 0.1 | 0.0 | 0.7 | 0.0 | 0.2 | 1.4 |
|  | Marguerite | n | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 | 14 |
|  |  | mean | 0.0 | 0.0 | 0.0 | 0.0 | 2.1 | 1.4 | 0.0 | 1.1 | 2.1 | 8.6 | 0.0 | 0.0 | 12.9 |
|  |  | SE | 0.0 | 0.0 | 0.0 | 0.0 | 2.1 | 0.3 | 0.0 | 0.2 | 2.1 | 1.8 | 0.0 | 0.0 | 4.7 |
|  | Agassiz | n | 45 | 45 | 45 | 45 | 45 | 45 | 45 | 44 | 45 | 45 | 45 | 45 | 45 |
|  |  | mean | 0.2 | 0.4 | 0.0 | 0.0 | 0.7 | 0.2 | 5.3 | 2.7 | 0.0 | 0.4 | 4.0 | 1.3 | 12.4 |
|  |  | SE | 0.2 | 0.3 | 0.0 | 0.0 | 0.7 | 0.1 | 1.7 | 0.2 | 0.0 | 0.4 | 1.5 | 0.5 | 2.6 |
|  | N．Thompson | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.2 | 0.0 | 0.0 | 0.0 | 0.5 | 1.2 | 0.5 | 2.4 | 0.5 | 0.3 | 0.0 | 2.0 | 4.0 |
|  |  | SE | 0.2 | 0.0 | 0.0 | 0.0 | 0.5 | 0.2 | 0.5 | 0.2 | 0.5 | 0.3 | 0.0 | 0.6 | 1.0 |
|  | Thompson | n | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.2 | 0.0 | 0.0 | 0.0 | 0.0 | 1.2 | 1.0 | 2.9 | 0.5 | 0.3 | 0.5 | 3.0 | 5.5 |
|  |  | SE | 0.2 | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 0.7 | 0.1 | 0.5 | 0.3 | 0.5 | 0.6 | 1.7 |

Table 87．Health assessment index and individual component variables for mountain whitefish－sample size （ n ），mean，and standard error（SE）．（continued）

| Year | Reach |  | $\begin{aligned} & \text { Es } \\ & \frac{5}{c} \end{aligned}$ | $\underset{i n}{n}$ | $\stackrel{0}{山 \quad 山}$ | 0 <br> 0 <br> 0 <br> O <br> 0 | $\stackrel{n}{\vdots}$ | $\stackrel{0}{0}$ | $\pm$ | $\stackrel{\widetilde{\sim}}{\stackrel{1}{2}}$ | $\begin{aligned} & \text { む } \\ & \frac{0}{む} \\ & \text { c } \end{aligned}$ |  | $\begin{gathered} \stackrel{\rightharpoonup}{\mathrm{o}} \\ \stackrel{\rightharpoonup}{0} \\ \stackrel{\rightharpoonup}{\mathrm{o}} \end{gathered}$ |  | ¢ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1996（1） | Hansard | n | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 |
|  |  | mean | 0.0 | 0.2 | 0.0 | 0.0 | 2.9 | 1.4 | 0.6 | 1.3 | 2.9 | 0.2 | 5.8 | 2.3 | 14.8 |
|  |  | SE | 0.0 | 0.2 | 0.0 | 0.0 | 1.2 | 0.1 | 0.6 | 0.1 | 1.2 | 0.2 | 1.7 | 0.6 | 2.5 |
|  | Woodpecker | n | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 38 |
|  |  | mean | 1.3 | 0.3 | 0.0 | 0.0 | 2.4 | 2.1 | 0.8 | 0.8 | 0.8 | 0.5 | 14.2 | 2.9 | 23.2 |
|  |  | SE | 0.9 | 0.3 | 0.0 | 0.0 | 1.3 | 0.1 | 0.8 | 0.1 | 0.8 | 0.4 | 2.5 | 0.7 | 3.7 |
| 1996（2） | Hansard | n | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 | 52 |
|  |  | mean | 0.2 | 0.2 | 0.0 | 0.0 | 8.1 | 1.5 | 1.2 | 1.1 | 5.2 | 1.7 | 5.8 | 2.3 | 24.6 |
|  |  | SE | 0.2 | 0.2 | 0.0 | 0.0 | 1.9 | 0.1 | 0.8 | 0.1 | 1.6 | 0.5 | 1.7 | 0.6 | 3.3 |
|  | Woodpecker | n | 38 | 37 | 38 | 38 | 37 | 38 | 38 | 38 | 38 | 38 | 38 | 38 | 37 |
|  |  | mean | 0.8 | 0.3 | 0.8 | 0.0 | 6.5 | 2.0 | 0.0 | 1.3 | 0.8 | 0.0 | 10.3 | 3.4 | 20.0 |
|  |  | SE | 0.8 | 0.3 | 0.8 | 0.0 | 2.1 | 0.1 | 0.0 | 0.1 | 0.8 | 0.0 | 2.3 | 0.8 | 3.3 |

Table 88．Health assessment index and individual component variables for starry flounder－sample size（n）， mean，and standard error（SE）．

| Year | Reach |  | $\begin{aligned} & \text { に. } \\ & \text { に } \end{aligned}$ | $\stackrel{n}{i n}$ | $\stackrel{0}{山}$ | $\begin{aligned} & 0 \\ & 0 \\ & 0 \\ & 0 \end{aligned}$ | © | $\stackrel{0}{0}$ | $\begin{aligned} & \text { ذ } \\ & \hline \end{aligned}$ | ＊ | ¢ | $\begin{aligned} & \text { N } \\ & \text { O } \\ & \text { N } \end{aligned}$ | $$ |  | ¢ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1994 | MainArm | n | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 |
|  |  | mean | 1.5 | 0.2 | 0.0 | 0.0 | 0.0 | 2.5 | 16.5 | 0.0 | 1.5 | 3.4 | 0.0 | 10.2 | 33.1 |
|  |  | SE | 0.5 | 0.2 | 0.0 | 0.0 | 0.0 | 0.1 | 1.9 | 0.0 | 0.8 | 0.7 | 0.0 | 0.2 | 2.2 |
|  | NorthArm | n | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 | 62 |
|  |  | mean | 1.9 | 0.0 | 0.0 | 0.0 | 0.0 | 2.8 | 16.5 | 0.0 | 1.0 | 1.6 | 1.0 | 9.4 | 31.3 |
|  |  | SE | 0.5 | 0.0 | 0.0 | 0.0 | 0.0 | 0.1 | 1.9 | 0.0 | 0.7 | 0.5 | 0.7 | 0.6 | 2.7 |
| 1995 | MainArm | n | 65 | 65 | 65 | 65 | 65 | 65 | 65 | 65 | 65 | 65 | 65 | 65 | 65 |
|  |  | mean | 0.2 | 0.2 | 0.0 | 0.0 | 0.5 | 2.3 | 11.5 | 0.0 | 0.0 | 2.6 | 0.0 | 11.4 | 26.3 |
|  |  | SE | 0.2 | 0.2 | 0.0 | 0.0 | 0.5 | 0.1 | 1.8 | 0.0 | 0.0 | 0.6 | 0.0 | 1.3 | 2.3 |
|  | NorthArm | n | 60 | 60 | 60 | 60 | 60 | 59 | 60 | 60 | 60 | 60 | 60 | 60 | 60 |
|  |  | mean | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 | 2.4 | 25.5 | 0.0 | 0.5 | 0.8 | 0.5 | 16.7 | 44.5 |
|  |  | SE | 0.0 | 0.0 | 0.0 | 0.0 | 0.5 | 0.1 | 1.4 | 0.0 | 0.5 | 0.4 | 0.5 | 1.2 | 2.0 |

Table 89. Decriptions of between-assesssor variability in the necropsy.

| Assessment | Variations |
| :---: | :---: |
| Fins | Assessor 2 saw slightly more fin erosion in peamouth chub from the Main Arm. |
| Skin | Assessor 2 saw more skin lesions in peamouth chub from all reaches; assessor 1 saw more skin lesions in mountain whitefish from Woodpecker. |
| Eyes | Only one abnormal Main Arm peamouth chub was recorded. |
| Gills | Assessor 2 saw more abnormal gills than assessor 1 in both mountain whitefish ( $p=0.0001$ ) and peamouth chub ( $p=0.0442$ ), especially frayed gills*. Goede (1993) notes that "Mere separation of gill lamellae can be construed to be "frayed" but that condition may have been caused by something as simple as the manner in which the gill was exposed by the investigator." |
| Opercula | Assessor 2 noted slightly shortened opercula for two peamouth chub from the Main Arm; assessor 1 saw no abnormalities. |
| Parasites | Assessor 2 saw very slightly more parasites. |
| Bile | Assessments were slightly variable between assessors in the distribution among codes, but mean assessments were not significantly different. |
| Liver | Assessor 1 called more peamouth chub livers discoloured at Woodpecker -- assessor 2 called reddish-brown livers normal, while assessor 1 called them discoloured. |
| Fat | Assessments were slightly variable between assessors in the distribution among codes, but mean assessments were not significantly different for peamouth chub. For mountain whitefish, the effect of assessor on the mean fat was different between the two reaches - assessor 1 rated Hansard fish as fattier than Woodpecker, but assessor 2 rated Woodpecker fish as fattier ( $p=0.0019$ ). |
| Spleen | Assessor 2 saw more abnormal spleens in Main Arm peamouth chub than assessor 1. The greatest variability between assessors was for Hansard mountain whitefish -- several spleens coded "granular" by assessor 1 were coded "nodular" by assessor 2. Goede (1993) notes that "nodular" spleens contain fistulas or other nodules, often cysts. For this study, spleens with bumps or lumps the same colour as the spleen were called "granular." It appears that assessor 1 reported "granular" spleens as "nodular". Granular is considered normal while nodular is abnormal. |
| Hindgut | Assessor 1 coded slightly more peamouth chub hindguts as inflamed, while assessor 2 coded more mountain whitefish hindguts as inflamed. Again, the interaction between reach and assessor was significant for mountain whitefish samples ( $p=0.0053$ ) -- assessor 1 rated Woodpecker hindguts as more inflamed than Hansard, but assessor 2 rated Woodpecker hindguts as more inflamed. |
| Kidney | Assessor 2 saw more black, grey, or brown kidneys in peamouth chub at Woodpecker, and assessor 1 saw more abnormal mountain whitefish kidneys (urolithic or brown) at Woodpecker. A colour chart would help assess the difference between dark red (normal) and brown (abnormal). |
| HAI | Duplicate assessments of peamouth chub showed the same trends among reaches, with assessor 2's HAI slightly higher, primarily due to higher "externals" assessments. Duplicate assessments of mountain whitefish showed different trends between reaches -- assessor 1 ranked Hansard as healthier than Woodpecker, assessor 2 ranked Woodpecker as healthier ( $p=0.0464$ ). |

Table 90. Health assessment index -- percent occurrence of abnormalities contributing to the index in each reach and year for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF).

| Sp. | Year | Reach | Parasite | Discol <br> Liv/ Kid | Enlarged Spleen | Urolithic Kidney | Pale Kidney | Hindgut | External | Other | HAI |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1994 | Nechako | 26.77 | 14.52 | 1.94 | 0.48 | 22.74 | 1.94 | 6.13 | 1.45 | 76.0 |
|  |  | Hansard | 2.83 | 16.00 | 1.50 | 0.00 | 25.00 | 8.00 | 6.67 | 0.50 | 60.5 |
|  |  | Woodpecker | 5.17 | 32.50 | 0.00 | 0.00 | 21.00 | 7.67 | 0.67 | 0.00 | 67.0 |
|  |  | Marguerite | 4.17 | 28.50 | 0.00 | 0.00 | 22.50 | 0.67 | 1.33 | 0.00 | 57.2 |
|  |  | Agassiz | 5.50 | 14.50 | 1.00 | 0.00 | 28.00 | 5.50 | 3.50 | 0.00 | 58.0 |
|  |  | Mission | 2.63 | 3.68 | 1.05 | 0.00 | 2.63 | 0.88 | 2.63 | 25.79 | 39.3 |
|  |  | Barnston | 6.29 | 6.77 | 1.94 | 0.00 | 13.06 | 1.45 | 1.94 | 13.06 | 44.5 |
|  |  | MainArm | 6.53 | 9.92 | 0.00 | 0.00 | 11.61 | 5.16 | 2.42 | 18.39 | 54.0 |
|  |  | NorthArm | 0.40 | 2.66 | 0.97 | 0.00 | 17.42 | 8.71 | 2.58 | 10.65 | 43.4 |
|  |  | N.Thompson | 5.67 | 6.50 | 7.50 | 0.00 | 29.50 | 0.33 | 1.00 | 0.00 | 50.5 |
|  |  | Thompson | 6.17 | 6.50 | 1.00 | 0.00 | 27.00 | 2.33 | 2.33 | 0.00 | 45.3 |
|  | 1995 | Nechako | 5.00 | 2.00 | 0.00 | 0.00 | 29.50 | 1.67 | 2.50 | 1.00 | 41.7 |
|  |  | Hansard | 1.25 | 3.21 | 0.54 | 0.00 | 29.46 | 0.71 | 2.86 | 1.07 | 39.1 |
|  |  | Woodpecker | 1.83 | 22.50 | 1.50 | 0.00 | 18.50 | 3.50 | 1.50 | 1.00 | 50.3 |
|  |  | Marguerite | 0.33 | 26.00 | 1.00 | 0.00 | 13.00 | 6.67 | 0.33 | 2.00 | 49.3 |
|  |  | Agassiz | 0.98 | 0.00 | 0.00 | 0.00 | 25.57 | 9.84 | 0.66 | 4.92 | 42.0 |
|  |  | MainArm | 3.17 | 5.50 | 0.00 | 0.00 | 30.00 | 8.50 | 2.17 | 0.00 | 49.3 |
|  |  | NorthArm | 1.50 | 4.50 | 4.00 | 0.00 | 30.00 | 1.67 | 0.83 | 0.50 | 43.0 |
|  |  | N.Thompson | 10.00 | 0.50 | 0.00 | 0.00 | 29.50 | 0.17 | 0.00 | 0.00 | 40.2 |
|  |  | Thompson | 0.83 | 2.00 | 0.00 | 0.00 | 28.00 | 0.50 | 0.17 | 0.00 | 31.5 |
|  | 1996(1) | Hansard | 0.50 | 0.00 | 1.00 | 0.00 | 30.00 | 0.17 | 1.00 | 0.00 | 32.7 |
|  |  | Woodpecker | 1.45 | 29.45 | 0.00 | 0.00 | 19.09 | 0.18 | 1.64 | 0.00 | 51.8 |
|  |  | MainArm | 1.33 | 0.00 | 0.50 | 0.00 | 29.50 | 1.50 | 4.17 | 0.50 | 37.5 |
|  |  | MainArm Sep | 2.26 | 2.83 | 0.00 | 0.00 | 30.00 | 3.96 | 0.38 | 0.00 | 39.4 |
|  | 1996(2) | Hansard | 0.67 | 0.00 | 0.50 | 0.00 | 30.00 | 0.33 | 2.83 | 0.00 | 34.3 |
|  |  | Woodpecker | 1.89 | 28.30 | 0.00 | 0.00 | 16.42 | 0.00 | 2.00 | 0.57 | 49.2 |
|  |  | MainArm | 1.17 | 0.00 | 1.50 | 0.00 | 30.00 | 1.17 | 8.27 | 0.50 | 42.7 |
|  |  | MainArm Sep | 2.64 | 3.96 | 0.57 | 0.00 | 30.00 | 4.72 | 2.08 | 0.57 | 44.5 |
| MW | 1994 | Nechako | 28.25 | 9.25 | 1.50 | 4.00 | 0.00 | 1.83 | 1.50 | 1.00 | 47.3 |
|  |  | Hansard | 7.08 | 9.25 | 0.50 | 0.00 | 0.00 | 4.33 | 7.97 | 2.00 | 30.5 |
|  |  | Woodpecker | 1.41 | 6.09 | 0.94 | 0.00 | 0.00 | 5.16 | 2.03 | 0.47 | 16.1 |
|  |  | Marguerite | 7.42 | 23.71 | 0.00 | 2.42 | 0.00 | 1.29 | 0.65 | 0.00 | 35.5 |
|  |  | Agassiz | 2.40 | 14.40 | 1.20 | 0.00 | 0.00 | 4.00 | 1.20 | 0.00 | 23.2 |
|  |  | N.Thompson | 0.17 | 4.00 | 0.50 | 0.00 | 0.00 | 2.17 | 1.67 | 0.00 | 8.5 |
|  |  | Thompson | 7.82 | 6.92 | 1.95 | 0.77 | 0.00 | 0.64 | 2.44 | 0.78 | 21.0 |
|  | 1995 | Nechako | 11.25 | 1.25 | 0.50 | 0.00 | 0.00 | 1.33 | 2.83 | 2.00 | 19.2 |
|  |  | Hansard | 2.50 | 0.50 | 1.50 | 0.00 | 0.00 | 0.17 | 1.17 | 0.00 | 5.8 |
|  |  | Woodpecker | 0.83 | 1.50 | 0.00 | 0.00 | 0.00 | 3.17 | 1.67 | 0.00 | 7.2 |
|  |  | Marguerite | 0.00 | 0.00 | 2.14 | 0.00 | 0.00 | 8.57 | 2.14 | 0.00 | 12.9 |
|  |  | Agassiz | 7.33 | 2.67 | 0.00 | 0.00 | 0.00 | 0.44 | 1.33 | 0.67 | 12.4 |
|  |  | N.Thompson | 2.00 | 0.50 | 0.50 | 0.00 | 0.00 | 0.33 | 0.67 | 0.00 | 4.0 |
|  |  | Thompson | 4.00 | 0.00 | 0.50 | 0.00 | 0.00 | 0.33 | 0.17 | 0.50 | 5.5 |
|  | 1996(1) | Hansard | 2.31 | 0.58 | 1.15 | 5.77 | 0.00 | 0.19 | 3.08 | 1.73 | 14.8 |
|  |  | Woodpecker | 2.89 | 4.74 | 0.79 | 10.26 | 0.00 | 0.53 | 3.95 | 0.00 | 23.2 |
|  | 1996(2) | Hansard | 2.31 | 1.15 | 0.58 | 5.77 | 0.00 | 1.73 | 8.46 | 4.62 | 24.6 |
|  |  | Woodpecker | 3.42 | 1.58 | 0.79 | 8.68 | 0.00 | 0.00 | 6.76 | 0.00 | 20.0 |
| SF | 1994 | MainArm | 22.02 | 4.60 | 1.45 | 0.00 | 0.00 | 3.39 | 1.61 | 0.00 | 33.1 |
|  |  | NorthArm | 16.37 | 10.40 | 0.97 | 0.00 | 0.00 | 1.61 | 1.94 | 0.00 | 31.3 |
|  | 1995 | MainArm | 16.46 | 6.46 | 0.00 | 0.00 | 0.00 | 2.62 | 0.77 | 0.00 | 26.3 |
|  |  | NorthArm | 22.17 | 19.50 | 0.50 | 0.00 | 0.00 | 0.83 | 0.50 | 1.00 | 44.5 |

Table 91. Summary of the results of regressions of HAI on log(age) by species, year, and reach for peamouth chub and mountain whitefish.

| Species | Reach | Year | $r^{2}$ | $p$ |
| :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | Nechako | 1994 | 0.00 | 0.8664 |
|  |  | 1995 | 0.00 | 0.9856 |
|  | Hansard | 1994 | 0.01 | 0.4933 |
|  |  | 1995 | 0.02 | 0.2700 |
|  |  | 1996 | 0.02 | 0.3188 |
|  | Woodpecker | 1994 | 0.01 | 0.4923 |
|  |  | 1995 | 0.02 | 0.3536 |
|  |  | 1996 | 0.00 | 0.7337 |
|  | Marguerite | 1994 | 0.09 | 0.0245 |
|  |  | 1995 | 0.11 | 0.0732 |
|  | Agassiz | 1994 | 0.06 | 0.0680 |
|  |  | 1995 | 0.00 | 0.9781 |
|  | Mission | 1994 | 0.05 | 0.1117 |
|  | Barnston | 1994 | 0.01 | 0.5456 |
|  | MainArm | 1994 | 0.21 | 0.0006 |
|  |  | 1995 | 0.08 | 0.0281 |
|  |  | 1996 | 0.03 | 0.0524 |
|  | NorthArm | 1994 | 0.04 | 0.1843 |
|  |  | 1995 | 0.00 | 0.7732 |
|  | N.Thompson | 1994 | 0.01 | 0.3730 |
|  |  | 1995 | 0.15 | 0.0021 |
|  | Thompson | 1994 | 0.06 | 0.0706 |
|  |  | 1995 | 0.17 | 0.0010 |
| Mountain Whitefish | Nechako | 1994 | 0.01 | 0.4165 |
|  |  | 1995 | 0.08 | 0.0299 |
|  | Hansard | 1994 | 0.21 | 0.0002 |
|  |  | 1995 | 0.01 | 0.4263 |
|  |  | 1996 | 0.06 | 0.0936 |
|  | Woodpecker | 1994 | 0.09 | 0.0158 |
|  |  | 1995 | 0.09 | 0.0188 |
|  |  | 1996 | 0.24 | 0.0018 |
|  | Marguerite | 1994 | 0.15 | 0.0361 |
|  |  | 1995 | 0.08 | 0.3347 |
|  | Agassiz | 1994 | 0.08 | 0.1667 |
|  |  | 1995 | 0.00 | 0.6714 |
|  | N.Thompson | 1994 | 0.03 | 0.2234 |
|  |  | 1995 | 0.01 | 0.5881 |
|  | Thompson | 1994 | 0.01 | 0.2895 |
|  |  | 1995 | 0.04 | 0.1313 |

Table 92. Necropsy fin ratings -- percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | Fins 0 | Fins 1 | Fins 2 | Fins 3 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 91.9 | 8.1 | 0.0 | 0.0 |
|  |  | Hansard | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Marguerite | 60 | 98.3 | 0.0 | 1.7 | 0.0 |
|  |  | Agassiz | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Mission | 57 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Barnston | 62 | 96.8 | 1.6 | 1.6 | 0.0 |
|  |  | MainArm | 62 | 93.5 | 4.8 | 1.6 | 0.0 |
|  |  | NorthArm | 62 | 95.2 | 4.8 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 96.7 | 0.0 | 3.3 | 0.0 |
|  |  | Thompson | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  | 1995 | Nechako | 60 | 98.3 | 0.0 | 1.7 | 0.0 |
|  |  | Hansard | 56 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Marguerite | 30 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Agassiz | 61 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 98.3 | 0.0 | 1.7 | 0.0 |
|  |  | NorthArm | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Thompson | 60 | 98.3 | 1.7 | 0.0 | 0.0 |
|  | 1996(1) | Hansard | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 55 | 98.2 | 0.0 | 0.0 | 1.8 |
|  |  | MainArm | 60 | 96.7 | 1.7 | 1.7 | 0.0 |
|  |  | MainArm Sep | 53 | 98.1 | 0.0 | 1.9 | 0.0 |
|  | 1996(2) | Hansard | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 55 | 98.2 | 1.8 | 0.0 | 0.0 |
|  |  | MainArm | 52 | 92.3 | 7.7 | 0.0 | 0.0 |
|  |  | MainArm Sep | 53 | 94.3 | 3.8 | 1.9 | 0.0 |
| Mountain Whitefish | 1994 | Nechako | 60 | 96.7 | 1.7 | 1.7 | 0.0 |
|  |  | Hansard | 60 | 96.7 | 3.3 | 0.0 | 0.0 |
|  |  | Woodpecker | 64 | 95.3 | 3.1 | 1.6 | 0.0 |
|  |  | Marguerite | 31 | 96.8 | 0.0 | 3.2 | 0.0 |
|  |  | Agassiz | 25 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Thompson | 78 | 96.2 | 3.8 | 0.0 | 0.0 |
|  | 1995 | Nechako | 60 | 91.7 | 8.3 | 0.0 | 0.0 |
|  |  | Hansard | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 98.3 | 1.7 | 0.0 | 0.0 |
|  |  | Marguerite | 14 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Agassiz | 45 | 95.6 | 4.4 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Thompson | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  | 1996(1) | Hansard | 52 | 98.1 | 1.9 | 0.0 | 0.0 |
|  |  | Woodpecker | 38 | 97.4 | 2.6 | 0.0 | 0.0 |
|  | 1996(2) | Hansard | 52 | 98.1 | 1.9 | 0.0 | 0.0 |
|  |  | Woodpecker | 37 | 97.3 | 2.7 | 0.0 | 0.0 |
| Starry Flounder | 1994 | MainArm | 62 | 98.4 | 1.6 | 0.0 | 0.0 |
|  |  | NorthArm | 62 | 100.0 | 0.0 | 0.0 | 0.0 |
|  | 1995 | MainArm | 65 | 98.5 | 1.5 | 0.0 | 0.0 |
|  |  | NorthArm | 60 | 100.0 | 0.0 | 0.0 | 0.0 |

Table 93. Necropsy skin ratings -- percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | Skin 0 | Skin 1 | Skin 2 | Skin 3 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 75.8 | 16.1 | 6.5 | 1.6 |
|  |  | Hansard | 60 | 93.3 | 6.7 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 93.3 | 6.7 | 0.0 | 0.0 |
|  |  | Marguerite | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Agassiz | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Mission | 57 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Barnston | 62 | 95.2 | 4.8 | 0.0 | 0.0 |
|  |  | MainArm | 62 | 93.5 | 6.5 | 0.0 | 0.0 |
|  |  | NorthArm | 62 | 90.3 | 8.1 | 1.6 | 0.0 |
|  |  | N.Thompson | 60 | 96.7 | 3.3 | 0.0 | 0.0 |
|  |  | Thompson | 60 | 91.7 | 8.3 | 0.0 | 0.0 |
|  | 1995 | Nechako | 60 | 91.7 | 8.3 | 0.0 | 0.0 |
|  |  | Hansard | 56 | 98.2 | 1.8 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 96.7 | 1.7 | 1.7 | 0.0 |
|  |  | Marguerite | 30 | 96.7 | 3.3 | 0.0 | 0.0 |
|  |  | Agassiz | 61 | 98.4 | 1.6 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 90.0 | 8.3 | 0.0 | 1.7 |
|  |  | NorthArm | 60 | 91.7 | 8.3 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Thompson | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  | 1996(1) | Hansard | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 55 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 98.3 | 1.7 | 0.0 | 0.0 |
|  |  | MainArm Sep | 53 | 100.0 | 0.0 | 0.0 | 0.0 |
|  | 1996(2) | Hansard | 60 | 95.0 | 0.0 | 1.7 | 3.3 |
|  |  | Woodpecker | 55 | 98.2 | 1.8 | 0.0 | 0.0 |
|  |  | MainArm | 52 | 96.2 | 3.8 | 0.0 | 0.0 |
|  |  | MainArm Sep | 53 | 100.0 | 0.0 | 0.0 | 0.0 |
| Mountain Whitefish | 1994 | Nechako | 60 | 95.0 | 5.0 | 0.0 | 0.0 |
|  |  | Hansard | 59 | 67.8 | 28.8 | 3.4 | 0.0 |
|  |  | Woodpecker | 64 | 90.6 | 9.4 | 0.0 | 0.0 |
|  |  | Marguerite | 31 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Agassiz | 25 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 93.3 | 6.7 | 0.0 | 0.0 |
|  |  | Thompson | 78 | 92.3 | 6.4 | 1.3 | 0.0 |
|  | 1995 | Nechako | 60 | 85.0 | 15.0 | 0.0 | 0.0 |
|  |  | Hansard | 60 | 90.0 | 10.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Marguerite | 14 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Agassiz | 45 | 97.8 | 2.2 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 98.3 | 1.7 | 0.0 | 0.0 |
|  |  | Thompson | 60 | 98.3 | 1.7 | 0.0 | 0.0 |
|  | 1996(1) | Hansard | 52 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 38 | 94.7 | 0.0 | 2.6 | 2.6 |
|  | 1996(2) | Hansard | 52 | 98.1 | 1.9 | 0.0 | 0.0 |
|  |  | Woodpecker | 38 | 97.4 | 0.0 | 0.0 | 2.6 |
| Starry Flounder | 1994 | MainArm | 62 | 85.5 | 14.5 | 0.0 | 0.0 |
|  |  | NorthArm | 62 | 80.6 | 19.4 | 0.0 | 0.0 |
|  | 1995 | MainArm | 65 | 98.5 | 1.5 | 0.0 | 0.0 |
|  |  | NorthArm | 60 | 100.0 | 0.0 | 0.0 | 0.0 |

Table 94. Necropsy gill conditions - percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | Frayed | Clubbed | Marginate | Pale | Other | Normal |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 6.5 | 0.0 | 0.0 | 0.0 | 0.0 | 93.5 |
|  |  | Hansard | 60 | 20.0 | 0.0 | 1.7 | 0.0 | 0.0 | 80.0 |
|  |  | Woodpecker | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Marguerite | 60 | 1.7 | 1.7 | 0.0 | 0.0 | 0.0 | 96.7 |
|  |  | Agassiz | 60 | 8.3 | 0.0 | 1.7 | 1.7 | 3.3 | 88.3 |
|  |  | Mission | 57 | 0.0 | 3.5 | 3.5 | 0.0 | 1.8 | 91.2 |
|  |  | Barnston | 62 | 0.0 | 0.0 | 3.2 | 0.0 | 0.0 | 96.8 |
|  |  | MainArm | 62 | 0.0 | 1.6 | 0.0 | 0.0 | 1.6 | 96.8 |
|  |  | NorthArm | 62 | 0.0 | 1.6 | 3.2 | 0.0 | 0.0 | 96.8 |
|  |  | N.Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Thompson | 60 | 0.0 | 0.0 | 3.3 | 0.0 | 1.7 | 95.0 |
|  | 1995 | Nechako | 60 | 1.7 | 0.0 | 0.0 | 1.7 | 0.0 | 96.7 |
|  |  | Hansard | 56 | 5.4 | 0.0 | 0.0 | 0.0 | 3.6 | 91.1 |
|  |  | Woodpecker | 60 | 3.3 | 0.0 | 0.0 | 0.0 | 0.0 | 96.7 |
|  |  | Marguerite | 30 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Agassiz | 61 | 1.6 | 0.0 | 0.0 | 1.6 | 0.0 | 98.4 |
|  |  | MainArm | 60 | 1.7 | 0.0 | 0.0 | 0.0 | 0.0 | 98.3 |
|  |  | NorthArm | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | N.Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  | 1996(1) | Hansard | 60 | 3.3 | 0.0 | 0.0 | 0.0 | 0.0 | 96.7 |
|  |  | Woodpecker | 55 | 0.0 | 0.0 | 0.0 | 0.0 | 3.6 | 96.4 |
|  |  | MainArm | 60 | 5.0 | 0.0 | 0.0 | 3.3 | 1.7 | 90.0 |
|  |  | MainArm Sep | 53 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  | 1996(2) | Hansard | 60 | 5.0 | 0.0 | 0.0 | 0.0 | 0.0 | 95.0 |
|  |  | Woodpecker | 55 | 0.0 | 0.0 | 0.0 | 0.0 | 5.5 | 94.5 |
|  |  | MainArm | 52 | 11.5 | 0.0 | 1.9 | 7.7 | 0.0 | 78.8 |
|  |  | MainArm Sep | 53 | 3.8 | 0.0 | 0.0 | 0.0 | 0.0 | 96.2 |
| Mountain Whitefish | 1994 | Nechako | 60 | 0.0 | 1.7 | 0.0 | 0.0 | 0.0 | 98.3 |
|  |  | Hansard | 60 | 13.3 | 0.0 | 0.0 | 0.0 | 0.0 | 86.7 |
|  |  | Woodpecker | 64 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Marguerite | 31 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Agassiz | 25 | 4.0 | 0.0 | 0.0 | 0.0 | 0.0 | 96.0 |
|  |  | N.Thompson | 60 | 1.7 | 0.0 | 1.7 | 0.0 | 0.0 | 96.7 |
|  |  | Thompson | 78 | 2.6 | 0.0 | 0.0 | 0.0 | 1.3 | 96.2 |
|  | 1995 | Nechako | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 1.7 | 98.3 |
|  |  | Hansard | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Woodpecker | 60 | 1.7 | 0.0 | 0.0 | 0.0 | 3.3 | 95.0 |
|  |  | Marguerite | 14 | 7.1 | 0.0 | 0.0 | 0.0 | 0.0 | 92.9 |
|  |  | Agassiz | 45 | 2.2 | 0.0 | 0.0 | 0.0 | 0.0 | 97.8 |
|  |  | N.Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 1.7 | 98.3 |
|  |  | Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  | 1996(1) | Hansard | 52 | 1.9 | 1.9 | 0.0 | 0.0 | 5.8 | 90.4 |
|  |  | Woodpecker | 38 | 7.9 | 0.0 | 0.0 | 0.0 | 0.0 | 92.1 |
|  | 1996(2) | Hansard | 52 | 26.9 | 0.0 | 0.0 | 0.0 | 0.0 | 73.1 |
|  |  | Woodpecker | 37 | 16.2 | 0.0 | 0.0 | 0.0 | 5.4 | 78.4 |
| Starry Flounder | 1994 | MainArm | 62 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | NorthArm | 62 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  | 1995 | MainArm | 65 | 0.0 | 0.0 | 0.0 | 1.5 | 0.0 | 98.5 |
|  |  | NorthArm | 60 | 0.0 | 0.0 | 0.0 | 1.7 | 0.0 | 98.3 |

Table 95. Necropsy parasite ratings -- percent occurrences in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | $\begin{gathered} \text { Parasite } \\ 0 \end{gathered}$ | Parasite 1 | Parasite $2$ | $\begin{gathered} \hline \text { Parasite } \\ 3 \end{gathered}$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 37.1 | 30.6 | 19.4 | 12.9 |
|  |  | Hansard | 60 | 81.7 | 18.3 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 73.3 | 26.7 | 0.0 | 0.0 |
|  |  | Marguerite | 60 | 63.3 | 36.7 | 0.0 | 0.0 |
|  |  | Agassiz | 60 | 45.0 | 55.0 | 0.0 | 0.0 |
|  |  | Mission | 57 | 89.5 | 10.5 | 0.0 | 0.0 |
|  |  | Barnston | 62 | 71.0 | 29.0 | 0.0 | 0.0 |
|  |  | MainArm | 62 | 85.5 | 14.5 | 0.0 | 0.0 |
|  |  | NorthArm | 62 | 98.4 | 1.6 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 70.0 | 3.3 | 26.7 | 0.0 |
|  |  | Thompson | 60 | 65.0 | 8.3 | 26.7 | 0.0 |
|  | 1995 | Nechako | 60 | 85.0 | 15.0 | 0.0 | 0.0 |
|  |  | Hansard | 56 | 92.9 | 7.1 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 81.7 | 18.3 | 0.0 | 0.0 |
|  |  | Marguerite | 30 | 96.7 | 3.3 | 0.0 | 0.0 |
|  |  | Agassiz | 61 | 90.2 | 9.8 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 80.0 | 15.0 | 3.3 | 1.7 |
|  |  | NorthArm | 60 | 90.0 | 10.0 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 53.3 | 0.0 | 40.0 | 6.7 |
|  |  | Thompson | 60 | 91.7 | 8.3 | 0.0 | 0.0 |
|  | 1996(1) | Hansard | 60 | 95.0 | 5.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 55 | 85.5 | 14.5 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 91.7 | 8.3 | 0.0 | 0.0 |
|  |  | MainArm Sep | 53 | 88.7 | 11.3 | 0.0 | 0.0 |
|  | 1996(2) | Hansard | 60 | 93.3 | 6.7 | 0.0 | 0.0 |
|  |  | Woodpecker | 54 | 79.6 | 20.4 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 88.3 | 11.7 | 0.0 | 0.0 |
|  |  | MainArm Sep | 53 | 84.9 | 15.1 | 0.0 | 0.0 |
| Mountain Whitefish | 1994 | Nechako | 60 | 10.0 | 25.0 | 40.0 | 25.0 |
|  | 1994 | Hansard | 60 | 70.0 | 26.7 | 3.3 | 0.0 |
|  |  | Woodpecker | 64 | 95.3 | 4.7 | 0.0 | 0.0 |
|  |  | Marguerite | 31 | 67.7 | 29.0 | 3.2 | 0.0 |
|  |  | Agassiz | 25 | 88.0 | 12.0 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 98.3 | 1.7 | 0.0 | 0.0 |
|  |  | Thompson | 78 | 67.9 | 28.2 | 3.8 | 0.0 |
|  | 1995 | Nechako | 60 | 30.0 | 43.3 | 23.3 | 3.3 |
|  |  | Hansard | 60 | 90.0 | 10.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 96.7 | 3.3 | 0.0 | 0.0 |
|  |  | Marguerite | 14 | 100.0 | 0.0 | 0.0 | 0.0 |
|  |  | Agassiz | 45 | 86.7 | 13.3 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 81.7 | 16.7 | 1.7 | 0.0 |
|  |  | Thompson | 60 | 70.0 | 30.0 | 0.0 | 0.0 |
|  | 1996(1) | Hansard | 52 | 76.9 | 23.1 | 0.0 | 0.0 |
|  |  | Woodpecker | 38 | 71.1 | 28.9 | 0.0 | 0.0 |
|  | 1996(2) | Hansard | 52 | 76.9 | 23.1 | 0.0 | 0.0 |
|  |  | Woodpecker | 38 | 65.8 | 34.2 | 0.0 | 0.0 |
| Starry Flounder | 1994 | MainArm | 62 | 0.0 | 98.4 | 1.6 | 0.0 |
|  |  | NorthArm | 62 | 12.9 | 80.6 | 6.5 | 0.0 |
|  | 1995 | MainArm | 65 | 29.2 | 43.1 | 12.3 | 15.4 |
|  |  | NorthArm | 60 | 6.7 | 46.7 | 20.0 | 26.7 |

Table 96. Parasites in peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF) 1994 to 1996; column headings are based on field note descriptions.

| Sp. | Reach | Year | Gill Flukes | Gill Copepod | Gill <br> White Spots or Parasites | Skin Copepod or Parasite | Black <br> Spot | Ligula | Encysted Worms | Worms or Nematodes | Air Bladder Nematodes | Gut <br> Parasites | Kidney <br> Parasites |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | Nechako | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ |  |  |  |  | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |  | x |  |
|  | Hansard | $\begin{aligned} & 1994 \\ & 1995 \\ & 1996 \end{aligned}$ |  |  | x |  | $\begin{aligned} & \mathrm{x} \\ & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |  | x |  |  |  |
|  | Woodpecker | $\begin{aligned} & 1994 \\ & 1995 \\ & 1996 \end{aligned}$ |  |  | x |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  | x | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |  |  |
|  | Marguerite | $\begin{aligned} & \hline 1994 \\ & 1995 \end{aligned}$ | x |  | x |  | X |  | x | x |  |  |  |
|  | Agassiz | $\begin{aligned} & \hline 1994 \\ & 1995 \end{aligned}$ | x |  |  |  | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ |  | x | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ |  |  |  |
|  | Mission | 1994 |  |  |  |  |  |  |  | X |  |  |  |
|  | Barnston | 1994 |  |  |  |  | x |  | x | x |  |  |  |
|  | MainArm | $\begin{aligned} & 1994 \\ & 1995 \\ & 1996 \end{aligned}$ |  |  | x |  | X |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ |  |  |  |
|  | NorthArm | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ |  |  | $\mathbf{x}$ |  | x |  | X |  |  |  |  |
|  | N.Thompson | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ |  | X | X |  |  | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ | X | X |  |  |  |
|  | Thompson | $\begin{aligned} & \hline 1994 \\ & 1995 \end{aligned}$ |  |  | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ |  | x | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  | x |  | x |  |
| MW | Nechako | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | x | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ |  | x | $\begin{aligned} & x \\ & \mathbf{x} \end{aligned}$ |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $x$ $\mathbf{x}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |
|  | Hansard | $\begin{aligned} & 1994 \\ & 1995 \\ & 1996 \end{aligned}$ | $\begin{aligned} & x \\ & x \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | x | x |  |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | X |  | $\mathbf{x}$ | $\begin{aligned} & \mathrm{x} \\ & \mathrm{x} \\ & \mathrm{x} \end{aligned}$ |
|  | Woodpecker | $\begin{aligned} & 1994 \\ & 1995 \\ & 1996 \end{aligned}$ | x | x | x |  |  |  | x | x |  |  |  |
|  | Marguerite | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | x |  |  |  | X |  | X |  |  |  | X |
|  | Agassiz | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ |  |  |  |  |  |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |
|  | N.Thompson | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | x | x |  |  |  |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |  |  |  |
|  | Thompson | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |  |  |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  | x | X |
| SF | MainArm | $\begin{aligned} & 1994 \\ & 1995 \end{aligned}$ |  |  |  |  |  |  | x | x |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |
|  | NorthArm | 1994 <br> 1995 |  |  |  |  |  |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  | $\begin{aligned} & \mathbf{x} \\ & \mathbf{x} \end{aligned}$ |  |

Table 97. Tapeworm (Ligula intestinalis) incidence and weights in peamouth chub from the North Thompson and Thompson rivers in 1994 and 1995.

| Reach | Year | Frequency <br> $(\%)$ | N | Mean Worm <br> Weight $(\boldsymbol{g})$ |
| :--- | :---: | :---: | :---: | :---: |
| N.Thompson | 1994 | 26.7 | 16 |  |
| Thompson | 1994 | 28.3 | 17 |  |
| N.Thompson | 1995 | 46.7 | 28 | 6.9 |
| Thompson | 1995 | 6.7 | 4 | 4.5 |

Table 98. Necropsy bile ratings -- percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder

| Species | Year | Reach | N | Bile 0 | Bile 1 | Bile 2 | Bile 3 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 59 | 1.7 | 39.0 | 33.9 | 25.4 |
|  |  | Hansard | 59 | 10.2 | 11.9 | 62.7 | 15.3 |
|  |  | Woodpecker | 60 | 28.3 | 48.3 | 18.3 | 5.0 |
|  |  | Marguerite | 59 | 16.9 | 78.0 | 5.1 | 0.0 |
|  |  | Agassiz | 60 | 31.7 | 41.7 | 21.7 | 5.0 |
|  |  | Mission | 57 | 1.8 | 19.3 | 68.4 | 10.5 |
|  |  | Barnston | 62 | 22.6 | 40.3 | 32.3 | 4.8 |
|  |  | MainArm | 62 | 3.2 | 24.2 | 72.6 | 0.0 |
|  |  | NorthArm | 62 | 8.1 | 11.3 | 66.1 | 14.5 |
|  |  | N.Thompson | 60 | 40.0 | 36.7 | 21.7 | 1.7 |
|  |  | Thompson | 58 | 50.0 | 48.3 | 1.7 | 0.0 |
|  | 1995 | Nechako | 60 | 23.3 | 16.7 | 48.3 | 11.7 |
|  |  | Hansard | 56 | 33.9 | 37.5 | 17.9 | 10.7 |
|  |  | Woodpecker | 60 | 18.3 | 71.7 | 8.3 | 1.7 |
|  |  | Marguerite | 30 | 33.3 | 56.7 | 10.0 | 0.0 |
|  |  | Agassiz | 60 | 80.0 | 20.0 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 41.7 | 23.3 | 25.0 | 10.0 |
|  |  | NorthArm | 60 | 21.7 | 40.0 | 26.7 | 11.7 |
|  |  | N.Thompson | 60 | 50.0 | 43.3 | 6.7 | 0.0 |
|  |  | Thompson | 60 | 63.3 | 28.3 | 8.3 | 0.0 |
|  | 1996(1) | Hansard | 60 | 26.7 | 28.3 | 20.0 | 25.0 |
|  |  | Woodpecker | 55 | 20.0 | 56.4 | 21.8 | 1.8 |
|  |  | MainArm | 60 | 38.3 | 41.7 | 15.0 | 5.0 |
|  |  | MainArm Sep | 53 | 54.7 | 24.5 | 20.8 | 0.0 |
|  | 1996(2) | Hansard | 60 | 20.0 | 26.7 | 25.0 | 28.3 |
|  |  | Woodpecker | 53 | 9.4 | 67.9 | 22.6 | 0.0 |
|  |  | MainArm | 60 | 33.3 | 43.3 | 18.3 | 5.0 |
|  |  | MainArm Sep | 51 | 54.9 | 31.4 | 13.7 | 0.0 |
| Mountain Whitefish | 1994 | Nechako | 57 | 3.5 | 5.3 | 43.9 | 47.4 |
|  |  | Hansard | 58 | 1.7 | 0.0 | 41.4 | 56.9 |
|  |  | Woodpecker | 64 | 3.1 | 0.0 | 39.1 | 57.8 |
|  |  | Marguerite | 31 | 38.7 | 35.5 | 22.6 | 3.2 |
|  |  | Agassiz | 25 | 64.0 | 8.0 | 28.0 | 0.0 |
|  |  | N.Thompson | 59 | 5.1 | 0.0 | 40.7 | 54.2 |
|  |  | Thompson | 78 | 14.1 | 23.1 | 61.5 | 1.3 |
|  | 1995 | Nechako | 60 | 48.3 | 1.7 | 35.0 | 15.0 |
|  |  | Hansard | 60 | 13.3 | 3.3 | 68.3 | 15.0 |
|  |  | Woodpecker | 59 | 1.7 | 0.0 | 84.7 | 13.6 |
|  |  | Marguerite | 14 | 28.6 | 7.1 | 57.1 | 7.1 |
|  |  | Agassiz | 45 | 91.1 | 0.0 | 8.9 | 0.0 |
|  |  | N.Thompson | 60 | 46.7 | 0.0 | 36.7 | 16.7 |
|  |  | Thompson | 60 | 41.7 | 0.0 | 55.0 | 3.3 |
|  | 1996(1) | Hansard | 52 | 32.7 | 1.9 | 53.8 | 11.5 |
|  |  | Woodpecker | 38 | 7.9 | 0.0 | 71.1 | 21.1 |
|  | 1996(2) | Hansard | 52 | 25.0 | 15.4 | 42.3 | 17.3 |
|  |  | Woodpecker | 38 | 7.9 | 0.0 | 73.7 | 18.4 |
| Starry Flounder | 1994 | MainArm | 62 | 1.6 | 0.0 | 43.5 | 54.8 |
|  |  | NorthArm | 62 | 0.0 | 0.0 | 21.0 | 79.0 |
|  | 1995 | MainArm | 65 | 1.5 | 1.5 | 66.2 | 30.8 |
|  |  | NorthArm | 59 | 0.0 | 0.0 | 62.7 | 37.3 |

Table 99. Necropsy liver condition -- percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | Nodular | Fatty | Discoloured | Normal |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 58.1 | 58.1 | 30.6 | 1.6 |
|  |  | Hansard | 60 | 3.3 | 31.7 | 38.3 | 31.7 |
|  |  | Woodpecker | 60 | 13.3 | 6.7 | 85.0 | 5.0 |
|  |  | Marguerite | 60 | 1.7 | 10.0 | 70.0 | 18.3 |
|  |  | Agassiz | 60 | 0.0 | 25.0 | 41.7 | 33.3 |
|  |  | Mission | 57 | 5.3 | 66.7 | 3.5 | 26.3 |
|  |  | Barnston | 62 | 11.3 | 62.9 | 9.7 | 22.6 |
|  |  | MainArm | 62 | 19.4 | 79.0 | 30.6 | 3.2 |
|  |  | NorthArm | 62 | 1.6 | 93.5 | 3.2 | 4.8 |
|  |  | N.Thompson | 60 | 0.0 | 23.3 | 20.0 | 56.7 |
|  |  | Thompson | 60 | 0.0 | 90.0 | 11.7 | 1.7 |
|  | 1995 | Nechako | 60 | 11.7 | 73.3 | 5.0 | 8.3 |
|  |  | Hansard | 56 | 1.8 | 62.5 | 8.9 | 28.6 |
|  |  | Woodpecker | 60 | 0.0 | 60.0 | 36.7 | 5.0 |
|  |  | Marguerite | 30 | 0.0 | 46.7 | 36.7 | 16.7 |
|  |  | Agassiz | 61 | 0.0 | 96.7 | 0.0 | 3.3 |
|  |  | MainArm | 60 | 1.7 | 88.3 | 18.3 | 6.7 |
|  |  | NorthArm | 60 | 1.7 | 88.3 | 15.0 | 5.0 |
|  |  | N.Thompson | 60 | 0.0 | 40.0 | 0.0 | 60.0 |
|  |  | Thompson | 60 | 0.0 | 98.3 | 0.0 | 1.7 |
|  | 1996(1) | Hansard | 60 | 0.0 | 81.7 | 0.0 | 18.3 |
|  |  | Woodpecker | 55 | 0.0 | 29.1 | 61.8 | 9.1 |
|  |  | MainArm | 60 | 1.7 | 93.3 | 0.0 | 5.0 |
|  |  | MainArm Sep | 53 | 3.8 | 69.8 | 9.4 | 17.0 |
|  | 1996(2) | Hansard | 60 | 0.0 | 76.7 | 0.0 | 23.3 |
|  |  | Woodpecker | 54 | 0.0 | 42.6 | 50.0 | 25.9 |
|  |  | MainArm | 60 | 0.0 | 90.0 | 0.0 | 8.3 |
|  |  | MainArm Sep | 53 | 3.8 | 58.5 | 13.2 | 24.5 |
| Mountain Whitfish | 1994 | Nechako | 60 | 11.7 | 0.0 | 8.3 | 80.0 |
|  |  | Hansard | 60 | 5.0 | 10.0 | 8.3 | 78.3 |
|  |  | Woodpecker | 64 | 3.1 | 1.6 | 14.1 | 81.3 |
|  |  | Marguerite | 31 | 0.0 | 0.0 | 61.3 | 38.7 |
|  |  | Agassiz | 25 | 4.0 | 16.0 | 20.0 | 60.0 |
|  |  | N.Thompson | 60 | 0.0 | 5.0 | 8.3 | 86.7 |
|  |  | Thompson | 78 | 6.4 | 1.3 | 7.7 | 84.6 |
|  | 1995 | Nechako | 60 | 5.0 | 1.7 | 0.0 | 95.0 |
|  |  | Hansard | 60 | 3.3 | 1.7 | 0.0 | 95.0 |
|  |  | Woodpecker | 60 | 1.7 | 0.0 | 5.0 | 93.3 |
|  |  | Marguerite | 14 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Agassiz | 45 | 8.9 | 4.4 | 2.2 | 82.2 |
|  |  | N.Thompson | 60 | 0.0 | 0.0 | 1.7 | 98.3 |
|  |  | Thompson | 60 | 3.3 | 0.0 | 0.0 | 96.7 |
|  | 1996(1) | Hansard | 52 | 0.0 | 0.0 | 1.9 | 98.1 |
|  |  | Woodpecker | 38 | 0.0 | 0.0 | 2.6 | 97.4 |
|  | 1996(2) | Hansard | 52 | 0.0 | 1.9 | 1.9 | 96.2 |
|  |  | Woodpecker | 38 | 0.0 | 0.0 | 0.0 | 100.0 |
| Starry Flounder | 1994 | MainArm | 62 | 43.5 | 14.5 | 9.7 | 45.2 |
|  |  | NorthArm | 62 | 27.4 | 25.8 | 12.9 | 45.2 |
|  | 1995 | MainArm | 65 | 16.9 | 20.0 | 4.6 | 61.5 |
|  |  | NorthArm | 60 | 23.3 | 55.0 | 25.0 | 15.0 |

Table 100. Necropsy mesenteric fat ratings -- percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | Fat 0 | Fat 1 | Fat 2 | Fat 3 | Fat 4 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 0.0 | 3.2 | 6.5 | 45.2 | 45.2 |
|  |  | Hansard | 60 | 0.0 | 23.3 | 21.7 | 43.3 | 11.7 |
|  |  | Woodpecker | 60 | 0.0 | 26.7 | 16.7 | 30.0 | 26.7 |
|  |  | Marguerite | 59 | 3.4 | 39.0 | 27.1 | 22.0 | 8.5 |
|  |  | Agassiz | 60 | 15.0 | 45.0 | 25.0 | 3.3 | 11.7 |
|  |  | Mission | 57 | 0.0 | 10.5 | 5.3 | 15.8 | 68.4 |
|  |  | Barnston | 62 | 1.6 | 22.6 | 17.7 | 25.8 | 32.3 |
|  |  | MainArm | 62 | 0.0 | 4.8 | 1.6 | 38.7 | 54.8 |
|  |  | NorthArm | 61 | 0.0 | 3.3 | 9.8 | 26.2 | 60.7 |
|  |  | N.Thompson | 60 | 5.0 | 50.0 | 20.0 | 18.3 | 6.7 |
|  |  | Thompson | 60 | 0.0 | 10.0 | 1.7 | 25.0 | 63.3 |
|  | 1995 | Nechako | 60 | 0.0 | 15.0 | 25.0 | 38.3 | 21.7 |
|  |  | Hansard | 56 | 0.0 | 17.9 | 23.2 | 28.6 | 30.4 |
|  |  | Woodpecker | 60 | 0.0 | 3.3 | 10.0 | 11.7 | 75.0 |
|  |  | Marguerite | 30 | 6.7 | 20.0 | 40.0 | 16.7 | 16.7 |
|  |  | Agassiz | 61 | 0.0 | 1.6 | 0.0 | 1.6 | 96.7 |
|  |  | MainArm | 60 | 0.0 | 5.0 | 13.3 | 20.0 | 61.7 |
|  |  | NorthArm | 60 | 1.7 | 0.0 | 1.7 | 25.0 | 71.7 |
|  |  | N.Thompson | 60 | 1.7 | 18.3 | 8.3 | 20.0 | 51.7 |
|  |  | Thompson | 60 | 0.0 | 3.3 | 5.0 | 28.3 | 63.3 |
|  | 1996(1) | Hansard | 60 | 0.0 | 0.0 | 5.0 | 25.0 | 70.0 |
|  |  | Woodpecker | 55 | 3.6 | 3.6 | 7.3 | 32.7 | 52.7 |
|  |  | MainArm | 60 | 0.0 | 10.0 | 1.7 | 11.7 | 76.7 |
|  |  | MainArm Sep | 53 | 0.0 | 0.0 | 7.5 | 41.5 | 50.9 |
|  | 1996(2) | Hansard | 60 | 0.0 | 1.7 | 11.7 | 26.7 | 60.0 |
|  |  | Woodpecker | 54 | 0.0 | 5.6 | 24.1 | 33.3 | 37.0 |
|  |  | MainArm | 60 | 3.3 | 3.3 | 10.0 | 20.0 | 63.3 |
|  |  | MainArm Sep | 53 | 0.0 | 0.0 | 7.5 | 20.8 | 71.7 |
| Mountain Whitefish | 1994 | Nechako | 60 | 23.3 | 30.0 | 23.3 | 21.7 | 1.7 |
|  |  | Hansard | 60 | 0.0 | 61.7 | 16.7 | 16.7 | 5.0 |
|  |  | Woodpecker | 64 | 9.4 | 73.4 | 14.1 | 3.1 | 0.0 |
|  |  | Marguerite | 31 | 6.5 | 61.3 | 12.9 | 19.4 | 0.0 |
|  |  | Agassiz | 25 | 20.0 | 56.0 | 24.0 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 10.0 | 25.0 | 18.3 | 43.3 | 3.3 |
|  |  | Thompson | 78 | 1.3 | 12.8 | 7.7 | 23.1 | 55.1 |
|  | 1995 | Nechako | 60 | 6.7 | 31.7 | 18.3 | 23.3 | 20.0 |
|  |  | Hansard | 60 | 0.0 | 26.7 | 38.3 | 18.3 | 16.7 |
|  |  | Woodpecker | 60 | 23.3 | 41.7 | 28.3 | 6.7 | 0.0 |
|  |  | Marguerite | 14 | 21.4 | 57.1 | 14.3 | 7.1 | 0.0 |
|  |  | Agassiz | 45 | 4.4 | 11.1 | 24.4 | 28.9 | 28.9 |
|  |  | N.Thompson | 60 | 6.7 | 35.0 | 8.3 | 8.3 | 41.7 |
|  |  | Thompson | 60 | 1.7 | 10.0 | 25.0 | 21.7 | 41.7 |
|  | 1996(1) | Hansard | 52 | 9.6 | 63.5 | 13.5 | 11.5 | 1.9 |
|  |  | Woodpecker | 38 | 21.1 | 73.7 | 5.3 | 0.0 | 0.0 |
|  | 1996(2) | Hansard | 52 | 9.6 | 75.0 | 7.7 | 7.7 | 0.0 |
|  |  | Woodpecker | 38 | 2.6 | 71.1 | 18.4 | 5.3 | 2.6 |
| Starry Flounder | 1994 | MainArm | 62 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | NorthArm | 62 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  | 1995 | MainArm | 65 | 98.5 | 1.5 | 0.0 | 0.0 | 0.0 |
|  |  | NorthArm | 60 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |

Table 101. Necropsy spleen condition -- percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | Nodular | Enlarged | Other | Granular | Normal |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 4.8 | 6.5 | 0.0 | 3.2 | 87.1 |
|  |  | Hansard | 60 | 0.0 | 5.0 | 0.0 | 1.7 | 93.3 |
|  |  | Woodpecker | 60 | 0.0 | 0.0 | 0.0 | 1.7 | 98.3 |
|  |  | Marguerite | 60 | 0.0 | 0.0 | 0.0 | 1.7 | 98.3 |
|  |  | Agassiz | 60 | 0.0 | 3.3 | 0.0 | 0.0 | 96.7 |
|  |  | Mission | 57 | 3.5 | 3.5 | 0.0 | 10.5 | 84.2 |
|  |  | Barnston | 62 | 0.0 | 6.5 | 0.0 | 9.7 | 87.1 |
|  |  | MainArm | 62 | 1.6 | 0.0 | 3.2 | 6.5 | 88.7 |
|  |  | NorthArm | 62 | 0.0 | 3.2 | 0.0 | 8.1 | 91.9 |
|  |  | N.Thompson | 60 | 0.0 | 25.0 | 0.0 | 3.3 | 71.7 |
|  |  | Thompson | 60 | 0.0 | 3.3 | 0.0 | 8.3 | 90.0 |
|  | 1995 | Nechako | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Hansard | 56 | 5.4 | 1.8 | 0.0 | 3.6 | 91.1 |
|  |  | Woodpecker | 60 | 0.0 | 5.0 | 0.0 | 21.7 | 73.3 |
|  |  | Marguerite | 30 | 0.0 | 3.3 | 0.0 | 0.0 | 96.7 |
|  |  | Agassiz | 61 | 1.6 | 0.0 | 0.0 | 0.0 | 98.4 |
|  |  | MainArm | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | NorthArm | 60 | 0.0 | 13.3 | 1.7 | 11.7 | 81.7 |
|  |  | N.Thompson | 60 | 0.0 | 0.0 | 0.0 | 5.0 | 95.0 |
|  |  | Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  | 1996(1) | Hansard | 60 | 0.0 | 3.3 | 0.0 | 18.3 | 80.0 |
|  |  | Woodpecker | 55 | 0.0 | 0.0 | 0.0 | 1.8 | 98.2 |
|  |  | MainArm | 60 | 0.0 | 1.7 | 0.0 | 0.0 | 98.3 |
|  |  | MainArm Sep | 53 | 0.0 | 0.0 | 0.0 | 5.7 | 94.3 |
|  | 1996(2) | Hansard | 60 | 0.0 | 1.7 | 0.0 | 16.7 | 81.7 |
|  |  | Woodpecker | 55 | 0.0 | 0.0 | 0.0 | 1.8 | 98.2 |
|  |  | MainArm | 60 | 0.0 | 5.0 | 0.0 | 0.0 | 95.0 |
|  |  | MainArm Sep | 53 | 1.9 | 1.9 | 0.0 | 5.7 | 90.6 |
| Mountain Whitefish | 1994 | Nechako | 60 | 0.0 | 5.0 | 1.7 | 30.0 | 65.0 |
|  |  | Hansard | 60 | 0.0 | 1.7 | 5.0 | 8.3 | 85.0 |
|  |  | Woodpecker | 64 | 1.6 | 3.1 | 0.0 | 14.1 | 81.3 |
|  |  | Marguerite | 31 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Agassiz | 25 | 0.0 | 4.0 | 0.0 | 0.0 | 96.0 |
|  |  | N.Thompson | 60 | 0.0 | 1.7 | 0.0 | 13.3 | 86.7 |
|  |  | Thompson | 77 | 1.3 | 6.5 | 0.0 | 14.3 | 84.4 |
|  | 1995 | Nechako | 60 | 1.7 | 1.7 | 5.0 | 15.0 | 76.7 |
|  |  | Hansard | 60 | 0.0 | 5.0 | 0.0 | 43.3 | 56.7 |
|  |  | Woodpecker | 60 | 0.0 | 0.0 | 0.0 | 1.7 | 98.3 |
|  |  | Marguerite | 14 | 0.0 | 7.1 | 0.0 | 7.1 | 92.9 |
|  |  | Agassiz | 45 | 0.0 | 0.0 | 0.0 | 2.2 | 97.8 |
|  |  | N.Thompson | 60 | 0.0 | 1.7 | 0.0 | 53.3 | 46.7 |
|  |  | Thompson | 60 | 0.0 | 1.7 | 0.0 | 0.0 | 98.3 |
|  | 1996(1) | Hansard | 52 | 5.8 | 3.8 | 0.0 | 17.3 | 76.9 |
|  |  | Woodpecker | 38 | 0.0 | 2.6 | 0.0 | 5.3 | 92.1 |
|  | 1996(2) | Hansard | 52 | 17.3 | 1.9 | 0.0 | 3.8 | 78.8 |
|  |  | Woodpecker | 38 | 0.0 | 2.6 | 0.0 | 7.9 | 89.5 |
| Starry Flounder | 1994 | MainArm | 62 | 0.0 | 4.8 | 0.0 | 0.0 | 95.2 |
|  |  | NorthArm | 62 | 0.0 | 3.2 | 0.0 | 0.0 | 96.8 |
|  | 1995 | MainArm | 65 | 0.0 | 0.0 | 0.0 | 1.5 | 98.5 |
|  |  | NorthArm | 60 | 0.0 | 1.7 | 0.0 | 0.0 | 98.3 |

Table 102. Necropsy hindgut ratings -- percent occurrence in each reach and year for peamouth chub, mountain whitefish, and starry flounder.

| Species | Year | Reach | N | Hindgut 0 | Hindgut 1 | Hindgut 2 |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub | 1994 | Nechako | 62 | 82.3 | 16.1 | 1.6 |
|  |  | Hansard | 60 | 38.3 | 43.3 | 18.3 |
|  |  | Woodpecker | 60 | 46.7 | 31.7 | 20.0 |
|  |  | Marguerite | 60 | 95.0 | 3.3 | 1.7 |
|  |  | Agassiz | 60 | 58.3 | 28.3 | 13.3 |
|  |  | Mission | 57 | 91.2 | 8.8 | 0.0 |
|  |  | Barnston | 62 | 85.5 | 14.5 | 0.0 |
|  |  | MainArm | 62 | 53.2 | 41.9 | 4.8 |
|  |  | NorthArm | 62 | 27.4 | 58.1 | 14.5 |
|  |  | N.Thompson | 60 | 96.7 | 3.3 | 0.0 |
|  |  | Thompson | 60 | 83.3 | 10.0 | 6.7 |
|  | 1995 | Nechako | 60 | 88.3 | 6.7 | 5.0 |
|  |  | Hansard | 56 | 92.9 | 7.1 | 0.0 |
|  |  | Woodpecker | 60 | 68.3 | 28.3 | 3.3 |
|  |  | Marguerite | 30 | 40.0 | 53.3 | 6.7 |
|  |  | Agassiz | 61 | 8.2 | 85.2 | 6.6 |
|  |  | MainArm | 60 | 30.0 | 58.3 | 8.3 |
|  |  | NorthArm | 60 | 83.3 | 16.7 | 0.0 |
|  |  | N.Thompson | 60 | 98.3 | 1.7 | 0.0 |
|  |  | Thompson | 60 | 95.0 | 5.0 | 0.0 |
|  | 1996(1) | Hansard | 60 | 98.3 | 1.7 | 0.0 |
|  |  | Woodpecker | 55 | 98.2 | 1.8 | 0.0 |
|  |  | MainArm | 60 | 86.7 | 11.7 | 1.7 |
|  |  | MainArm Sep | 53 | 62.3 | 35.8 | 1.9 |
|  | 1996(2) | Hansard | 60 | 96.7 | 3.3 | 0.0 |
|  |  | Woodpecker | 53 | 100.0 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 90.0 | 8.3 | 1.7 |
|  |  | MainArm Sep | 53 | 64.2 | 24.5 | 11.3 |
| Mountain Whitefish | 1994 | Nechako | 60 | 83.3 | 15.0 | 1.7 |
|  |  | Hansard | 60 | 65.0 | 26.7 | 8.3 |
|  |  | Woodpecker | 64 | 56.3 | 35.9 | 7.8 |
|  |  | Marguerite | 31 | 90.3 | 6.5 | 3.2 |
|  |  | Agassiz | 25 | 64.0 | 32.0 | 4.0 |
|  |  | N.Thompson | 60 | 80.0 | 18.3 | 1.7 |
|  |  | Thompson | 78 | 93.6 | 6.4 | 0.0 |
|  | 1995 | Nechako | 60 | 86.7 | 13.3 | 0.0 |
|  |  | Hansard | 60 | 98.3 | 1.7 | 0.0 |
|  |  | Woodpecker | 60 | 70.0 | 28.3 | 1.7 |
|  |  | Marguerite | 14 | 28.6 | 57.1 | 14.3 |
|  |  | Agassiz | 45 | 97.8 | 0.0 | 2.2 |
|  |  | N.Thompson | 60 | 98.3 | 0.0 | 1.7 |
|  |  | Thompson | 60 | 98.3 | 0.0 | 1.7 |
|  | 1996(1) | Hansard | 52 | 98.1 | 1.9 | 0.0 |
|  |  | Woodpecker | 38 | 94.7 | 5.3 | 0.0 |
|  | 1996(2) | Hansard | 52 | 82.7 | 17.3 | 0.0 |
|  |  | Woodpecker | 38 | 100.0 | 0.0 | 0.0 |
| Starry Flounder | 1994 | MainArm | 62 | 69.4 | 27.4 | 3.2 |
|  |  | NorthArm | 62 | 83.9 | 16.1 | 0.0 |
|  | 1995 | MainArm | 65 | 75.4 | 23.1 | 1.5 |
|  |  | NorthArm | 60 | 91.7 | 8.3 | 0.0 |

Table 103. Necropsy kidney condition -- percent occurrence in each reach and year for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF).

| Species | Year | Reach | N | Mottled | Granular | Urolithic | Pink | Brown or Grey | Parasites | Other | Normal |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1994 | Nechako | 62 | 14.5 | 3.2 | 1.6 | 75.8 | 4.8 | 0.0 | 0.0 | 0.0 |
|  |  | Hansard | 60 | 3.3 | 3.3 | 0.0 | 83.3 | 8.3 | 0.0 | 1.7 | 0.0 |
|  |  | Woodpecker | 60 | 3.3 | 1.7 | 0.0 | 70.0 | 23.3 | 0.0 | 0.0 | 1.7 |
|  |  | Marguerite | 60 | 5.0 | 3.3 | 0.0 | 75.0 | 16.7 | 0.0 | 0.0 | 0.0 |
|  |  | Agassiz | 60 | 3.3 | 3.3 | 0.0 | 93.3 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | Mission | 57 | 8.8 | 0.0 | 0.0 | 8.8 | 0.0 | 0.0 | 82.5 | 0.0 |
|  |  | Barnston | 62 | 12.9 | 0.0 | 0.0 | 43.5 | 0.0 | 0.0 | 43.5 | 0.0 |
|  |  | MainArm | 62 | 4.8 | 0.0 | 0.0 | 38.7 | 0.0 | 0.0 | 56.5 | 0.0 |
|  |  | NorthArm | 62 | 4.8 | 0.0 | 0.0 | 58.1 | 1.6 | 0.0 | 35.5 | 0.0 |
|  |  | N.Thompson | 60 | 1.7 | 0.0 | 0.0 | 98.3 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | Thompson | 60 | 10.0 | 0.0 | 0.0 | 90.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  | 1995 | Nechako | 60 | 0.0 | 0.0 | 0.0 | 98.3 | 1.7 | 0.0 | 0.0 | 0.0 |
|  |  | Hansard | 56 | 0.0 | 0.0 | 0.0 | 98.2 | 1.8 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 60 | 0.0 | 0.0 | 0.0 | 61.7 | 38.3 | 0.0 | 0.0 | 0.0 |
|  |  | Marguerite | 30 | 0.0 | 0.0 | 0.0 | 43.3 | 50.0 | 0.0 | 0.0 | 6.7 |
|  |  | Agassiz | 61 | 0.0 | 0.0 | 0.0 | 85.2 | 0.0 | 0.0 | 0.0 | 14.8 |
|  |  | MainArm | 60 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | NorthArm | 60 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | N.Thompson | 60 | 1.7 | 0.0 | 0.0 | 98.3 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | Thompson | 60 | 6.7 | 0.0 | 0.0 | 93.3 | 0.0 | 0.0 | 0.0 | 0.0 |
|  | 1996(1) | Hansard | 60 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 55 | 0.0 | 0.0 | 0.0 | 63.6 | 36.4 | 0.0 | 0.0 | 0.0 |
|  |  | MainArm | 60 | 0.0 | 0.0 | 0.0 | 98.3 | 0.0 | 0.0 | 1.7 | 0.0 |
|  |  | MainArm Sep | 53 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  | 1996(2) | Hansard | 60 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | Woodpecker | 53 | 0.0 | 0.0 | 0.0 | 54.7 | 43.4 | 0.0 | 1.9 | 0.0 |
|  |  | MainArm | 60 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
|  |  | MainArm Sep | 53 | 0.0 | 0.0 | 0.0 | 100.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| MW | 1994 | Nechako | 60 | 18.3 | 1.7 | 11.7 | 0.0 | 5.0 | 23.3 | 0.0 | 40.0 |
|  |  | Hansard | 60 | 6.7 | 6.7 | 0.0 | 0.0 | 0.0 | 8.3 | 0.0 | 76.7 |
|  |  | Woodpecker | 64 | 0.0 | 3.1 | 0.0 | 0.0 | 1.6 | 0.0 | 0.0 | 95.3 |
|  |  | Marguerite | 31 | 6.5 | 6.5 | 6.5 | 0.0 | 6.5 | 12.9 | 0.0 | 61.3 |
|  |  | Agassiz | 25 | 12.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 88.0 |
|  |  | N.Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Thompson | 78 | 9.0 | 5.1 | 2.6 | 0.0 | 0.0 | 7.7 | 1.3 | 74.4 |
|  | 1995 | Nechako | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 3.3 | 0.0 | 0.0 | 96.7 |
|  |  | Hansard | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 1.7 | 0.0 | 98.3 |
|  |  | Woodpecker | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Marguerite | 14 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Agassiz | 45 | 2.2 | 0.0 | 0.0 | 0.0 | 0.0 | 11.1 | 0.0 | 86.7 |
|  |  | N.Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | Thompson | 60 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 1.7 | 98.3 |
|  | 1996(1) | Hansard | 52 | 0.0 | 0.0 | 19.2 | 0.0 | 0.0 | 0.0 | 0.0 | 80.8 |
|  |  | Woodpecker | 38 | 0.0 | 0.0 | 34.2 | 0.0 | 13.2 | 0.0 | 0.0 | 52.6 |
|  | 1996(2) | Hansard | 52 | 0.0 | 0.0 | 19.2 | 0.0 | 0.0 | 0.0 | 0.0 | 80.8 |
|  |  | Woodpecker | 38 | 0.0 | 0.0 | 28.9 | 0.0 | 5.3 | 0.0 | 0.0 | 65.8 |
| SF | 1994 | MainArm | 62 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | NorthArm | 62 | 3.2 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 96.8 |
|  | 1995 | MainArm | 65 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
|  |  | NorthArm | 60 | 0.0 | 1.7 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 98.3 |

Table 104. Histology blind duplicate results -- percent occurrence of conditions by histologist.
PC=Peamouth chub; MW=Mountain Whitefish; SF=Starry Flounder

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | $\begin{aligned} & .0 . \\ & \stackrel{0}{0} \\ & \text { in } \\ & \hline \end{aligned}$ | $\geq$ |  |  | $\begin{aligned} & \text { 듞 } \\ & \text { 은 } \\ & \text { 힌 } \\ & \hline \end{aligned}$ |  |  |  |  |  |  |  | 윤 <br> 흘 <br> 흔 |  |  |  |  |  |  |  |  |  |  |  |  |
| A | MW | 88 | 16 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 3 | 10 | 0 | 0 | 0 | 0 | 34 | 0 | 0 | 0 | 0 | 0 | 77 | 0 |
| B |  | 88 | 93 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 5 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 56 | 20 |
| A | PC | 122 | 4 | 0 | 0 | 7 | 0 | 0 | 6 | 12 | 0 | 0 | 19 | 0 | 0 | 0 | 0 | 77 | 0 | 0 | 0 | 0 | 0 | 82 | 0 |
| B |  | 122 | 61 | 2 | 0 | 1 | 0 | 0 | 5 | 1 | 0 | 1 | 7 | 0 | 0 | 0 | 0 | 54 | 0 | 0 | 0 | 0 | 0 | 59 | 20 |
| A | SF | 28 | 71 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 21 | 0 | 0 | 0 | 0 | 0 | 25 | 0 |
| B |  | 28 | 96 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 14 | 7 |


|  | $\begin{aligned} & .0 .0 \\ & \text { © } \\ & \text { io } \\ & 0 \end{aligned}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | Melanin Melanosis |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| A | MW | 87 | 89 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 1 | 0 | 0 | 14 | 0 | 2 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| B |  | 87 | 95 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 2 | 0 | 0 | 0 | 0 | 3 |
| A | PC | 122 | 66 | 0 | 0 | 2 | 25 | 0 | 0 | 0 | 0 | 0 | 14 | 0 | 0 | 25 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| B |  | 122 | 96 | 0 | 1 | 0 | 20 | 2 | 16 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 2 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 6 |
| A | SF | 30 | 90 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 3 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 13 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 |
| B |  | 30 | 97 | 3 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 |

Table 104. Histology blind duplicate results -- percent occurrence of conditions by histologist. (continued)

| SPL |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | 2 |  |  |  | $\frac{\stackrel{\rightharpoonup}{0}}{\substack{2}}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | $\begin{aligned} & \text { 든 } \\ & \text { 을 } \end{aligned}$ |  |
| A | MW | 87 | 77 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 14 | 0 | 11 | 0 | 0 | 0 | 0 | 0 |
| B |  | 87 | 98 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 5 | 0 | 1 | 0 | 0 | 0 | 0 | 1 |
| A | PC | 120 | 98 | 98 | 8 | 1 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 8 | 0 | 0 | 1 | 0 | 0 |
| B |  | 120 | 95 | 98 | 3 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 |
| A | SF | 30 | 90 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 |
| B |  | 30 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |


|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | 2 |  |  |  |  |  | $\stackrel{\rightharpoonup}{\stackrel{\rightharpoonup}{0}}$ |  |  | 든 든 든 듣 |  |  |  |  |  |  |  |  |  |  |  |  |  |
| A | MW | 85 | 96 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 0 |
| B |  | 85 | 99 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 6 |
| A | PC | 121 | 98 | 0 | 88 | 0 | 9 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 1 | 0 |
| B |  | 121 | 97 | 0 | 98 | 0 | 5 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0 | 3 |
| A | SF | 29 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| B |  | 29 | 97 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 104. Histology blind duplicate results -- percent occurrence of conditions by histologist. (continued)


|  | .$\ddot{0}$ $\stackrel{0}{0}$ 0 0 | $\geq$ |  |  |  |  |  | [ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| A | MW | 61 | 31 | 0 | 0 | 0 | 0 | 0 |
| B |  | 61 | 31 | 0 | 0 | 0 | 0 | 0 |
| A | PC | 103 | 66 | 0 | 0 | 0 | 0 | 0 |
| B |  | 103 | 66 | 0 | 0 | 0 | 2 | 14 |
| A | SF | 29 | 55 | 0 | 0 | 0 | 0 | 0 |
| B |  | 29 | 55 | 0 | 0 | 0 | 3 | 41 |

Table 105. Histology field splits results -- percent occurrence of conditions by split.
PC=Peamouth chub; MW=Mountain Whitefish; SF=Starry Flounder

| GIL |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | $\begin{aligned} & .0 \\ & \stackrel{0}{0} \\ & \text { io } \\ & \hline 0 \end{aligned}$ | 2 | $\begin{aligned} & \overline{\widetilde{0}} \\ & \stackrel{\rightharpoonup}{0} \\ & \stackrel{y}{2} \end{aligned}$ |  |  |  |  |  |  |  |  |  |  |  |  | $\begin{array}{\|l\|l} \stackrel{n}{0} \\ \stackrel{0}{2} \\ \stackrel{\Delta}{2} \\ \hline \end{array}$ |  |  |  |  |  |  |  |  |  |
| 1 | MW | 58 | 97 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 7 | 0 | 0 | 0 | 0 | 16 | 0 | 0 | 0 | 0 | 0 | 26 | 17 |
| 2 |  | 58 | 97 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 28 | 31 |
| 1 | PC | 93 | 34 | 1 | 0 | 3 | 0 | 0 | 5 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 72 | 0 | 0 | 0 | 0 | 0 | 35 | 34 |
| 2 |  | 93 | 24 | 1 | 0 | 3 | 0 | 0 | 11 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 78 | 0 | 0 | 0 | 0 | 0 | 33 | 44 |
| 1 | SF | 9 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 9 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |


| LIV |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $\begin{array}{\|l} \hline \frac{7}{2} \\ 0 \\ \frac{0}{2} \\ i \frac{2}{2} \\ \hline \end{array}$ | $\begin{aligned} & \mathscr{0} \\ & \stackrel{0}{0} \\ & \stackrel{0}{2} \end{aligned}$ | $z$ |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| 1 | MW | 59 | 95 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 5 | 2 | 2 | 0 | 0 | 0 | 2 | 2 | 0 | 0 | 0 | 2 | 7 |
| 2 |  | 59 | 98 | 2 | 0 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 7 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 5 |
| 1 | PC | 93 | 96 | 0 | 0 | 0 | 17 | 4 | 11 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3 |
| 2 |  | 93 | 97 | 0 | 0 | 1 | 23 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| 1 | SF | 11 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 11 | 100 | 9 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9 |

Table 105. Histology field splits results -- percent occurrence of conditions by split. (continued)

| P |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | 2 | $\begin{aligned} & \overline{0} \\ & \stackrel{0}{E} \\ & \text { Z } \end{aligned}$ |  |  | $\stackrel{\pi}{\omega}$ |  |  |  |  |  |  |  | (әбле/әлош) ешоןnuел |  |  |  |  |  |  |  | $\begin{aligned} & 0.0 \\ & \stackrel{0}{0} \\ & 0 \\ & 0 \\ & 0 \\ & \hline 0 \\ & \hline \end{aligned}$ |  | $\begin{aligned} & \text { 들 } \\ & \text { 을 } \end{aligned}$ |  |
| 1 | MW | 55 | 96 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 4 | 0 | 2 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 55 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0 | 0 | 0 | 0 | 0 | 0 | 5 |
| 1 | PC | 90 | 96 | 98 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 90 | 98 | 100 | 1 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 1 | SF | 8 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 8 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |


|  | $\begin{aligned} & \mathscr{0} \\ & \stackrel{0}{0} \\ & \stackrel{0}{\infty} \\ & \hline \end{aligned}$ | $\geq$ | 든 을 |  | 를 0 3 3 0 0 0 0 2 2 |  |  | $\stackrel{\stackrel{\rightharpoonup}{\infty}}{\substack{0}}$ |  |  |  |  |  |  |  |  |  |  |  |  | $\begin{aligned} & \frac{\infty}{\infty} \\ & \frac{1}{0} \\ & 0 \\ & \frac{0}{0} \\ & \stackrel{\rightharpoonup}{5} \end{aligned}$ |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| 1 | MW | 54 | 96 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 4 |
| 2 |  | 54 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 0 | 0 | 0 | 0 | 0 | 7 |
| 1 | PC | 92 | 98 | 0 | 100 | 0 | 3 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 92 | 100 | 0 | 100 | 0 | 3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0 | 0 | 0 |
| 1 | SF | 10 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 |
| 2 |  | 10 | 100 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10 |

Table 105. Histology field splits results -- percent occurrence of conditions by split. (continued)

| HIN | GU |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | $\because$ $\stackrel{0}{0}$ 0 0 0 | $\geq$ |  |  |  | $\qquad$ |  |  | $$ |  |  | Inflammation in pancreas | $$ | $\begin{aligned} & \text { 흘 } \\ & \hline \frac{0}{5} \end{aligned}$ |  |  |
| 1 | MW | 59 | 98 | 0 | 0 | 0 | 3 | 0 | 0 | 0 | 2 | 5 | 0 | 0 | 3 | 0 |
| 2 |  | 59 | 100 | 0 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 1 | PC | 86 | 100 | 3 | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 86 | 100 | 0 | 1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 |
| 1 | SF | 10 | 90 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 10 | 100 | 0 | 0 | 10 | 0 | 0 | 0 | 0 | 10 | 0 | 20 | 0 | 0 | 0 |


| GONAD |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | $\begin{aligned} & .0 .0 \\ & \frac{0}{0} \\ & 0.0 \\ & 0 \end{aligned}$ | 2 | $\stackrel{\stackrel{0}{0}}{\stackrel{0}{0}}$ |  |  |  |  |  |
| 1 | MW | 31 | 35 | 0 | 0 | 0 | 0 | 0 |
| 2 |  | 31 | 35 | 0 | 0 | 0 | 0 | 0 |
| 1 | PC | 55 | 56 | 0 | 0 | 0 | 0 | 9 |
| 2 |  | 55 | 56 | 0 | 0 | 0 | 2 | 5 |
| 1 | SF | 8 | 50 | 0 | 0 | 0 | 0 | 50 |
| 2 |  | 8 | 50 | 0 | 0 | 0 | 0 | 50 |

Table 106. Comparison of HAI and histology results - percent occurrence of histological conditions in necropsy categories.
PC=Peamouth chub; MW=Mountain Whitefish; SF=Starry Flounder

| GILLS |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| $\begin{aligned} & \stackrel{e}{\ddot{0}} \\ & \stackrel{0}{0} \\ & 0 \end{aligned}$ |  | 2 |  |  | $\begin{array}{r} \widetilde{0} \\ \text { N } \\ \text { O} \\ \text { bin } \end{array}$ |  |  | $\begin{aligned} & \frac{\pi}{0} \\ & \stackrel{0}{0} \\ & \frac{6}{4} \end{aligned}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| MW | Normal | 770 | 64.0 | 0 | 0.5 | 0.1 | 0 | 0.3 | 0 | 0 | 0.1 | 0.3 | 6.2 | 0.3 | 0.3 | 0.1 | 0 | 23.0 | 0.9 | 0.1 | 0.1 | 0.1 | 0.8 | 47.9 | 12.1 |
|  | Frayed | 17 | 52.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.9 | 0 | 0 | 0 | 0 | 35.3 | 0 | 0 | 0 | 0 | 0 | 47.1 | 23.5 |
|  | Clubbed | 2 | 50.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 50.0 | 0 |
|  | Marginate | 1 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Other | 13 | 92.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15.4 | 0 | 0 | 0 | 0 | 0 | 15.4 | 0 |
| PC | Normal | 1309 | 36.8 | 0.8 | 0.3 | 2.3 | 0 | 0 | 6.3 | 3.9 | 0.3 | 0.2 | 9.3 | 0 | 0 | 0 | 0.1 | 59.1 | 0.2 | 0 | 0 | 0 | 0.2 | 39.9 | 23.0 |
|  | Fraved | 32 | 9.4 | 3.1 | 0 | 3.1 | 0 | 0 | 3.1 | 9.4 | 3.1 | 0 | 3.1 | 0 | 0 | 0 | 0 | 81.3 | 0 | 0 | 0 | 0 | 0 | 34.4 | 28.1 |
|  | Clubbed | 5 | 20.0 | 0 | 0 | 0 | 0 | 0 | 20.0 | 20.0 | 0 | 0 | 60.0 | 0 | 0 | 0 | 0 | 80.0 | 0 | 0 | 0 | 0 | 0 | 60.0 | 20.0 |
|  | Marginate | 9 | 11.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11.1 | 0 | 0 | 0 | 0 | 44.4 | 0 | 0 | 0 | 0 | 0 | 77.8 | 44.4 |
|  | Pale | 4 | 50.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 25.0 | 0 | 0 | 0 | 0 | 75.0 | 0 | 0 | 0 | 0 | 0 | 25.0 | 50.0 |
|  | Other | 9 | 11.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 66.7 | 0 | 0 | 0 | 0 | 0 | 55.6 | 33.3 |
| SF | Normal | 234 | 67.9 | 0.4 | 0 | 0.4 | 0.9 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | 26.5 | 2.6 | 0 | 0.4 | 4.7 | 0.9 | 16.2 | 7.3 |
|  | Pale | 2 | 50.0 | 0 | 0 | 0 | 50.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 50.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |



Table 106. Comparison of HAI and histology results - percent occurrence of histological conditions in necropsy categories. (continued)


Table 106. Comparison of HAI and histology results - percent occurrence of histological conditions in necropsy categories. (continued)

| KIDNEY |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  | $\geq$ |  |  | 를 <br> 0 <br> 0 <br> 0 <br> 0 <br> 0.0 <br> 0.0 <br>  |  |  | $\stackrel{\pi}{\infty}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| MW | Normal | 679 | 90.6 | 0 | 0.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0 | 0 | 2.8 | 7.8 | 0 | 0 | 0 | 0 | 0 | 2.1 |
|  | Swollen | 4 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Mottled | 26 | 84.6 | 0 | 0 | 0 | 0 | 0 | 3.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11.5 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Granular | 13 | 61.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7.7 | 30.8 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Urolithic | 37 | 78.4 | 0 | 0 | 0 | 2.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.4 | 16.2 | 0 | 0 | 0 | 0 | 0 | 5.4 |
|  | Asymetrical | 1 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Black | 2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Brown | 10 | 80.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 20.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Parasites | 35 | 68.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.7 | 25.7 | 0 | 0 | 0 | 0 | 0 | 0 |
| PC | Normal | 11 | 90.9 | 0 | 100 | 0 | 9.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Pink | 1082 | 66.2 | 0 | 95.0 | 0.1 | 4.7 | 0.2 | 0 | 0.1 | 0.3 | 0.2 | 0.4 | 0 | 0.1 | 0.2 | 0.1 | 0.1 | 0 | 0 | 0.5 | 0.1 | 0 | 0.6 |
|  | Swollen | 1 | 0 | 0 | 100 | 0 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Mottled | 49 | 44.9 | 0 | 95.9 | 0 | 6.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Granular | 10 | 10.0 | 0 | 70.0 | 0 | 20.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10.0 | 0 | 0 | 0 |
|  | Urolithic | 1 | 0 | 0 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Black/Grey | 44 | 72.7 | 0 | 97.7 | 0 | 6.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4.5 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Brown | 49 | 63.3 | 0 | 91.8 | 0 | 2.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4.1 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | Missing | 126 | 70.6 | 0 | 96.0 | 0 | 9.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| SF | Normal | 234 | 95.7 | 0.4 | 0 | 3 | 0 | 0.4 | 0 | 0 | 0.4 | 0.4 | 0.4 | 0 | 0 | 0 | 0 | 0 | 0 | 48.3 | 0 | 0 | 0 | 6.0 |
|  | Mottled | 2 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100 | 0 | 0 | 0 | 0 |
|  | Granular | 1 | 100 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 106. Comparison of HAI and histology results - percent occurrence of histological conditions in necropsy categories. (continued)

HINDGUT

|  | $\begin{aligned} & \stackrel{\rightharpoonup}{0} \\ & \stackrel{0}{0} \\ & \stackrel{0}{2} \\ & \hline \end{aligned}$ | 2 | $\begin{aligned} & \overline{0} \\ & \text { E흘 } \\ & \hline \end{aligned}$ | $\begin{aligned} & \text { 를 } \\ & 0 . \\ & 0 \\ & 0 \\ & \frac{0}{0} \\ & 0 \\ & 0 \\ & \hline 1 \\ & \hline \end{aligned}$ |  |  |  |  | $\begin{gathered} \stackrel{\rightharpoonup}{w} \\ \vdots \\ \hline \end{gathered}$ |  |  |  | © <br> 들 <br> 듕 <br> © | $\stackrel{\text { ¢ }}{\text { ¢ }}$ |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| MW | 0 | 680 | 98.5 | 0 | 0 | 0 | 0.7 | 0 | 0.3 | 0.4 | 0.9 | 0.3 | 0 | 0.1 | 0.3 | 0 |
|  |  | 109 | 99.1 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0.9 | 0.9 | 0 | 0 | 0 | 0 |
|  | 2 | 19 | 94.7 | 0 | 0 | 0 |  | 0 | 0 | 0 | 5.3 | 0 | 5.3 | 0 | 0 |  |
| PC | 0 | 984 | 98.3 | 2.9 | 0.1 | 0 | 4.0 | 0.1 | 0 | 0.2 | 0.6 | 0 | 0 | 0 | 0.1 | 0.4 |
|  |  | 305 | 97.4 | 4.6 | 0 | 0 | 3.3 | 0 | 0 | 0.7 | 0.3 | 0.3 | 0 | 0 | 0 | 0 |
|  | 2 | 65 | 93.8 | 1.5 | 0 | 0 | 4.6 | 0 | 0 | 1.5 | 4.6 | 0 | 1.5 | 0 | 0 |  |
| SF | 0 | 190 | 96.3 | 0 | 0 | 0 | 3.7 | 0 | 0 | 0.5 | 2.6 | 0 | 2.6 | 0 | 0 | 0 |
|  |  | 44 | 95.5 | 0 | 0 | 2.3 | 4.5 | 0 | 0 | 0 | 0 | 0 | 2.3 | 0 | 0 | 0 |

Table 107. Gill histology for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF) percent occurrence of abnormalities.

| $\begin{aligned} & \mathscr{0} \\ & \stackrel{0}{0} \\ & \text { © } \\ & \hline \end{aligned}$ | $\begin{aligned} & \text { む̀ } \\ & \hline \end{aligned}$ | Reach | $N$ |  |  |  | $\begin{aligned} & \text { E } \\ & \text { E } \\ & \stackrel{0}{6} \\ & 0 . \\ & \stackrel{E}{0} \\ & 0 \\ & \hline 0 \end{aligned}$ |  |  |  | Microsporidian parasite |  |  |  |  |  | $\begin{aligned} & \frac{\infty}{\omega} \\ & \frac{0}{0} \\ & \underline{\mathbb{Q}} \\ & \hline \end{aligned}$ | $\begin{aligned} & \text { 足 } \\ & \text { O } \\ & \text { 틀 } \\ & \frac{0}{1} \end{aligned}$ |  |  |  |  |  |  | $\begin{aligned} & \frac{\varepsilon}{5} \\ & \frac{0}{3} \\ & \stackrel{\rightharpoonup}{4} \\ & \hline \end{aligned}$ |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1994 | Nechako | 62 | 21.0 | 0 | 0 | 4.8 | 0 | 0 | 4.8 | 8.1 | 0 | 0 | 6.5 | 0 | 0 | 0 | 0 | 66.1 | 0 | 0 | 0 | 0 | 0 | 40.3 | 8.1 |
|  |  | Hansard | 58 | 15.5 | 0 | 0 | 0 | 0 | 0 | 6.9 | 10.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 70.7 | 0 | 0 | 0 | 0 | 0 | 41.4 | 10.3 |
|  |  | Woodpecker | 60 | 28.3 | 0 | 0 | 0 | 0 | 0 | 6.7 | 18.3 | 0 | 0 | 6.7 | 0 | 0 | 0 | 0 | 65.0 | 0 | 0 | 0 | 0 | 0 | 20.0 | 15.0 |
|  |  | Marguerite | 56 | 12.5 | 0 | 0 | , | 0 | 0 | 6.9 | 25.9 | 0 | 0 | 17.2 | 0 | 0 | 0 | 0 | 55.2 | 0 | 0 | 0 | 0 | 0 | 66.1 | 1.7 |
|  |  | Agassiz | 55 | 32.7 | 0 |  | 5.5 | 0 | 0 | 5.5 | 7.3 | 0 | 0 | 20.0 | 0 | 0 | 0 | 0 | 40.0 | 0 | 0 | 0 | 0 | 0 | 25.5 | 9.1 |
|  |  | Mission | 57 | 7.0 | 0 | , | 3.5 | 0 | 0 | 3.5 | 0 | 0 | 1.8 | 10.5 | 0 | 0 | 0 | 0 | 31.6 | 0 | 0 | 0 | 0 | , | 84.2 | 47.4 |
|  |  | Barnston | 57 | 17.5 | 0 | 0 | 0 | 0 | 0 | 8.8 | 0 | 0 | 0 | 22.8 | 0 | 0 | 0 | 0 | 38.6 | 1.8 | 0 | 0 | 0 | 1.8 | 36.8 | 28.1 |
|  |  | MainArm | 56 | 35.7 | 0 | 1.8 | 1.8 | 0 | 0 | 3.6 | 0 | 0 | 0 | 21.4 | 0 | 0 | 0 | 0 | 25.0 | 0 | 0 | 0 | 0 | 0 | 32.1 | 41.1 |
|  |  | NorthArm | 59 | 16.9 | 0 | 0 | 1.7 | 0 | 0 | 3.4 | 1.7 | 0 | 0 | 32.2 | 0 | 0 | 0 | 0 | 52.5 | 3.4 | 0 | 0 | 0 | 1.7 | 44.1 | 3.4 |
|  |  | Thompson | 58 | 32.8 | 0 | 0 | 1.7 | 0 | 0 | 8.6 | 3.4 | 0 | 0 | 6.9 | 0 | 0 | 0 | 0 | 50.0 | 0 | 0 | 0 | 0 | 0 | 37.9 | 22.4 |
|  |  | N.Thompson | 58 | 24.1 | 0 | 0 | 8.6 | 0 | 0 | 17.2 | 17.2 | 0 | 0 | 24.1 | 0 | 0 | 0 | 1.7 | 72.4 | 0 | 0 | 0 | 0 | 0 | 22.4 | 10.3 |
|  | 1995 | Nechako | 60 | 40.0 | 0 | 0 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 93.3 | 0 | 0 | 0 | 0 | 0 | 40.0 | 21.7 |
|  |  | Hansard | 54 | 0 | 0 | 1.9 | 3.7 | 0 | 0 | 7.4 | 0 | 9.3 | 0 | 0 | 0 | 0 | 0 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 87.0 | 0 |
|  |  | Woodpecker | 59 | 71.2 | 0 | 0 | 3.4 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 5.1 | 0 | 0 | 0 | 0 | 33.9 | 0 | 0 | 0 | 0 | 0 | 45.8 | 59.3 |
|  |  | Marguerite | 30 | 60.0 | 0 | 0 | 0 | 0 | 0 | 13.3 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 0 | 66.7 | 0 | 0 | 0 | 0 | 0 | 23.3 | 23.3 |
|  |  | Agassiz | 59 | 61.0 | 1.7 | 1.7 | 0 | 0 | 0 | 3.4 | 0 | 0 | 1.7 | 6.8 | 0 | 0 | 0 | 0 | 16.9 | 0 | 0 | 0 | 0 | 0 | 64.4 | 69.5 |
|  |  | MainArm | 59 | 33.9 | 1.7 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 78.0 | 0 | 0 | 0 | 0 | 0 | 54.2 | 49.2 |
|  |  | NorthArm | 60 | 45.0 | 3.3 | 0 | 0 | 0 | 0 | 8.3 | 0 | 0 | 0 | 13.3 | 0 | 0 | 0 | 0 | 80.0 | 0 | 0 | 0 | 0 | 0 | 75.0 | 35.0 |
|  |  | Thompson | 60 | 96.7 | 5.0 | 1.7 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 8.3 | 0 | 0 | 0 | 0 | 11.7 | 0 | 0 | 0 | 0 | 0 | 50.0 | 5.0 |
|  |  | N.Thompson | 60 | 95.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.0 | 0 | 0 | 0 | 0 | 15.0 | 0 | 0 | 0 | 0 | 0 | 50.0 | 13.3 |
|  | 1996 | Hansard | 60 | 0 | 8.3 | 0 | 6.7 | 0 | 0 | 10.0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 6.7 |
|  |  | Woodpecker | 55 | 49.1 | 0 | 0 | 12.7 | 0 | 0 | 5.5 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 87.3 | 0 | 0 | 0 | 0 | 0 | 1.8 | 20.0 |
|  |  | MainArm | 60 | 26.7 | 0 | 0 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 5.0 | 0 | 0 | 0 | 0 | 93.3 | 0 | 0 | 0 | 0 | 0 | 6.7 | 41.7 |
|  |  | MainArmSep | 53 | 43.4 | 0 | 0 |  | 0 | 0 | 5.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 98.1 | 0 | 0 | 0 | 0 | 0 | 0 | 18.9 |
| MW | 1994 | Nechako | 60 | 25.0 | 0 | 1.7 | , | 0 | 0 | 0 | 0 | 0 | 0 | 8.3 | 0 | 0 | 0 | 0 | 55.0 | 0 | 0 | 0 | 0 | 3.3 | 38.3 | 1.7 |
|  |  | Hansard | 59 | 13.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10.2 | 0 | 0 | 1.7 | 0 | 71.2 | 10.2 | 1.7 | 0 | 0 | 0 | 62.7 | 5.1 |
|  |  | Woodpecker | 63 | 30.2 | 0 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0 | 1.6 | 0 | 0 | 0 | 0 | 44.4 | 1.6 | 0 | 0 | 0 | 3.2 | 38.1 | 1.6 |
|  |  | Marguerite | 29 | 13.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 13.3 | 0 | 0 | 0 | 0 | 43.3 | 0 | 0 | 0 | 0 | 0 | 72.4 | 0 |
|  |  | Agassiz | 19 | 31.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 31.6 | 0 | 0 | 0 | 0 | 52.6 | 0 | 0 | 0 | 0 | 5.3 | 21.1 | 0 |
|  |  | Thompson | 69 | 13.0 | 0 | 0 | 0 | 0 | 1.4 | 0 | 0 | 0 | 0 | 7.2 | 0 | 0 | 0 | 0 | 21.7 | 0 | 0 | 0 | 0 | 0 | 82.6 | 15.9 |
|  |  | N.Thompson | 57 | 45.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 12.3 | 0 | 0 | 0 | 0 | 35.1 | 0 | 0 | 1.8 | 1.8 | 1.8 | 33.3 | 10.5 |
|  | 1995 | Nechako | 60 | 95.0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 3.3 | 1.7 | 3.3 | 3.3 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 63.3 | 0 |
|  |  | Hansard | 60 | 90.0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 10.0 | 0 | 0 | 0 | 0 | 0 | 58.3 | 1.7 |
|  |  | Woodpecker | 60 | 98.3 | 0 | 0 |  | 0 | 0 | 0 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 43.3 | 51.7 |
|  |  | Marguerite | 14 | 85.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 35.7 | 28.6 |
|  |  | Agassiz | 45 | 97.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 46.7 | 46.7 |
|  |  | Thompson | 60 | 96.7 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 53.3 | 15.0 |
|  |  | N.Thompson | 60 | 96.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 1.7 | 0 | 0 | 0 | 0 | 5.0 | 0 | 0 | 0 | 0 | 0 | 61.7 | 5.0 |
|  | 1996 | Hansard | 52 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9.6 | 0 | 0 | 0 | 0 | 0 | 0 | 5.8 |
|  |  | Woodpecker | 36 | 97.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 16.7 | 0 | 0 | 0 | 0 | - | 2.8 | 8.3 |
| SF | 1994 | MainArm | 61 | 60.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4.9 | 0 | 0 | 0 | 0 | 14.8 | 1.6 | 0 | 0 | 1.6 | 0 | 19.7 | 6.6 |
|  |  | NorthArm | 54 | 20.4 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 70.4 | 9.3 | 0 | 0 | 18.5 | 3.7 | 33.3 | 5.6 |
|  | 1995 | MainArm | 64 | 93.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4.7 | 0 | 0 | 0 | 0 | 14.1 | 0 | 0 | 0 | 0 | 0 | 7.8 | 9.4 |
|  |  | NorthArm | 57 | 91.2 | 1.8 | 0 | 0 | 5.3 | 0 | 0 | 0 | 0 | 0 | 3.5 | 0 | 0 | 0 | 0 | 12.3 | 0 | 0 | 1.8 | 0 | 0 | 5.3 | 7.0 |

Table 108. Liver histology for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF) - \% occurrence of abnormalities

|  |  | Reach | $N$ |  |  |  | $\begin{aligned} & \text { n } \\ & 0 \\ & 0 \\ & 0 \\ & 0 \\ & 00 \\ & 0 . \\ & 0 \\ & 0 \end{aligned}$ |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  | Scattered pyknotic cells |  |  |  |  |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1994 | Nechako | 62 | 14.5 | 0 | 0 | 0 | 61.3 | 0 | 58.1 | 12.9 | 0 | 0 | 1.6 | 3.2 | 6.5 | 8.1 | 0 | 0 | 0 | 0 | 1.6 | 0 | 0 | 16.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 59 | 13.6 | 0 | 0 | 1.7 | 23.7 | 0 | 33.9 | 15.3 | 0 | 1.7 | 0 | 3.4 | 13.6 | 18.6 | 0 | 0 | 3.4 | 1.7 | 0 | 0 | 1.7 | 5.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 59 | 39.0 | 0 | 0 | 1.7 | 13.6 | 0 | 22.0 | 0 | 0 | 10.2 | 1.7 | 3.4 | 10.2 | 6.8 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 59 | 74.6 | 0 | 0 | 0 | 3.4 | 0 | 79.7 | 3.4 | 0 | 11.9 | 5.1 | 0 | 1.7 | 3.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 55 | 9.1 | 0 | 0 | , | 27.3 | 0 | 18.2 | 1.8 | 0 | 20.0 | 21.8 | 0 | 7.3 | 23.6 | 7.3 | 0 | 1.8 | 0 | 3.6 | 0 | 1.8 | 5.5 | 0 | 0 | 0 |  | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Mission | 56 | 10.7 | 0 | 0 | 0 | 16.1 | 0 | 57.1 | 5.4 | 0 | 1.8 | 3.6 | 3.6 | 5.4 | 25.0 | 3.6 | 0 | 1.8 | 0 | 12.5 | 0 | 0 | 12.5 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 1.8 | 0 | 0 | 0 |
|  |  | Barnston | 58 | 17.2 | 0 | 0 | 0 | 31.0 | 0 | 50.0 | 0 | 0 | 1.7 | 5.2 | 1.7 | 8.6 | 29.3 | 0 | 0 | 8.6 | 0 | 1.7 | 0 | 0 | 5.2 | 0 | 0 | 0 | , | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 |
|  |  | MainArm | 56 | 17.9 | 0 | 0 | 1.8 | 8.9 | 0 | 44.6 | 1.8 | 0 | 12.5 | 17.9 | 3.6 | 10.7 | 10.7 | 0 | 1.8 | 5.4 | 0 | 0 | 0 | 0 | 3.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 60 | 13.3 | 0 | 0 | 1.7 | 13.3 | 0 | 50.0 | 1.7 | 0 | 0 | 6.7 | 1.7 | 15.0 | 16.7 | 1.7 | 0 | 1.7 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 1.7 | - | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 60 | 26.7 | 0 | 0 | 0 | 1 | 0 | 86.7 | 1.7 | 0 | 0 | 0 | 11.7 | 3.3 | 10.0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 16.7 | 0 | 0 | 1.7 | 1.7 | 5.0 | 70.0 | 1.7 | 1.7 | 1.7 | 10.0 | 0 | 0 | 6.7 | 0 | 0 | 1.7 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | , | - | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 |
|  | 1995 | Nechako | 60 | 81.7 | 0 | 0 | 0 | 91.7 | 5.0 | 36.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 |
|  |  | Hansard | 54 | 94.4 | 0 | 0 | 0 | 9.3 | 7.4 | 85.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | - | 0 | 0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 59 | 98.3 | 0 | 0 | 0 | 5.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 3.4 |
|  |  | Marguerite | 30 | 93.3 | 0 | 0 | 0 | 23.3 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |  | 3.3 |
|  |  | Agassiz | 61 | 98.4 | 0 | 0 | 0 | 9.8 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 |
|  |  | MainArm | 59 | 91.5 | 0 | 1.7 | 0 | 20.3 | 5.1 | 3.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | , | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 57 | 98.2 | 0 | 0 | 1.8 | 5.3 | 1.8 | 3.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 3.5 | 0 | 0 | 0 | 0 | - | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 |
|  |  | Thompson | 60 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 100.0 | 0 | 0 | 0 | 5.0 | 3.3 | 16.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | - | 5.0 |
|  | 1996 | Hansard | 60 | 96.7 | 0 | 0 | 0 | 3.3 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 55 | 98.2 | 0 | 0 | 0 | 3.6 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 59 | 100.0 | 0 | 0 | 1.7 | 10.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | , | - | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArmSep | 53 | 100.0 | 0 | 0 | 0 | 32.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| MW | 1994 | Nechako | 60 | 83.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 1.7 | 1.7 | 0 | 3.3 | 0 | 0 | 8.3 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 |  | 0 | 0 | 0 |
|  |  | Hansard | 59 | 91.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 3.4 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 64 | 84.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 12.5 | 0 | 0 | 0 | 0 | 3.1 | 1.6 | 0 | 0 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 29 | 79.3 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10.3 | 6.9 | 0 | 0 | 6.9 | 0 | 0 | 0 | 0 | , | - | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 19 | 52.6 | 0 | 0 | 0 | 0 | 0 | 5.3 | 0 | 0 | 5.3 | 0 | 0 | 0 | 0 | 0 | 5.3 | 15.8 | 0 | 5.3 | 0 | 10.5 | 5.3 | 0 | 0 | 0 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 70 | 42.9 | 0 | 0 | 0 | 1.4 | 0 | 0 | 0 | 0 | 0 | 0 | 1.4 | 0 | 0 | 0 | 21.4 | 32.9 | 1.4 | 0 | 0 | 0 | 7.1 | 0 | 0 | 0 | 0 | 1.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 59 | 84.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 1.7 | 0 | 0 | 1.7 | 0 | 3.4 | 0 | 6.8 | 1.7 | 0 | 0 | 0 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | Nechako | 60 | 90.0 | 0 | 0 | 0 | 3.3 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | , | , | 0 | 1.7 | 0 | 0 | 0 | 0 | 5.0 |
|  |  | Hansard | 59 | 96.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | , | 5.1 |
|  |  | Woodpecker | 60 | 98.3 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.0 | 1.7 | 0 | 0 | , | - | 0 | 0 | 0 | 0 | 0 | 0 | 6.7 |
|  |  | Marquerite | 14 | 100.0 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | , | 0 | 0 | 50.0 | 0 | 0 | , | 0 | 0 | 0 | 0 | 0 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 45 | 97.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17.8 | 0 | 0 | 0 | 0 | 4.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 60 | 91.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 38.3 | 0 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 0 | , | - | 5.0 | 1.7 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | , | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 |
|  | 1996 | Hansard | 51 | 98.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.9 | 5.9 |
|  |  | Woodpecker | 37 | 100.0 | 0 | 0 | 0 | 5.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 13.5 | 0 | 0 | 0 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.7 |
| SF | 1994 | MainArm | 62 | 71.0 | 0 | 0 | 3.2 | 8.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 9.7 | 0 | 0 | 0 | 0 | 0 | 4.8 | 0 | 0 | 11.3 | 0 | 0 | 0 | 3.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 53 | 77.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.8 | 1.9 | 0 | 11.3 | 1.9 | 0 | 0 | 3.8 | 0 | 0 | 0 | 1.9 | 9.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | MainArm | 64 | 98.4 | 0 | 0 | 0 | 4.7 | , | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.1 | 0 | 0 | 1.6 | 0 | 1.6 | 0 | 0 | - | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.3 |
|  |  | NorthArm | 60 | 98.3 | 1.7 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 |

Table 109. Spleen histology for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF) percent occurrence of abnormalities.

| $\begin{aligned} & \text { © } \\ & \stackrel{0}{0} \\ & \stackrel{0}{6} \end{aligned}$ | $\begin{aligned} & \text { ฐ̀ } \\ & \hline \end{aligned}$ | Reach | $N$ |  |  |  | $\begin{array}{\|c} \stackrel{\rightharpoonup}{m} \\ \hat{0} \end{array}$ |  |  | Slight hepatocellular steatosis |  |  |  |  |  |  |  |  |  | -arge melanomacrophage centres |  |  | $\begin{aligned} & 0 \\ & \stackrel{0}{\ddot{1}} \\ & 0 \\ & 0 \\ & \hline 0 \\ & \hline \end{aligned}$ |  | $\begin{aligned} & \text { 든 } \\ & .0 \\ & \hline 13 \end{aligned}$ |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1994 | Nechako | 62 | 1.6 | 98.3 | 8.1 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |  | 0 | 0 | 1.6 | 3.2 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 59 | 0 | 98.3 | 5.1 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 3.4 | 0 | 15.3 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 58 | 0 | 94.8 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 22.4 | 0 | 10.3 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 59 | 0 | 100.0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |  | 0 | 8.5 | 25.4 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 55 | 0 | 98.2 | 7.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |  |  |  | 0 | 0 | 0 | 25.5 | 0 | 0 | 0 | 0 | 0 |
|  |  | Mission | 57 | 96.5 | 100.0 | 8.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 12.3 | 0 | 0 | 0 | 0 | 0 |
|  |  | Barnston | 57 | 86.0 | 100.0 | 7.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 19.3 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 57 | 42.1 | 100.0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 60 | 0 | 100.0 | 3.3 | 0 | 0 | 0 | 0 | 1.7 | 1.7 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 6.7 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 59 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.1 | 1.7 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 1.7 | 98.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 78.3 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | Nechako | 60 | 96.7 | 96.7 | 21.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 |
|  |  | Hansard | 54 | 88.9 | 94.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.9 | 0 | 0 | 7.4 | 0 | 0 | 0 | 0 | 0 | 0 | 1.9 | 1.9 | 0 | 0 |
|  |  | Woodpecker | 57 | 96.5 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 30 | 96.7 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 58 | 96.6 | 100.0 | 5.2 | 0 | 0 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 59 | 91.5 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 5.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 59 | 86.4 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.8 | 0 | 1.7 |
|  |  | Thompson | 60 | 100.0 | 98.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 100.0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1996 | Hansard | 60 | 100.0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 55 | 96.4 | 96.4 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 52 | 98.1 | 100.0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArmSep | 52 | 100.0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| MW | 1994 | Nechako | 59 | 42.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 28.8 | 13.6 | 0 | 20.3 | 0 | 0 | 0 | 0 | 3.4 |
|  |  | Hansard | 59 | 76.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8.5 | 8.5 | 0 | 8.5 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 62 | 50.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 14.5 | 30.6 | 0 | 6.5 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 28 | 75.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7.1 | 17.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 18 | 88.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.6 | 5.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 69 | 94.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 59 | 69.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | - | 0 | 16.9 | 15.3 | 0 | 3.4 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | Nechako | 60 | 90.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.0 | 1.7 | 0 | 13.3 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 |
|  |  | Hansard | 58 | 91.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8.6 | 0 | 0 | 6.9 | 0 | 3.4 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 60 | 98.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 14 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 45 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 60 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 96.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.7 | 0 | 0 | 1.7 | 0 | 1.7 | 0 | 0 | 0 | 0 | 6.7 |
|  | 1996 | Hansard | 50 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 4.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.0 |
|  |  | Woodpecker | 36 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| SF | 1994 | MainArm | 59 | 74.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100 | 5.1 | 20.3 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 54 | 79.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100 | 0 | 20.4 | 0 | 0 | 0 | 0 |
|  | 1995 | MainArm | 63 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 57 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |

Table 110. Kidney histology for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF) percent occurence of abnormalities.

| $\begin{aligned} & \text { e } \\ & \stackrel{0}{\dddot{0}} \\ & \dot{0} \\ & \hline \end{aligned}$ | $\begin{aligned} & \text { ※ } \\ & \end{aligned}$ | Reach | $N$ | 즌 을 |  | 른 0.2 0 0 0.0 0 0 2 2 |  |  | $\begin{gathered} \pi \\ 0 \\ \hline \end{gathered}$ |  | Inflammatory foci/focus |  |  |  |  |  |  |  |  |  |  |  | $\begin{aligned} & \infty \\ & \frac{0}{0} \\ & \frac{0}{n} \\ & \frac{1}{0} \\ & \frac{0}{0} \\ & \vdots \overline{0} \\ & \hline \end{aligned}$ | 0 0 0 0 0 0 0. 0. |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1994 | Nechako | 62 | 6.5 | 0 | 91.9 | 0 | 25.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 59 | 1.7 | 0 | 96.6 | 1.7 | 8.5 | 0 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 |
|  |  | Woodpecker | 59 | 15.3 | 0 | 81.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 59 | 0 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.1 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 55 | 21.8 | 0 | 74.5 | 0 | 20.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 |
|  |  | Mission | 57 | 98.2 | 0 | 98.2 | 0 | 7.0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | - | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Barnston | 58 | 100.0 | 0 | 100.0 | 0 | 12.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 57 | 45.6 | 0 | 96.5 | 0 | 7.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 |
|  |  | NorthArm | 60 | 10.0 | 0 | 90.0 | 0 | 8.3 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 60 | 1.7 | 0 | 98.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 35.0 | 0 | 66.7 | 0 | 3.3 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | Nechako | 58 | 94.8 | 0 | 94.8 | 0 | 19.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.9 |
|  |  | Hansard | 55 | 96.4 | 0 | 96.4 | 0 | 3.6 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 |
|  |  | Woodpecker | 59 | 100.0 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 30 | 96.7 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 |
|  |  | Agassiz | 59 | 98.3 | 0 | 100.0 | 0 | 1.7 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 |
|  |  | MainArm | 60 | 95.0 | 0 | 100.0 | 0 | 3.3 | 1.7 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 58 | 87.9 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 1.7 |
|  |  | Thompson | 60 | 100.0 | 0 | 96.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 100.0 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1996 | Hansard | 60 | 100.0 | 0 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 55 | 100.0 | 0 | 100.0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 59 | 98.3 | 0 | 100.0 | 0 | 1.7 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArmSep | 53 | 98.1 | 0 | 100.0 | 0 | 3.8 | 0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| MW | 1994 | Nechako | 59 | 49.2 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 16.9 | 33.9 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 59 | 83.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.4 | 13.6 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 64 | 53.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15.6 | 31.3 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 29 | 89.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10.3 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 19 | 89.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 10.5 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 70 | 94.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.7 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 83.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 6.7 | 10.0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | Nechako | 60 | 96.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 15.0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 57 | 98.2 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 59 | 98.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 22.0 |
|  |  | Marguerite | 14 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 45 | 97.8 | 0 | 0 | 0 | 0 | 0 | 2.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 60 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 58 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1996 | Hansard | 52 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.8 |
|  |  | Woodpecker | 37 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.7 |
| SF | 1994 | MainArm | 61 | 96.7 | 0 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 100.0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 54 | 87.0 | 0 | 0 | 11.1 | 0 | 0 | 0 | 0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 100.0 | 0 | 0 | 0 | 0 |
|  | 1995 | MainArm | 63 | 100.0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11.1 |
|  |  | NorthArm | 59 | 98.3 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 11.9 |

Table 111. Hindgut histology for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF) -percent occurrence of abnormalities.

| $\begin{aligned} & \stackrel{0}{0} \\ & \stackrel{0}{0} \\ & 0 \end{aligned}$ |  | Reach | $N$ | $\begin{aligned} & \overline{0} \\ & 0 \\ & 0 \\ & \hline 0 \end{aligned}$ |  |  |  |  |  | $\stackrel{\pi}{\omega}$ |  |  | Inflammation in pancreas | $\begin{array}{r} \mathbb{N} \\ \frac{0}{2} \\ \frac{0}{2} \\ \stackrel{\rightharpoonup}{0} \\ \hline \end{array}$ | $\begin{aligned} & \grave{0} \\ & \frac{0}{0} \end{aligned}$ |  |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1994 | Nechako | 62 | 98.4 | 0 | 0 | 0 | 21.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 58 | 96.6 | 3.4 | 0 | 0 | 3.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 58 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 58 | 98.3 | 0 | 0 | 0 | 5.2 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 54 | 94.4 | 1.9 | 0 | 0 | 3.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Mission | 57 | 98.2 | 0 | 0 | 0 | 5.3 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Barnston | 57 | 100.0 | 1.8 | 0 | 0 | 8.8 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 54 | 98.1 | 1.9 | 0 | 0 | 0 | 0 | 0 | 1.9 | 3.7 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 60 | 95.0 | 1.7 | 0 | 0 | 3.3 | 0 | 0 | 1.7 | 1.7 | 0 | 1.7 | 0 | - | 0 |
|  |  | Thompson | 57 | 94.7 | 1.8 | 0 | 0 | 3.5 | 1.7 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 93.3 | 6.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | Nechako | 59 | 94.9 | 25.4 | 0 | 0 | 23.7 | 0 | 0 | 0 | 5.1 | 0 | 0 | 0 | 1.7 | 0 |
|  |  | Hansard | 54 | 96.3 | 13.0 | 0 | 0 | 5.6 | 0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 1.9 |
|  |  | Woodpecker | 58 | 98.3 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 30 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 59 | 98.3 | 6.8 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 1.7 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 59 | 100.0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 59 | 96.6 | 5.1 | 0 | 0 | 1.7 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 1.7 |
|  |  | Thompson | 60 | 100.0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 59 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1996 | Hansard | 60 | 100.0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 |
|  |  | Woodpecker | 55 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 57 | 98.2 | 0 | 0 | 0 | 3.5 | 0 | 0 | 0 | 1.8 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArmSep | 52 | 100.0 | 1.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| MW | 1994 | Nechako | 59 | 98.3 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 59 | 98.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 0 |
|  |  | Woodpecker | 64 | 98.4 | 0 | 0 | 0 | 3.1 | 0 | 0 | 1.6 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 29 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 19 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 70 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | N.Thompson | 60 | 95.0 | 0 | 0 | 0 | 1.7 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | Nechako | 60 | 95.0 | 0 | 0 | 0 | 5.0 | 0 | 0 | 0 | 5.0 | 3.3 | 0 | 0 | 1.7 | 0 |
|  |  | Hansard | 60 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 60 | 96.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 14 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 45 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 60 | 98.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5.0 | 0 | 0 | 1.7 | 0 | 0 |
|  |  | N.Thompson | 60 | 100.0 | 0 | 0 | 0 | 0 | 0 | 3.3 | 0 | 0 | 1.7 | 0 | 0 | 1.7 | 0 |
|  | 1996 | Hansard | 52 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Woodpecker | 37 | 100.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| SF | 1994 | MainArm | 61 | 98.4 | 0 | 0 | 0 | 3.3 | 0 | 0 | 0 | 0 | 0 | 1.6 | 0 | 0 | 0 |
|  |  | NorthArm | 53 | 98.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.9 | 0 | 0 | 0 | 0 | 0 |
|  | 1995 | MainArm | 64 | 98.4 | 0 | 0 | 0 | 4.7 | 0 | 0 | 0 | 3.1 | 0 | 0 | 0 | 0 | 0 |
|  |  | NorthArm | 59 | 89.8 | 0 | 0 | 1.7 | 6.8 | 0 | 0 | 1.7 | 3.4 | 0 | 8.5 | 0 | 0 | 0 |

Table 112. Gonad histology for peamouth chub (PC), mountain whitefish (MW), and starry flounder (SF) percent occurrence of abnormalities.

| $\begin{aligned} & \text { © } \\ & \text { © } \\ & \text { © } \end{aligned}$ | $\begin{aligned} & \text { ※ } \\ & \hline \end{aligned}$ | Reach | $N$ |  |  |  | $\begin{aligned} & \frac{\infty}{\infty} \\ & \text { D } \\ & \text { in } \\ & \hline \end{aligned}$ | 든 든 옫 |  |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| PC | 1995 | Nechako | 44 | 63.6 | 0 | 0 | 0 | 6.8 | 0 |
|  |  | Hansard | 49 | 36.7 | 0 | 0 | 0 | 0 | 2.0 |
|  |  | Woodpecker | 58 | 67.2 | 0 | 0 | 0 | 0 | 19.0 |
|  |  | Marguerite | 30 | 66.7 | 0 | 0 | 0 | 0 | 30.0 |
|  |  | Agassiz | 48 | 45.8 | 0 | 0 | 0 | 0 | 0 |
|  |  | MainArm | 60 | 45.0 | 0 | 0 | 0 | 0 | 26.7 |
|  |  | NorthArm | 52 | 86.5 | 0 | 0 | 0 | 0 | 13.5 |
|  |  | Thompson | 52 | 46.2 | 0 | 0 | 0 | 0 | 30.8 |
|  |  | N.Thompson | 49 | 63.3 | 0 | 0 | 0 | 0 | 14.3 |
|  | 1996 | Hansard | 34 | 17.6 | 0 | 0 | 0 | 0 | 5.9 |
|  |  | Woodpecker | 29 | 69.0 | 0 | 0 | 0 | 0 | 24.1 |
|  |  | MainArm | 27 | 14.8 | 0 | 0 | 0 | 0 | 0 |
| MW | 1995 | Nechako | 37 | 16.2 | 0 | 0 | 0 | 0 | 0 |
|  |  | Hansard | 42 | 40.5 | 0 | 2.4 | 0 | 0 | 2.4 |
|  |  | Woodpecker | 45 | 20.0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Marguerite | 11 | 63.6 | 0 | 0 | 0 | 0 | 0 |
|  |  | Agassiz | 25 | 48.0 | 0 | 0 | 0 | 0 | 0 |
|  |  | Thompson | 50 | 54.0 | 2.0 | 0 | 0 | 0 | 2.0 |
|  |  | N.Thompson | 35 | 37.1 | 0 | 0 | 0 | 0 | 0 |
|  | 1996 | Hansard | 18 | 0 | 0 | 0 | 0 | 0 | 5.6 |
|  |  | Woodpecker | 14 | 0 | 0 | 0 | 0 | 0 | 0 |
| SF | 1995 | MainArm | 56 | 51.8 | 0 | 0 | 0 | 1.8 | 46.4 |
|  |  | NorthArm | 50 | 62.0 | 0 | 0 | 0 | 0 | 38.0 |

Figures


Figure 1. The Fraser basin study area.





| $\ldots \ldots .$. | Period of Record |
| :--- | :--- |
| $-\quad 1994$ |  |
| $-\cdots$ | 1995 |
| $-\cdots$ | 1996 |

Figure 2. Daily discharges averaged by month over the periods of record and for 1994, 1995, and 1996, for hydrometric stations on the Fraser River at Hansard and Hope, the Nechako River at Prince George, and the Thompson River at Spences Bridge. Periods of record are 1952 to 1996 at Hansard, 1912 to 1996 at Hope, and 1951 to 1996 at Spences Bridge.


Figure 3. Summer water temperature data from continuous monitoring sites in the Fraser basin (Barnes and Walthers 1997, Brown et al. 1998, Lauzier et al. 1995).


Figure 4. Water temperature regimes at Federal-Provincial water quality monitoring sites on the Fraser, Nechako, and Thompson rivers in 1994; diamonds represent water temperatures measured by this study during fish sampling.


Figure 5. CPUEs, by reach and year, for the six most common species of fish caught by beach seine from 1994 to 1996 (only the Hansard, Woodpecker, and Main Arm reaches were sampled in 1996).


Figure 6. General plan for presentation of tissue contaminants summaries for 1994 and 1995 analyses from the three Fraser River fish species.


Polychlorinated dibenzo-p-dioxin


Polychlorinated dibenzofuran
Figure 7. Basic structure of the polychlorodibenzodioxin and polychlorodibenzofuran molecules.


Figure 8. Chlorinated dioxin and furan concentrations in fish muscle composites from the Fraser basin. Non-detects were assigned a zero value for both display and for TEQ calculations. Error bars indicate standard errors. n/s = Not Sampled


Figure 9. Chlorinated dioxin and furan concentrations in fish liver composites from the Fraser basin. Non-detects were assigned a zero value for both display and for TEQ calculations. Error bars indicate standard error; $\mathrm{n} / \mathrm{s}=$ reach not sampled.


Figure 10. Lipid-normalized total tetrachlorodioxin concentrations in peamouth chub, whitefish and starry flounder tissues from the Fraser basin. Peamouth were collected throughout the basin, whitefish from Hansard to Marguerite and in the Thompson drainage. Starry flounder were collected only in the Main and North Arms. ND= not detected; NS=not sampled.


Figure 11. Lipid-normalized total tetrachlorofuran concentrations in peamouth chub, whitefish and starry flounder tissues from the Fraser basin, 1994. Peamouth chub were collected throughout the basin, whitefish from Hansard to Marguerite and in the Thompson drainage. Starry flounder were collected only in the Fraser River Main and North Arms. ND= not detected; NS=not sampled.


Figure 12. Lipid-normalized octachlorodibenzodioxin concentrations in peamouth chub, whitefish and starry flounder tissues from the Fraser basin, 1994. Peamouth chub were collected throughout the basin, whitefish from Hansard to Mission and in the Thompson drainage. Starry flounder were collected only in the Main and North Axm.


Figure 13. PCA ordination of the dioxin/furan profiles from fish muscle tissues, showing a clear separation of collections in the Thompson drainage (in bold) from reaches elsewhere in the basin.


Figure 14. Historical measurements of dioxin and furan concentrations in mountain whitefish muscle from the Fraser basin. Two sets of data from 1994 are Mellor (1995) - "1994(1)", and this study- "1994(2)". (data for plots from Dwernychuk 1992?)


Figure 15. Representative chlorophenolics.


Figure 16.Chlorophenolic congener profiles in peamouth, whitefish and starry flounder muscle tissue from reaches in the Fraser basin, 1994. Bars show mean concentrations per reach with standard error.


Figure 17. Chlorophenolic congener profiles in peamouth, whitefish and starry flounder liver tissue from reaches in the Fraser basin, 1994.


Figure 18. Chlorophenolic congener profiles in peamouth and whitefish bile from reaches in the Fraser basin, 1994.


Figure 19. Chlorophenolic profiles for measurements in bile from Fraser River fish, 1995. Note that all three species were not collected in all reaches; concentrations of congeners in $\mathrm{ng} / \mathrm{mL}$.


Figure 20. Representative naturally-occurring resin acids.


Figure 21. Total resin acids in bile from fish in the Fraser basin, 1994 and 1995. Samples were composites of up to 30 individuals. Reaches where no samples were obtained are marked "NS".


Figure 22. Basic PCB molecule.


Figure 23. Detection frequencies of individual PCB congeners over all tissue analyses in both 1994 and 1995, compared with congener composition in commercial Arochlor formulations. Compositional data from Schulz (1988).


Figure 24. Total PCBs in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow foreground bars represent the 1995 data for the same species and reach. Non-detected congeners were set at $1 / 2$ the sample detection limit for calculating totals.


Figure 25. Total PCBs in fish tissues from the Fraser basin, 1994 and 1995 normalized to tissue lipid content. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow foreground bars represent the 1995 data for the same species and reach. Non-detected congeners were set at $1 / 2$ the sample detection limit for calculating totals.


Figure 26. Total numbers of congeners detected in individual peamouth chub and mountain whitefish tissue analyses in the Fraser basin, 1994 and 1995. Open triangles represent liver tissues, dark triangles are muscle tissue analyses.


Figure 27. PCB Congeners in peamouth chub muscle, 1994 (dark bars) and 1995 (light bars). Shown are means of 4-5 composite samples with standard error bars.


Figure 28. PCB Congeners in starry flounder and whitefish muscle, 1994 (dark bars) and 1995 (light bars). Shown are means of 4-5 composite samples with standard error bars.


Figure 29. PCB Congeners in peamouth chub liver 1994 (dark bars) and 1995 (light bars). Shown are means of $1-3$ composite samples of up to 30 individual fish.


Figure 30. PCB Congeners in starry flounder and whitefish liver, 1994 (dark bars) and 1995 (light bars). Shown are means of 1-3 composite samples ( up to 30 individual fish.


Figure 31. Profiles of non-ortho PCB congeners measured in effluent from the NorthWood Pulp and Paper discharge in October 1994 and measured profiles in peamouth chub collected both upstream (Hansard) and downstream (Woodpecker and Marguerite) September/October 1994.


Figure 32. Concentrations of coplanar PCBs measured in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow foreground bars represent the 1995 data for the same species and reach.


Figure 33. Comparison of measured coplanar PCBs and non-ortho PCBs in fish muscle tissues in the Fraser basin, 1994 and 1995.


Figure 34. TCDD toxic equivalents in fish muscle tissues due to PCB congeners, based on TEFs (van den Berg et al. 1998) for mammals, fish and bird species. Resultant TEQs for each taxon are shown in sequence for each reach/species pair.


Figure 35. TCDD toxic equivalents in fish liver due to PCB congeners, based on TEFs (van den Berg et al. 1998) for mammals, fish and bird species. Resultant TEQs for each taxon are shown in sequence for each reach/species pair.


Figure 36. Combined WHO 2,3,7,8-TCDD TEQs from dioxin/furan and PCB sources, from fish in the Fraser basin in 1994. Letters refer to species sampled; PC - peamouth chub, MW - whitefish and SF - starry founder. The plots show clearly that the contribution due to PCBs alone may exceed the dioxin/furan component of the total TEQs in many areas of the basin.


Figure 37. Detection frequencies of target organochlorine pesticides measured in fish tissues from the Fraser basin in 1994 and 1995. Asterisks indicate a significant difference between years by $\chi^{2}$.


Figure 38. Comparison of detections in fish muscle and fish liver tissues for peamouth, whitefish and starry flounder from the Fraser basin collected in 1994 and 1995. As is evident, detections overall were extremely similar between years, with only a few exceptions.


Figure 39. Comparison of average within-reach concentrations of all target analytes measured in 1994 and 1995, showing both the high degree of similarity between years and the generally low measured levels. The dashed line indicates equal concentrations between years.



Figure 40. Structures of selected cyclodiene insecticides


Figure 41. Patterns in tissue concentration of dieldrin in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of $4-5$ composites of 5 fish, while livers are means of 1-3 composites of up to 30 fish.

Sulphate

Figure 42. Structure of endosulphan sulphate.


Figure 43. Patterns in tissue concentration of Endosulphan sulphate in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of $1-3$ composites of up to 30 fish.


Figure 44. Patterns in tissue concentration of alpha-Endosuphan (Endosuphan I) in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of $1-3$ composites of up to 30 fish.


$p, p^{\prime}-D D D$

$p, p^{\prime}-$ DDE

Figure 45. DDT and principal metabolites.


Figure 46. Patterns in tissue concentration of the parent p,p-DDT in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of 1-3 composites of up to 30 fish.


Figure 47. Patterns in tissue concentration of DDD in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of $4-5$ composites of 5 fish, while livers are means of 1-3 composites of up to 30 fish.


Figure 48. Patterns in tissue concentration of $p, p^{\prime}$ 'DDE in fish from the Fraser basin, 1994 and 1995.
Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of $1-3$ composites of up to 30 fish.


Figure 49. Representative stereoisomers of hexachlorocyclohexane


Figure 50. Patterns in tissue concentration of gamma - hexachlorocyclohexane (lindane) in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of $1-3$ composites of up to 30 fish.


Figure 51. Patterns in tissue concentration of alpha-HCH in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of $1-3$ composites of up to 30 fish.


Figure 52. Patterns in tissue concentration of beta-HCH in fish from the Fraser basin, 1994 and 1995.
Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of 1-3 composites of up to 30 fish.


Hexachlorobenzene
Figure 53. Structure of HCB.


Figure 54. Patterns in tissue concentration of hexachlorobenzene in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of 1-3 composites of up to 30 fish.

cis-Chlordane H

trans-Chlordane Cl



Figure 55. Structures of representative chlordane components.


Figure 56. Patterns in tissue concentration of trans-nonachlor in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of 1-3 composites of up to 30 fish.


Figure 57. Patterns in tissue concentration of cis-chlordane in fish from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of 1-3 composites of up to 30 fish.


Figure 58. Basic chlorobornane component in toxaphene.


Figure 59. Patterns in tissue concentration of toxaphene in fish from the Fraser basin, 1994 and 1995.
Background bar colours correspond to 1994 data for each species, while foreground narrow bars represent 1995 data for the same species and reach. Muscle values represent means and standard error of 4-5 composites of 5 fish, while livers are means of $1-3$ composites of up to 30 fish.


Phenanthrene


Benzo (a) pyrene

Figure 60. Representative 2,3 and 5-ring PAHs


Figure 61. Total low and high molecular weight PAH concentrations in peamouth liver and muscle tissue from the reaches in the lower Fraser River, 1994.


Figure 62. Total low and high molecular weight PAH concentrations in starry flounder liver and muscle tissue from reaches in the lower Fraser River, 1994.


Figure 63. Equivalent concentrations of PAHs in bile in peamouth chub and mountain whitefish from sites in the Fraser basin, 1994. Equivalents are the concentrations of compounds containing 2, 3 and 6 aromatic rings representing napthalene, phenanthrene and benzo(a)pyrene homologs, respectively, without reference to lateral substitution patterns.




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Figure 64. PAH groups in different peamouth chub matrices in the lower Fraser River, 1994. Concentrations in bile are expressed as PAH equivalents. Concentrations in liver and muscle are summed concentrations of 2 -ring (napthalenes), 3 -ring (phenanthrenes) and $4-5$ ring ( $\mathrm{B}[\mathrm{a}] \mathrm{P}$ ) parent compounds measured in tissues.


Figure 65. Total cadmium in fish liver tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach


Figure 66. Total arsenic in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach.


Figure 67. Total mercury in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach. $\mathrm{NS}=$ no sample; $\mathrm{ND}=$ not detected.


Figure 68. Total zinc in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach. NS=no sample


Figure 69. Total selenium in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach. As indicated, sufficient liver tissue was not available for analysis at a number of reaches in both 1994 and 1995. NS= no sample collected in this reach.


Figure 70. Total iron in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach.


Figure 71. Total manganese in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach.


Figure 72. Total nickel in fish tissues from the Fraser basin, 1994 and 1995. Background bar colours correspond to 1994 data for each species sampled within a reach, while narrow white foreground bars represent the 1995 data for the same species and reach.


Figure 73. EROD activity in mountain whitefish, peamouth chub, and starry flounder 1994 to 1996.


Figure 74. Discharge at six reaches in the Fraser and Thompson rivers during FRAP fish sampling 1994-1996.


Figure 75. EROD versus ECOD activities in peamouth chub, mountain whitefish, and starry flounder in the Fraser basin in 1994.


Figure 76. EROD activity versus the gonadosomatic index in mountain whitefish females.


Figure 77. Mean EROD activity in liver versus mean contaminant levels in liver 1994 to 1995.


Figure 78. Box plots of age, length, and weight by reach for peamouth chub 1994 -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 79. Box plots of age, length, and weight by reach for peamouth chub 1995 -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 80. Box plots of age, length, and weight by reach for peamouth chub 1996 -- median, means (grey bars), outliers, and 10 th, 25 th, 75 th, and 90 th percentiles.


Figure 81. Box plots of age, length and weight by reach for mountain whitefish 1994 -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 82. Box plots of age, length, and weight by reach for mountain whitefish 1995 -- median, means (grey bars), outliers, and 10th, 25th, 75 th, and 90 th percentiles.


Figure 83. Box plots of age, length, and weight by reach for mountain whitefish 1996 -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 84. Box plots of age, length, and weight by reach for starry flounder 1994 -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 85. Box plots of age, length, and weight by reach for starry flounder 1995 -- median, means (grey bars), outliers, and 10 th, 25 th, 75 th, and 90 th percentiles.


Figure 86. Age frequency distributions for peamouth chub 1994.


Figure 87. Age frequency distributions for peamouth chub 1995.


Figure 88. Age frequency distributions for peamouth chub 1996.


Figure 89. Length frequency distributions for peamouth chub 1994.


Figure 90. Length frequency distributions for peamouth chub 1995.


Figure 91. Length frequency distributions for peamouth chub 1996.


Figure 92. Age frequency distributions for mountain whitefish 1994.


Figure 93. Age frequency distributions for mountain whitefish 1995.


Figure 94. Age frequency distributions for mountain whitefish 1996.


Figure 95. Length frequency distributions for mountain whitefish 1994.


Figure 96. Length frequency distributions for mountain whitefish 1995.


Figure 97. Length frequency distributions for mountain whitefish 1996.


Figure 98. Age frequency distributions for starry flounder 1994.


Figure 99. Age frequency distributions for starry flounder 1995.


Figure 100. Length frequency distributions for starry flounder 1994.


Figure 101. Length frequency distributions for starry flounder 1995.


Figure 102. Peamouth chub mean lengths, adjusted for mean age 6.03 years.


Figure 103. Mountain whitefish mean lengths, adjusted for age 4.10 years.


Figure 104. Mean fork length vs age for peamouth chub and mountain whitefish from reaches in the Fraser basin; Columbia and McGregor data from (Ash et al. 1984) and (McLeod et al. 1979), cited in (Zimmerman 1994).


Figure 105. Peamouth chub muscle and liver lipid and mesenteric fat, 1994 to 1996.


Figure 106. Mountain whitefish muscle and liver lipid and mesenteric fat, 1994 to 1996.


Figure 107. Starry flounder muscle and liver lipid, 1994 to 1996 (mesenteric fat indices=0).


Figure 108. Box plots of condition and hepatosomatic indices by reach for peamouth chub -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 109. Box plots of condition, relative weight, and hepatosomatic indices by reach for mountain whitefish -median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 110. Box plots of condition and hepatosomatic indices by reach for starry flounder -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 111. Mean condition index (by reach and year) versus sampling day for peamouth chub and mountain whitefish.


Figure 112. Box plots of the gonadosomatic index by reach for peamouth chub 1994 to 1996 -median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 113. Box plots of the gonadosomatic index by reach for mountain whitefish -- median, means (grey bars), outliers, and 10th, 25th, 75th, and 90th percentiles.


Figure 114. Scatter plots of gonad weight versus gutted weight by reach for peamouth chub females.


Figure 115. Scatter plots of gonad weight versus gutted weight by reach for peamouth chub males.


Figure 116. Scatter plots of gonad weight versus gutted weight by reach for mountain whitefish females 1994-6.


Figure 117. Scatter plots of gonad weight versus gutted weight by reach for mountain whitefish males.

## Peamouth Chub Females



Figure 118. Peamouth chub females in fall 1995 to 1996 -- plots of \% mature fish versus age by reach. Note that 1994 data are not included because immature fish were called "U" and sex was not identified histologically.


Figure 119. Peamouth chub females in fall 1995 to 1996 -- plots of \% mature fish versus length by reach. Note that 1994 data are not included because immature fish were called "U" and sex was not identified histologically.


Figure 120. Mountain whitefish females in fall 1994 to 1996-- plots of $\%$ mature fish versus age by reach.


Figure 121. Mountain whitefish females in fall 1994 to 1996 -- plots of \% mature fish versus length by reach.


Figure 122. Mountain whitefish males in fall 1994 to 1996 -- plots of \% mature fish versus age by reach.


Figure 123. Mountain whitefish males in fall 1994 to 1996 -- plots of $\%$ mature fish versus length by reach.


Figure 124. Scatter plots of log fecundity on log total weight by reach for peamouth chub and mountain whitefish in 1996.


Figure 125. Distributions of peamouth chub and mountain whitefish egg diameters in 1996.


Figure 126. Peamouth chub mean gut index (\%) and bile index (with SE) by reach and year.


Figure 127. Mountain whitefish mean gut index (\%) and bile index (with SE) by reach and year.


Figure 128. Starry flounder mean gut index (\%) and bile index (with SE) by reach and year.


Figure 129. Gut contents of peamouth chub and starry flounder in the Fraser River estuary in 1994.


Figure 130. Health assessment indices (HAI) for peamouth chub, mountain whitefish, and starry flounder - stacked bar charts showing abnormalities contributing to the HAI in each reach and year.


Figure 131. Fin assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of ratings and means and standard errors of index in each reach and year


Figure 132. Skin assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of ratings and means and standard errors of index in each reach and year


Figure 133. Gills assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of abnormalities and means and standard errors of index in each reach and year.


Figure 134. Parasite assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of ratings and means and standard errors of index in each reach and year.


Figure 135. Bile assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of ratings and means and standard errors of index in each reach and year.











|  | Nodular |
| :--- | :--- |
| mand | Fatty |
| max | Discoloured |
| $\rightarrow$ | Normal |
|  | Mean |

Note: Dark liver discolouration (grey, brown, and black) only in northern reaches: Nechako
Hansard to Marguerite, and Hansard to Marguerite, and North Thompson

Figure 136. Liver assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of abnormalities and means and standard errors of index in each reach and year.


Figure 137. Mesenteric fat assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of ratings and means and standard errors of index in each reach and year.


Figure 138. Spleen assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of abnormalities and means and standard errors of index in each reach and year.


Figure 139 Hindgut assessments for peamouth chub, mountain whitefish, and starry flounder -- incidences of ratings and means and standard errors of index in each reach and year.


Figure 140. Kidney assessments for peamouth chub -- incidences of abnormalities; all index values are 30 and are not plotted.







| L | Mottled |
| :---: | :---: |
| Timb | Granular |
| 区 $\times$ 奴 $\times$ | Urolithic |
|  | Brown/Grey |
| ־ | Parasites |
|  | Normal |
| $\bigcirc$ | Mean |

Figure 141. Kidney assessments for mountain whitefish and starry flounder -- incidences of abnormalities and means and standard errors of index in each reach and year.


Figure 142 Effect of larval tapeworm infestations on the GSI of peamouth chub from the Thompson and North Thompson rivers in 1994 and 1995 (M=male; U=unknown; mean GSI with standard error).


Figure 143. Gill histological assessment for peamouth chub, mountain whitefish, and starry flounder -- incidences of abnormalities.



LIVER HISTOLOGY

|  | Helminths |
| :--- | :--- |
| Fatty/Glycogen Vacuolation |  |
| Hydropic Vacuolation |  |
| Hepatocellular Steatosis |  |
| Inflammation |  |
| Granulomas |  |

Figure 144. Liver histological assessment for peamouth chub -- incidences of abnormalities.





LIVER HISTOLOGY

| Helminths |
| :--- |
| Fatty/Glycogen Vacuolation |
| Hydropic Vacuolation |
| Hepatocellular |
| Granulomas |
| Inflammation |

Figure 145. Liver histological assessment for mountain whitefish and starry flounder -- incidences of abnormalities.


Figure 146. Spleen histological assessment for peamouth chub, mountain whitefish, and starry flounder -- incidences of abnormalities.


Figure 147. Kidney histological assessment for peamouth chub, mountain whitefish, and starry flounder -- incidences of abnormalities.


Figure 148. Hindgut histological assessment for peamouth chub, mountain whitefish, and starry flounder -- incidences of abnormalities.

## Appendices

Appendix 1. ANOVA tables and multiple comparison test results of $\log 10$ EROD activities in peamouth chub, mountain whitefish, and starry flounder 1994 to 1996.

| Year | Sex | Source | DF | Sums of Squares | Mean Square F Value |  | Pr > F |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Peamouth Chub |  |  |  |  |  |  |  |
| 1994 |  | REACH | 9 | 21.980 | 2.442 | 14.89 | 0.0001 |
|  |  | SEX | 1 | 0.987 | 0.987 | 6.02 | 0.0153 |
|  |  | REACH*SEX | 9 | 0.468 | 0.052 | 0.32 | 0.9684 |
| 1995 |  | REACH | 8 | 2.071 | 0.259 | 6.24 | 0.0001 |
|  |  | SEX | 1 | 0.117 | 0.117 | 2.81 | 0.0961 |
|  |  | REACH*SEX | 8 | 0.155 | 0.019 | 0.47 | 0.8767 |
| 1996 |  | REACH | 3 | 0.569 | 0.190 | 3.72 | 0.0169 |
|  |  | SEX | 1 | 0.051 | 0.051 | 1.00 | 0.3231 |
|  |  | REACH*SEX | 3 | 0.035 | 0.012 | 0.23 | 0.8756 |
| Mountain Whitefish |  |  |  |  |  |  |  |
| 1994 | F | REACH | 6 | 10.750 | 1.792 | 3.84 | 0.0024 |
| 1994 | M | REACH | 6 | 25.686 | 4.281 | 19.18 | 0.0001 |
| 1995 | F | REACH | 6 | 1.664 | 0.277 | 3.27 | 0.0084 |
| 1995 | M | REACH | 6 | 1.683 | 0.280 | 8.67 | 0.0001 |
| 1996 | F | REACH | 1 | 0.039 | 0.039 | 0.31 | 0.5843 |
| 1996 | M | REACH | 1 | 0.108 | 0.108 | 1.06 | 0.3193 |
| Starry Flounder |  |  |  |  |  |  |  |
| 1995 |  | REACH | 1 | 0.813 | 0.813 | 14.66 | 0.0012 |

Peamouth chub 1994 - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN(i) =LSMEAN(j).
$\left.\begin{array}{lcccccccccc}\hline \mathrm{l} / \mathrm{j} & \text { Nechako } & \text { Hansard } & \begin{array}{c}\text { Wood- } \\ \text { pecker }\end{array} & \begin{array}{c}\text { Marg- } \\ \text { uerite }\end{array} & \text { Agassiz } & \text { Mission } & \text { Barnston } & \text { Main Arm } & \begin{array}{c}\text { North } \\ \text { Thomp- } \\ \text { son }\end{array} \\ \hline \text { Nechako } & . & 0.4973 & 0.0001 & 0.0001 & 0.0969 & 0.0001 & 0.0001 & 0.0001 & 0.2710 & 0.0001 \\ \text { son }\end{array}\right]$

Appendix 1. ANOVA tables and multiple comparison test results of $\log 10$ EROD activities in peamouth chub, mountain whitefish, and starry flounder 1994 to 1996. (continued)

Peamouth chub 1995 - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN(i) =LSMEAN(j).

| $\mathrm{l} / \mathrm{j}$ | Nechako | Hansard | Wood- <br> pecker | Marg- <br> uerite | Agassiz | Main Arm | North <br> Arm | North <br> Thomp- <br> son | Thomp- <br> son |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | . | 0.4660 | 0.5026 | 0.3277 | 0.0066 | 0.0001 | 0.0219 | 0.7457 | 0.0262 |
| Hansard | 0.4660 | . | 0.1699 | 0.1210 | 0.0007 | 0.0001 | 0.0022 | 0.6597 | 0.0024 |
| Woodpecker | 0.5026 | 0.1699 | . | 0.6781 | 0.0420 | 0.0007 | 0.1305 | 0.3128 | 0.1607 |
| Marguerite | 0.3277 | 0.1210 | 0.6781 | . | 0.1765 | 0.0212 | 0.4144 | 0.2089 | 0.4875 |
| Agassiz | 0.0066 | 0.0007 | 0.0420 | 0.1765 | . | 0.3782 | 0.4561 | 0.0020 | 0.3471 |
| MainArm | 0.0001 | 0.0001 | 0.0007 | 0.0212 | 0.3782 | . | 0.0517 | 0.0001 | 0.0232 |
| NorthArm | 0.0219 | 0.0022 | 0.1305 | 0.4144 | 0.4561 | 0.0517 | . | 0.0062 | 0.8459 |
| N.Thompson | 0.7457 | 0.6597 | 0.3128 | 0.2089 | 0.0020 | 0.0001 | 0.0062 | . | 0.0069 |
| Thompson | 0.0262 | 0.0024 | 0.1607 | 0.4875 | 0.3471 | 0.0232 | 0.8459 | 0.0069 | . |

Peamouth chub 1996 - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}:$ LSMEAN(i) $=\operatorname{LSMEAN}(\mathrm{j})$.

| i/j | Hansard | Wood- <br> pecker | Main Arm Main Arm <br> Sep |  |
| :--- | :---: | :---: | :---: | :---: |
| Hansard | . | 0.4172 | 0.0502 | 0.0037 |
| Woodpecker | 0.4172 | . | 0.1764 | 0.0691 |
| MainArm | 0.0502 | 0.1764 | . | 0.8330 |
| MainArm Sep | 0.0037 | 0.0691 | 0.8330 | . |

Mountain whitefish females 1994 - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : $\operatorname{LSMEAN}(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})$.

| $\mathrm{i} / \mathrm{j}$ | Nechako | Hansard | Wood- <br> pecker | Marg- <br> uerite | Agassiz | North <br> Thomp- <br> son | Thomp- <br> son |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | . | 1.0000 | 0.8085 | 0.0058 | 0.0049 | 0.0742 | 0.0045 |
| Hansard | 1.0000 | . | 0.8193 | 0.0094 | 0.0077 | 0.0918 | 0.0079 |
| Woodpecker | 0.8085 | 0.8193 | . | 0.0115 | 0.0095 | 0.1210 | 0.0095 |
| Marguerite | 0.0058 | 0.0094 | 0.0115 | . | 0.7962 | 0.3415 | 0.9543 |
| Agassiz | 0.0049 | 0.0077 | 0.0095 | 0.7962 | . | 0.2577 | 0.7470 |
| N.Thompson | 0.0742 | 0.0918 | 0.1210 | 0.3415 | 0.2577 | . | 0.3454 |
| Thompson | 0.0045 | 0.0079 | 0.0095 | 0.9543 | 0.7470 | 0.3454 | . |

Mountain whitefish males 1994 - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN $(\mathrm{i})=\mathrm{LSMEAN}(\mathrm{j})$.

| $\mathrm{i} / \mathrm{j}$ | Nechako | Hansard | Wood- <br> pecker | Marg- <br> uerite | Agassiz | North <br> Thomp- <br> son | Thomp- <br> son |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | . | 0.0748 | 0.0001 | 0.0001 | 0.0001 | 0.0117 | 0.0001 |
| Hansard | 0.0748 | . | 0.0001 | 0.0001 | 0.0002 | 0.4408 | 0.0193 |
| Woodpecker | 0.0001 | 0.0001 | . | 0.3603 | 0.0546 | 0.0001 | 0.0001 |
| Marguerite | 0.0001 | 0.0001 | 0.3603 | . | 0.3548 | 0.0001 | 0.0085 |
| Agassiz | 0.0001 | 0.0002 | 0.0546 | 0.3548 | . | 0.0017 | 0.0879 |
| N.Thompson | 0.0117 | 0.4408 | 0.0001 | 0.0001 | 0.0017 | . | 0.1025 |
| Thompson | 0.0001 | 0.0193 | 0.0001 | 0.0085 | 0.0879 | 0.1025 | . |

Appendix 1. ANOVA tables and multiple comparison test results of $\log 10$ EROD activities in peamouth chub, mountain whitefish, and starry flounder 1994 to 1996. (continued)

Mountain whitefish females 1995 - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN(i) $=\mathrm{LSMEAN}(\mathrm{j})$.

| i/j | Nechako | Hansard | Wood- <br> pecker | Marg- <br> uerite | Agassiz | North <br> Thomp- <br> son | Thomp- <br> son |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | . | 0.9542 | 0.4767 | 0.0066 | 0.8529 | 0.0739 | 0.7967 |
| Hansard | 0.9542 | . | 0.4484 | 0.0075 | 0.8082 | 0.0655 | 0.8400 |
| Woodpecker | 0.4767 | 0.4484 | . | 0.0034 | 0.5747 | 0.4433 | 0.3632 |
| Marguerite | 0.0066 | 0.0075 | 0.0034 | . | 0.0043 | 0.0001 | 0.0132 |
| Agassiz | 0.8529 | 0.8082 | 0.5747 | 0.0043 | . | 0.1075 | 0.6615 |
| N.Thompson | 0.0739 | 0.0655 | 0.4433 | 0.0001 | 0.1075 | . | 0.0470 |
| Thompson | 0.7967 | 0.8400 | 0.3632 | 0.0132 | 0.6615 | 0.0470 | . |

Appendix 1. ANOVA tables and multiple comparison test results of $\log 10$ EROD activities in peamouth chub, mountain whitefish, and starry flounder 1994 to 1996. (continued)
Mountain whitefish males 1995 - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN(i) $=\operatorname{LSMEAN}(\mathrm{j})$.

| i/j | Nechako | Hansard | Wood- <br> pecker | Marg- <br> uerite | Agassiz | North <br> Thomp- <br> son | Thomp- <br> son |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | . | 0.1363 | 0.1426 | 0.0179 | 0.0001 | 0.7097 | 0.0001 |
| Hansard | 0.1363 | . | 0.8300 | 0.0809 | 0.0006 | 0.2715 | 0.0018 |
| Woodpecker | 0.1426 | 0.8300 | . | 0.1147 | 0.0061 | 0.2534 | 0.0110 |
| Marguerite | 0.0179 | 0.0809 | 0.1147 | . | 0.8620 | 0.0272 | 0.8468 |
| Agassiz | 0.0001 | 0.0006 | 0.0061 | 0.8620 | . | 0.0001 | 0.9608 |
| N.Thompson | 0.7097 | 0.2715 | 0.2534 | 0.0272 | 0.0001 | . | 0.0001 |
| Thompson | 0.0001 | 0.0018 | 0.0110 | 0.8468 | 0.9608 | 0.0001 | . |

Appendix 2. Peamouth chub age, length and weight ANOVA tables and multiple comparison tests.

| Source of Variation | DF | Type IV SS | Mean Square | F Value | Pr > F |
| :--- | ---: | ---: | ---: | ---: | ---: |
| Log Age |  |  |  |  |  |
| REACH | 10 | 9.880320 | 0.988032 | 42.46 | 0.0001 |
| YEAR | 1 | 0.220922 | 0.220922 | 9.49 | 0.0021 |
| REACH*YEAR | 8 | 0.959425 | 0.119928 | 5.15 | 0.0001 |
| Log Length |  |  |  |  |  |
| REACH | 10 | 0.354718 | 0.035472 | 21.82 | 0.0001 |
| YEAR | 1 | 0.019152 | 0.019152 | 11.78 | 0.0006 |
| REACH*YEAR | 8 | 0.086263 | 0.010783 | 6.63 | 0.0001 |
| Log Weight |  |  |  |  |  |
| REACH | 10 | 4.866469 | 0.486647 | 26.51 | 0.0001 |
| YEAR | 1 | 0.052514 | 0.052514 | 2.86 | 0.0910 |
| REACH*YEAR | 8 | 1.503272 | 0.187909 | 10.24 | 0.0001 |

Appendix 2. Peamouth chub age, length and weight ANOVA tables and multiple comparison tests. (continued)
Peamouth chub age - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN $(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})$.

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j \& YEAR \& Agassiz

1994 \& Barnston

1994 \& Hansard

1994 \& | Main |
| :--- |
| Arm |
| 1994 | \& Marguerite 1994 \& Mission

1994 \& Nechako

1994 \& | North |
| :--- |
| Arm |
| 1994 | \& North

Thomp-
son
1994 \& Thompson 1994 \& Woodpecker

$$
1994
$$ \& Agassiz

1995 \& Hansard

1995 \& | Main |
| :--- |
| Arm |
| 1995 | \& Marguerite 1995 \& Nechako

1995 \& | North |
| :--- |
| Arm |
| 1995 | \& North

Thomp-
son
1995 \& Thompson 1995 \& Woodpecker

$$
1995
$$ <br>

\hline Agassiz \& 1994 \& \& 1.000 \& 0.000 \& 1.000 \& 0.140 \& 0.311 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 1.000 \& 0.993 \& 0.000 \& 1.000 \& 0.992 \& 0.900 \& 1.000 \& 0.012 \& 0.998 \& 0.029 <br>
\hline Barnston \& 1994 \& 1.000 \& \& 0.000 \& 1.000 \& 0.624 \& 0.055 \& 0.000 \& 1.000 \& 0.000 \& 0.006 \& 1.000 \& 0.660 \& 0.000 \& 0.995 \& 1.000 \& 1.000 \& 1.000 \& 0.108 \& 1.000 \& 0.214 <br>
\hline Hansard \& 1994 \& 0.000 \& 0.000 \& \& 0.000 \& 0.000 \& 0.000 \& 0.404 \& 0.000 \& 1.000 \& 0.009 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 <br>
\hline MainArm \& 1994 \& 1.000 \& 1.000 \& 0.000 \& \& 0.434 \& 0.178 \& 0.000 \& 1.000 \& 0.000 \& 0.003 \& 1.000 \& 0.937 \& 0.000 \& 1.000 \& 1.000 \& 0.998 \& 1.000 \& 0.062 \& 1.000 \& 0.128 <br>
\hline Marguerite \& 1994 \& 0.140 \& 0.624 \& 0.000 \& 0.434 \& \& 0.000 \& 0.913 \& 0.007 \& 0.004 \& 1.000 \& 0.978 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.003 \& 1.000 \& 1.000 \& 1.000 <br>
\hline Mission \& 1994 \& 0.311 \& 0.055 \& 0.000 \& 0.178 \& 0.000 \& . \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.009 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.998 \& 0.000 \& 0.000 \& 0.000 <br>
\hline Nechako \& 1994 \& 0.000 \& 0.000 \& 0.404 \& 0.000 \& 0.913 \& 0.000 \& \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.856 \& 0.135 \& 0.000 \& 1.000 \& 0.034 \& 0.999 <br>
\hline NorthArm \& 1994 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 0.007 \& 1.000 \& 0.000 \& \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 0.463 \& 0.150 \& 1.000 \& 0.000 \& 0.427 \& 0.001 <br>
\hline N.Thompson \& 1994 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.004 \& 0.000 \& 1.000 \& 0.000 \& \& 0.512 \& 0.000 \& 0.000 \& 0.283 \& 0.000 \& 0.012 \& 0.000 \& 0.000 \& 0.059 \& 0.000 \& 0.023 <br>
\hline Thompson \& 1994 \& 0.000 \& 0.006 \& 0.009 \& 0.003 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.512 \& \& 0.043 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 0.990 \& 0.000 \& 1.000 \& 0.803 \& 1.000 <br>
\hline Woodpecker \& 1994 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 0.978 \& 0.009 \& 0.000 \& 1.000 \& 0.000 \& 0.043 \& \& 0.218 \& 0.000 \& 0.779 \& 1.000 \& 1.000 \& 1.000 \& 0.446 \& 1.000 \& 0.680 <br>
\hline Agassiz \& 1995 \& 0.993 \& 0.660 \& 0.000 \& 0.937 \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.218 \& \& 0.000 \& 1.000 \& 0.011 \& 0.001 \& 1.000 \& 0.000 \& 0.003 \& 0.000 <br>
\hline Hansard \& 1995 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.283 \& 0.000 \& 0.000 \& 0.000 \& \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 <br>
\hline MainArm \& 1995 \& 1.000 \& 0.995 \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.779 \& 1.000 \& 0.000 \& . \& 0.067 \& 0.007 \& 1.000 \& 0.000 \& 0.031 \& 0.000 <br>
\hline Marguerite \& 1995 \& 0.992 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.856 \& 0.463 \& 0.012 \& 1.000 \& 1.000 \& 0.011 \& 0.000 \& 0.067 \& \& 1.000 \& 0.373 \& 1.000 \& 1.000 \& 1.000 <br>
\hline Nechako \& 1995 \& 0.900 \& 1.000 \& 0.000 \& 0.998 \& 1.000 \& 0.000 \& 0.135 \& 0.150 \& 0.000 \& 0.990 \& 1.000 \& 0.001 \& 0.000 \& 0.007 \& 1.000 \& \& 0.089 \& 1.000 \& 1.000 \& 1.000 <br>
\hline NorthArm \& 1995 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 0.003 \& 0.998 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 0.373 \& 0.089 \& \& 0.000 \& 0.300 \& 0.000 <br>
\hline N.Thompson \& 1995 \& 0.012 \& 0.108 \& 0.000 \& 0.062 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.059 \& 1.000 \& 0.446 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& . \& 1.000 \& 1.000 <br>
\hline Thompson \& 1995 \& 0.998 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.034 \& 0.427 \& 0.000 \& 0.803 \& 1.000 \& 0.003 \& 0.000 \& 0.031 \& 1.000 \& 1.000 \& 0.300 \& 1.000 \& \& 1.000 <br>
\hline Woodpecker \& 1995 \& 0.029 \& 0.214 \& 0.000 \& 0.128 \& 1.000 \& 0.000 \& 0.999 \& 0.001 \& 0.023 \& 1.000 \& 0.680 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& . <br>
\hline
\end{tabular}

Appendix 2. Peamouth chub age, length and weight ANOVA tables and multiple comparison tests.(continued)
Peamouth chub length - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN $(\mathrm{i})=$ LSMEAN $(\mathrm{j})$.

| $\begin{aligned} & \text { REACH } \\ & \mathrm{i} / \mathrm{j} \end{aligned}$ |  | Agassiz | Barnston | Hansard | MainAr m | Marguerite | Mission | Nechako | North Arm | North Thompson | Thompson | Woodpecker | Agassiz | Hansard | Main <br> Arm | Marguerite | Nechako | North Arm | North Thompson | Thompson | Woodpecker |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | YEAR | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1995 | 1995 | 1995 | 1995 | 1995 | 1995 | 1995 | 1995 | 1995 |
| Agassiz | 1994 |  | 1.000 | 0.784 | 1.000 | 0.022 | 0.626 | 0.000 | 1.000 | 0.000 | 0.775 | 0.917 | 1.000 | 0.031 | 1.000 | 0.105 | 1.000 | 1.000 | 0.573 | 0.997 | 1.000 |
| Barnston | 1994 | 1.000 |  | 0.982 | 1.000 | 0.004 | 0.245 | 0.000 | 1.000 | 0.000 | 0.980 | 0.547 | 1.000 | 0.105 | 1.000 | 0.032 | 1.000 | 1.000 | 0.906 | 1.000 | 1.000 |
| Hansard | 1994 | 0.784 | 0.982 |  | 1.000 | 0.000 | 0.000 | 0.001 | 1.000 | 0.676 | 1.000 | 0.000 | 0.378 | 1.000 | 0.019 | 0.000 | 1.000 | 1.000 | 1.000 | 1.000 | 0.159 |
| MainArm | 1994 | 1.000 | 1.000 | 1.000 |  | 0.000 | 0.000 | 0.000 | 1.000 | 0.112 | 1.000 | 0.001 | 0.956 | 1.000 | 0.187 | 0.000 | 1.000 | 1.000 | 1.000 | 1.000 | 0.736 |
| Marguerite | 1994 | 0.022 | 0.004 | 0.000 | 0.000 |  | 1.000 | 0.000 | 0.001 | 0.000 | 0.000 | 1.000 | 0.085 | 0.000 | 0.816 | 1.000 | 0.001 | 0.001 | 0.000 | 0.000 | 0.244 |
| Mission | 1994 | 0.626 | 0.245 | 0.000 | 0.000 | 1.000 | . | 0.000 | 0.091 | 0.000 | 0.000 | 1.000 | 0.934 | 0.000 | 1.000 | 1.000 | 0.080 | 0.093 | 0.000 | 0.000 | 0.998 |
| Nechako | 1994 | 0.000 | 0.000 | 0.001 | 0.000 | 0.000 | 0.000 |  | 0.000 | 1.000 | 0.001 | 0.000 | 0.000 | 0.116 | 0.000 | 0.000 | 0.000 | 0.000 | 0.002 | 0.000 | 0.000 |
| NorthArm | 1994 | 1.000 | 1.000 | 1.000 | 1.000 | 0.001 | 0.091 | 0.000 |  | 0.000 | 1.000 | 0.254 | 1.000 | 0.273 | 1.000 | 0.012 | 1.000 | 1.000 | 0.995 | 1.000 | 1.000 |
| N.Thompson | 1994 | 0.000 | 0.000 | 0.676 | 0.112 | 0.000 | 0.000 | 1.000 | 0.000 |  | 0.711 | 0.000 | 0.000 | 1.000 | 0.000 | 0.000 | 0.001 | 0.001 | 0.864 | 0.200 | 0.000 |
| Thompson | 1994 | 0.775 | 0.980 | 1.000 | 1.000 | 0.000 | 0.000 | 0.001 | 1.000 | 0.711 |  | 0.000 | 0.371 | 1.000 | 0.019 | 0.000 | 1.000 | 1.000 | 1.000 | 1.000 | 0.156 |
| Woodpecker | 1994 | 0.917 | 0.547 | 0.000 | 0.001 | 1.000 | 1.000 | 0.000 | 0.254 | 0.000 | 0.000 |  | 0.998 | 0.000 | 1.000 | 1.000 | 0.226 | 0.256 | 0.000 | 0.001 | 1.000 |
| Agassiz | 1995 | 1.000 | 1.000 | 0.378 | 0.956 | 0.085 | 0.934 | 0.000 | 1.000 | 0.000 | 0.371 | 0.998 |  | 0.007 | 1.000 | 0.266 | 1.000 | 1.000 | 0.218 | 0.887 | 1.000 |
| Hansard | 1995 | 0.031 | 0.105 | 1.000 | 1.000 | 0.000 | 0.000 | 0.116 | 0.273 | 1.000 | 1.000 | 0.000 | 0.007 |  | 0.000 | 0.000 | 0.349 | 0.312 | 1.000 | 1.000 | 0.002 |
| MainArm | 1995 | 1.000 | 1.000 | 0.019 | 0.187 | 0.816 | 1.000 | 0.000 | 1.000 | 0.000 | 0.019 | 1.000 | 1.000 | 0.000 |  | 0.942 | 1.000 | 1.000 | 0.009 | 0.125 | 1.000 |
| Marguerite | 1995 | 0.105 | 0.032 | 0.000 | 0.000 | 1.000 | 1.000 | 0.000 | 0.012 | 0.000 | 0.000 | 1.000 | 0.266 | 0.000 | 0.942 |  | 0.010 | 0.012 | 0.000 | 0.000 | 0.515 |
| Nechako | 1995 | 1.000 | 1.000 | 1.000 | 1.000 | 0.001 | 0.080 | 0.000 | 1.000 | 0.001 | 1.000 | 0.226 | 1.000 | 0.349 | 1.000 | 0.010 | . | 1.000 | 0.999 | 1.000 | 1.000 |
| NorthArm | 1995 | 1.000 | 1.000 | 1.000 | 1.000 | 0.001 | 0.093 | 0.000 | 1.000 | 0.001 | 1.000 | 0.256 | 1.000 | 0.312 | 1.000 | 0.012 | 1.000 |  | 0.997 | 1.000 | 1.000 |
| N.Thompson | 1995 | 0.573 | 0.906 | 1.000 | 1.000 | 0.000 | 0.000 | 0.002 | 0.995 | 0.864 | 1.000 | 0.000 | 0.218 | 1.000 | 0.009 | 0.000 | 0.999 | 0.997 |  | 1.000 | 0.082 |
| Thompson | 1995 | 0.997 | 1.000 | 1.000 | 1.000 | 0.000 | 0.000 | 0.000 | 1.000 | 0.200 | 1.000 | 0.001 | 0.887 | 1.000 | 0.125 | 0.000 | 1.000 | 1.000 | 1.000 |  | 0.596 |
| Woodpecker | 1995 | 1.000 | 1.000 | 0.159 | 0.736 | 0.244 | 0.998 | 0.000 | 1.000 | 0.000 | 0.156 | 1.000 | 1.000 | 0.002 | 1.000 | 0.515 | 1.000 | 1.000 | 0.082 | 0.596 | . |

Appendix 2. Peamouth chub age, length and weight ANOVA tables and multiple comparison tests. (continued)
Peamouth chub weight - multiple comparison table; $\mathrm{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN(i) =LSMEAN(j).

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j \& YEAR \& Agassiz

1994 \& Barnston

1994 \& Hansard

1994 \& Main Arm

$$
1994
$$ \& Marguerite 1994 \& Mission

1994 \& Nechako

1994 \& North Arm 1994 \& North Thompson 1994 \& | Thomp- |
| :--- |
| son |
| 1994 | \& Woodpecker 1994 \& Agassiz

1995 \& Hansard

1995 \& Main Arm 1995 \& Marguerite 1995 \& Nechako

1995 \& North Arm 1995 \& North Thompson 1995 \& | Thomp- |
| :--- |
| son |
| 1995 | \& Woodpecker 1995 <br>

\hline Agassiz \& 1994 \& \& 0.985 \& 0.025 \& 0.008 \& 0.362 \& 1.000 \& 0.000 \& 0.300 \& 0.000 \& 0.000 \& 1.000 \& 0.071 \& 0.000 \& 1.000 \& 0.999 \& 0.481 \& 1.000 \& 0.000 \& 0.000 \& 1.000 <br>
\hline Barnston \& 1994 \& 0.985 \& \& 1.000 \& 1.000 \& 0.000 \& 0.604 \& 0.000 \& 1.000 \& 0.008 \& 0.046 \& 0.034 \& 1.000 \& 0.293 \& 0.991 \& 0.016 \& 1.000 \& 1.000 \& 0.470 \& 0.153 \& 1.000 <br>
\hline Hansard \& 1994 \& 0.025 \& 1.000 \& . \& 1.000 \& 0.000 \& 0.003 \& 0.000 \& 1.000 \& 0.889 \& 0.999 \& 0.000 \& 1.000 \& 1.000 \& 0.029 \& 0.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 0.572 <br>
\hline MainArm \& 1994 \& 0.008 \& 1.000 \& 1.000 \& \& 0.000 \& 0.001 \& 0.000 \& 1.000 \& 0.984 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.009 \& 0.000 \& 1.000 \& 0.999 \& 1.000 \& 1.000 \& 0.294 <br>
\hline Marguerite \& 1994 \& 0.362 \& 0.000 \& 0.000 \& 0.000 \& . \& 0.934 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.325 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.011 <br>
\hline Mission \& 1994 \& 1.000 \& 0.604 \& 0.003 \& 0.001 \& 0.934 \& . \& 0.000 \& 0.053 \& 0.000 \& 0.000 \& 1.000 \& 0.010 \& 0.000 \& 1.000 \& 1.000 \& 0.105 \& 0.878 \& 0.000 \& 0.000 \& 1.000 <br>
\hline Nechako \& 1994 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& . \& 0.000 \& 0.199 \& 0.042 \& 0.000 \& 0.000 \& 0.008 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.002 \& 0.011 \& 0.000 <br>
\hline NorthArm \& 1994 \& 0.300 \& 1.000 \& 1.000 \& 1.000 \& 0.000 \& 0.053 \& 0.000 \& \& 0.175 \& 0.571 \& 0.001 \& 1.000 \& 0.981 \& 0.336 \& 0.001 \& 1.000 \& 1.000 \& 0.999 \& 0.900 \& 0.998 <br>
\hline N.Thompson \& 1994 \& 0.000 \& 0.008 \& 0.889 \& 0.984 \& 0.000 \& 0.000 \& 0.199 \& 0.175 \& . \& 1.000 \& 0.000 \& 0.626 \& 1.000 \& 0.000 \& 0.000 \& 0.112 \& 0.003 \& 1.000 \& 1.000 \& 0.000 <br>
\hline Thompson \& 1994 \& 0.000 \& 0.046 \& 0.999 \& 1.000 \& 0.000 \& 0.000 \& 0.042 \& 0.571 \& 1.000 \& \& 0.000 \& 0.970 \& 1.000 \& 0.000 \& 0.000 \& 0.421 \& 0.019 \& 1.000 \& 1.000 \& 0.000 <br>
\hline Woodpecker \& 1994 \& 1.000 \& 0.034 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.001 \& 0.000 \& 0.000 \& \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.002 \& 0.100 \& 0.000 \& 0.000 \& 0.935 <br>
\hline Agassiz \& 1995 \& 0.071 \& 1.000 \& 1.000 \& 1.000 \& 0.000 \& 0.010 \& 0.000 \& 1.000 \& 0.626 \& 0.970 \& 0.000 \& \& 1.000 \& 0.082 \& 0.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 0.853 <br>
\hline Hansard \& 1995 \& 0.000 \& 0.293 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.008 \& 0.981 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& . \& 0.000 \& 0.000 \& 0.933 \& 0.145 \& 1.000 \& 1.000 \& 0.003 <br>
\hline MainArm \& 1995 \& 1.000 \& 0.991 \& 0.029 \& 0.009 \& 0.325 \& 1.000 \& 0.000 \& 0.336 \& 0.000 \& 0.000 \& 1.000 \& 0.082 \& 0.000 \& \& 0.999 \& 0.526 \& 1.000 \& 0.000 \& 0.000 \& 1.000 <br>
\hline Marguerite \& 1995 \& 0.999 \& 0.016 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.001 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.999 \& \& 0.002 \& 0.039 \& 0.000 \& 0.000 \& 0.526 <br>
\hline Nechako \& 1995 \& 0.481 \& 1.000 \& 1.000 \& 1.000 \& 0.000 \& 0.105 \& 0.000 \& 1.000 \& 0.112 \& 0.421 \& 0.002 \& 1.000 \& 0.933 \& 0.526 \& 0.002 \& \& 1.000 \& 0.990 \& 0.786 \& 1.000 <br>
\hline NorthArm \& 1995 \& 1.000 \& 1.000 \& 1.000 \& 0.999 \& 0.000 \& 0.878 \& 0.000 \& 1.000 \& 0.003 \& 0.019 \& 0.100 \& 1.000 \& 0.145 \& 1.000 \& 0.039 \& 1.000 \& \& 0.254 \& 0.069 \& 1.000 <br>
\hline N.Thompson \& 1995 \& 0.000 \& 0.470 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.002 \& 0.999 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.990 \& 0.254 \& \& 1.000 \& 0.006 <br>
\hline Thompson \& 1995 \& 0.000 \& 0.153 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.011 \& 0.900 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.786 \& 0.069 \& 1.000 \& \& 0.001 <br>
\hline Woodpecker \& 1995 \& 1.000 \& 1.000 \& 0.572 \& 0.294 \& 0.011 \& 1.000 \& 0.000 \& 0.998 \& 0.000 \& 0.000 \& 0.935 \& 0.853 \& 0.003 \& 1.000 \& 0.526 \& 1.000 \& 1.000 \& 0.006 \& 0.001 \& <br>
\hline
\end{tabular}

Appendix 3. Mountain whitefish age, length and weight ANOVA tables and multiple comparison tests.

| Source | DF | Type III SS | Mean Square | F Value | $\boldsymbol{P r}>\boldsymbol{F}$ |
| :--- | :---: | ---: | ---: | ---: | ---: |
| Log Age |  |  |  |  |  |
| REACH | 6 | 10.170370 | 1.695062 | 43.01 | 0.0001 |
| YEAR | 1 | 0.438775 | 0.438775 | 11.13 | 0.0009 |
| REACH*YEAR | 6 | 2.891746 | 0.481958 | 12.23 | 0.0001 |
| Log Length |  |  |  |  |  |
| REACH | 6 | 0.119865 | 0.019977 | 10.31 | 0.0001 |
| YEAR | 1 | 0.005105 | 0.005105 | 2.64 | 0.1049 |
| REACH*YEAR | 6 | 0.096851 | 0.016142 | 8.33 | 0.0001 |
| Log Weight |  |  |  |  |  |
| REACH | 6 | 1.705381 | 0.284230 | 13.02 | 0.0001 |
| YEAR | 1 | 0.187726 | 0.187726 | 8.60 | 0.0035 |
| REACH*YEAR | 6 | 1.173077 | 0.195513 | 8.96 | 0.0001 |

Mountain whitefish age - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN $(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})$.

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline $$
\overline{\mathrm{REACH}}
$$
i/j \& YEAR \& Agassiz

1994 \& Hansard

1994 \& | Margue- |
| :---: |
| rite |

1994 \& Nechako

1994 \& North Thompson 1994 \& | Thomp- |
| :--- |
| son |
| 1994 | \& Woodpecker 1994 \& Agassiz

1995 \& Hansard

1995 \& $$
\begin{gathered}
\begin{array}{c}
\text { Margue- } \\
\text { rite }
\end{array} \\
1995
\end{gathered}
$$ \& Nechako

1995 \& North Thompson 1995 \& Thompson 1995 \& Woodpecker 1995 <br>
\hline Agassiz \& 1994 \& \& 0.000 \& 0.918 \& 0.000 \& 0.003 \& 1.000 \& 0.000 \& 0.015 \& 0.000 \& 0.982 \& 0.000 \& 0.000 \& 1.000 \& 0.702 <br>
\hline Hansard \& 1994 \& 0.000 \& . \& 0.273 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.990 \& 1.000 \& 1.000 \& 0.000 \& 0.028 <br>
\hline Marguerite \& 1994 \& 0.918 \& 0.273 \& . \& 0.010 \& 0.999 \& 0.908 \& 0.473 \& 1.000 \& 0.000 \& 1.000 \& 0.238 \& 0.001 \& 0.565 \& 1.000 <br>
\hline Nechako \& 1994 \& 0.000 \& 1.000 \& 0.010 \& . \& 0.590 \& 0.000 \& 1.000 \& 0.511 \& 0.000 \& 0.490 \& 1.000 \& 1.000 \& 0.000 \& 0.000 <br>
\hline N.Thompson \& 1994 \& 0.003 \& 1.000 \& 0.999 \& 0.590 \& . \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.148 \& 0.000 \& 0.927 <br>
\hline Thompson \& 1994 \& 1.000 \& 0.000 \& 0.908 \& 0.000 \& 0.000 \& \& 0.000 \& 0.001 \& 0.000 \& 0.994 \& 0.000 \& 0.000 \& 1.000 \& 0.434 <br>
\hline Woodpecker \& 1994 \& 0.000 \& 1.000 \& 0.473 \& 1.000 \& 1.000 \& 0.000 \& . \& 1.000 \& 0.000 \& 0.999 \& 1.000 \& 0.973 \& 0.000 \& 0.073 <br>
\hline Agassiz \& 1995 \& 0.015 \& 1.000 \& 1.000 \& 0.511 \& 1.000 \& 0.001 \& 1.000 \& \& 0.000 \& 1.000 \& 1.000 \& 0.131 \& 0.000 \& 0.999 <br>
\hline Hansard \& 1995 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& . \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 <br>
\hline Marguerite \& 1995 \& 0.982 \& 0.990 \& 1.000 \& 0.490 \& 1.000 \& 0.994 \& 0.999 \& 1.000 \& 0.000 \& . \& 0.985 \& 0.208 \& 0.909 \& 1.000 <br>
\hline Nechako \& 1995 \& 0.000 \& 1.000 \& 0.238 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.985 \& \& 1.000 \& 0.000 \& 0.022 <br>
\hline N.Thompson \& 1995 \& 0.000 \& 1.000 \& 0.001 \& 1.000 \& 0.148 \& 0.000 \& 0.973 \& 0.131 \& 0.000 \& 0.208 \& 1.000 \& \& 0.000 \& 0.000 <br>
\hline Thompson \& 1995 \& 1.000 \& 0.000 \& 0.565 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.909 \& 0.000 \& 0.000 \& . \& 0.142 <br>
\hline Woodpecker \& 1995 \& 0.702 \& 0.028 \& 1.000 \& 0.000 \& 0.927 \& 0.434 \& 0.073 \& 0.999 \& 0.000 \& 1.000 \& 0.022 \& 0.000 \& 0.142 \& . <br>
\hline
\end{tabular}

Appendix 3. Mountain whitefish age, length and weight ANOVA tables and multiple comparison tests.

Mountain whitefish length - multiple comparison table; $\operatorname{Pr}>|T|, H_{0}$ : LSMEAN(i) =LSMEAN(j).

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline \multirow[t]{2}{*}{\begin{tabular}{l}
REACH \\
i/j
\end{tabular}} \& \& \begin{tabular}{l}
Agassi \\
z
\end{tabular} \& Hansard

1994 \& Marguerite \& Nechako \& North Thompson \& Thompson \& Woodpecker \& Agassiz

1995 \& | Hansar |
| :--- |
| d | \& Margue -rite \& Nechak 0 \& North Thomp-son \& Thompson \& Woodpecker <br>

\hline \& YEAR \& 1994 \& 1994 \& 1994 \& 1994 \& 1994 \& 1994 \& 1994 \& 1995 \& 1995 \& 1995 \& 1995 \& 1995 \& 1995 \& 1995 <br>
\hline Agassiz \& 1994 \& \& 0.923 \& 1.000 \& 1.000 \& 0.715 \& 0.179 \& 0.998 \& 0.279 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 <br>
\hline Hansard \& 1994 \& 0.923 \& \& 0.571 \& 0.022 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 1.000 \& 0.636 \& 0.648 \& 0.005 \& 0.007 \& 0.776 <br>
\hline Marguerite \& 1994 \& 1.000 \& 0.571 \& . \& 1.000 \& 0.767 \& 0.033 \& 1.000 \& 0.301 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 <br>
\hline Nechako \& 1994 \& 1.000 \& 0.022 \& 1.000 \& . \& 0.787 \& 0.000 \& 1.000 \& 0.272 \& 0.635 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 <br>
\hline N.Thompson \& 1994 \& 0.715 \& 0.000 \& 0.767 \& 0.787 \& . \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 0.038 \& 0.983 \& 0.958 \& 0.023 <br>
\hline Thompson \& 1994 \& 0.179 \& 1.000 \& 0.033 \& 0.000 \& 0.000 \& \& 0.000 \& 0.000 \& 0.865 \& 0.109 \& 0.016 \& 0.000 \& 0.000 \& 0.028 <br>
\hline Woodpecker \& 1994 \& 0.998 \& 0.000 \& 1.000 \& 1.000 \& 1.000 \& 0.000 \& . \& 1.000 \& 0.004 \& 1.000 \& 0.443 \& 1.000 \& 1.000 \& 0.323 <br>
\hline Agassiz \& 1995 \& 0.279 \& 0.000 \& 0.301 \& 0.272 \& 1.000 \& 0.000 \& 1.000 \& \& 0.000 \& 1.000 \& 0.006 \& 0.599 \& 0.500 \& 0.004 <br>
\hline Hansard \& 1995 \& 1.000 \& 1.000 \& 1.000 \& 0.635 \& 0.000 \& 0.865 \& 0.004 \& 0.000 \& . \& 0.998 \& 1.000 \& 0.276 \& 0.360 \& 1.000 <br>
\hline Marguerite \& 1995 \& 1.000 \& 0.636 \& 1.000 \& 1.000 \& 1.000 \& 0.109 \& 1.000 \& 1.000 \& 0.998 \& . \& 1.000 \& 1.000 \& 1.000 \& 1.000 <br>
\hline Nechako \& 1995 \& 1.000 \& 0.648 \& 1.000 \& 1.000 \& 0.038 \& 0.016 \& 0.443 \& 0.006 \& 1.000 \& 1.000 \& \& 1.000 \& 1.000 \& 1.000 <br>
\hline N.Thompson \& 1995 \& 1.000 \& 0.005 \& 1.000 \& 1.000 \& 0.983 \& 0.000 \& 1.000 \& 0.599 \& 0.276 \& 1.000 \& 1.000 \& . \& 1.000 \& 1.000 <br>
\hline Thompson \& 1995 \& 1.000 \& 0.007 \& 1.000 \& 1.000 \& 0.958 \& 0.000 \& 1.000 \& 0.500 \& 0.360 \& 1.000 \& 1.000 \& 1.000 \& . \& 1.000 <br>
\hline Woodpecker \& 1995 \& 1.000 \& 0.776 \& 1.000 \& 1.000 \& 0.023 \& 0.028 \& 0.323 \& 0.004 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& . <br>
\hline
\end{tabular}

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline $$
\begin{aligned}
& \hline \text { REACH } \\
& \text { i/j }
\end{aligned}
$$ \& YEAR \& $$
\begin{gathered}
\text { Agassi } \\
\text { z } \\
\\
1994
\end{gathered}
$$ \& Hansard

1994 \& Marguerite

$$
1994
$$ \& Nechako

1994 \& North Thompson 1994 \& Thompson

$$
1994
$$ \& Woodpecker

$$
1994
$$ \& Agassiz

1995 \& Hansard

1995 \& Marguerite

$$
1995
$$ \& Nechako

1995 \& North Thompson 1995 \& $$
\begin{gathered}
\hline \begin{array}{c}
\text { Thomp- } \\
\text { son }
\end{array} \\
\\
1995
\end{gathered}
$$ \& Woodpecker

$$
1995
$$ <br>

\hline Agassiz \& 1994 \& . \& 0.994 \& 1.000 \& 1.000 \& 0.043 \& 1.000 \& 0.413 \& 0.000 \& 1.000 \& 1.000 \& 1.000 \& 0.955 \& 0.998 \& 1.000 <br>
\hline Hansard \& 1994 \& 0.994 \& \& 0.992 \& 0.006 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.998 \& 0.720 \& 0.628 \& 0.000 \& 0.000 \& 0.006 <br>
\hline Marguerite \& 1994 \& 1.000 \& 0.992 \& \& 1.000 \& 0.010 \& 1.000 \& 0.168 \& 0.000 \& 1.000 \& 1.000 \& 1.000 \& 0.776 \& 0.964 \& 1.000 <br>
\hline Nechako \& 1994 \& 1.000 \& 0.006 \& 1.000 \& . \& 0.197 \& 0.150 \& 0.969 \& 0.000 \& 0.917 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 <br>
\hline N.Thompson \& 1994 \& 0.043 \& 0.000 \& 0.010 \& 0.197 \& \& 0.000 \& 1.000 \& 0.715 \& 0.000 \& 0.996 \& 0.001 \& 0.998 \& 0.928 \& 0.215 <br>
\hline Thompson \& 1994 \& 1.000 \& 1.000 \& 1.000 \& 0.150 \& 0.000 \& . \& 0.000 \& 0.000 \& 1.000 \& 0.998 \& 1.000 \& 0.001 \& 0.005 \& 0.149 <br>
\hline Woodpecker \& 1994 \& 0.413 \& 0.000 \& 0.168 \& 0.969 \& 1.000 \& 0.000 \& . \& 0.054 \& 0.002 \& 1.000 \& 0.034 \& 1.000 \& 1.000 \& 0.976 <br>
\hline Agassiz \& 1995 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.715 \& 0.000 \& 0.054 \& \& 0.000 \& 0.059 \& 0.000 \& 0.004 \& 0.001 \& 0.000 <br>
\hline Hansard \& 1995 \& 1.000 \& 0.998 \& 1.000 \& 0.917 \& 0.000 \& 1.000 \& 0.002 \& 0.000 \& . \& 1.000 \& 1.000 \& 0.054 \& 0.185 \& 0.913 <br>
\hline Marguerite \& 1995 \& 1.000 \& 0.720 \& 1.000 \& 1.000 \& 0.996 \& 0.998 \& 1.000 \& 0.059 \& 1.000 \& \& 1.000 \& 1.000 \& 1.000 \& 1.000 <br>
\hline Nechako \& 1995 \& 1.000 \& 0.628 \& 1.000 \& 1.000 \& 0.001 \& 1.000 \& 0.034 \& 0.000 \& 1.000 \& 1.000 \& \& 0.464 \& 0.832 \& 1.000 <br>
\hline N.Thompson \& 1995 \& 0.955 \& 0.000 \& 0.776 \& 1.000 \& 0.998 \& 0.001 \& 1.000 \& 0.004 \& 0.054 \& 1.000 \& 0.464 \& . \& 1.000 \& 1.000 <br>
\hline \& 1995 \& 0.998 \& 0.000 \& 0.964 \& 1.000 \& 0.928 \& 0.005 \& 1.000 \& 0.001 \& 0.185 \& 1.000 \& 0.832 \& 1.000 \& . \& 1.000 <br>
\hline Woodpecker \& 1995 \& 1.000 \& 0.006 \& 1.000 \& 1.000 \& 0.215 \& 0.149 \& 0.976 \& 0.000 \& 0.913 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& . <br>
\hline
\end{tabular}

Appendix 4. Starry flounder size, age, length and weight ANOVA tables and multiple comparison tests.

| Source | DF | Type III SS | Mean Square | F Value | Pr $>\boldsymbol{F}$ |
| :--- | :---: | ---: | ---: | ---: | ---: |
| Log Age |  |  |  |  |  |
| REACH | 1 | 0.007453 | 0.007453 | 0.38 | 0.5381 |
| YEAR | 1 | 0.472998 | 0.472998 | 24.13 | 0.0001 |
| REACH*YEAR | 1 | 0.050144 | 0.050144 | 2.56 | 0.1110 |
| Log Length |  |  |  |  |  |
| REACH | 1 | 0.004900 | 0.004900 | 2.81 | 0.0949 |
| YEAR | 1 | 0.001556 | 0.001556 | 0.89 | 0.3456 |
| REACH*YEAR | 1 | 0.006063 | 0.006063 | 3.48 | 0.0634 |
| Log Weight |  |  |  |  |  |
| REACH | 1 | 0.053476 | 0.053476 | 3.50 | 0.0627 |
| YEAR | 1 | 0.000937 | 0.000937 | 0.06 | 0.8047 |
| REACH*YEAR | 1 | 0.071919 | 0.071919 | 4.70 | 0.0311 |

Starry flounder age - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN(i) =LSMEAN(j).

| REACH |  | MainArm | NorthArm | MainArm | NorthArm |
| :--- | :---: | :---: | :---: | :---: | :---: |
| i/j | YEAR | 1994 | 1994 | 1995 | 1995 |
| MainArm | 1994 | . | 0.9823 | 0.1030 | 0.0008 |
| NorthArm | 1994 | 0.9823 | . | 0.0152 | 0.0001 |
| MainArm | 1995 | 0.1030 | 0.0152 | . | 0.5188 |
| NorthArm | 1995 | 0.0008 | 0.0001 | 0.5188 | . |

Starry flounder length - multiple comparison table; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H}_{0}$ : LSMEAN(i) =LSMEAN(j).

| REACH |  | MainArm | NorthArm | MainArm | NorthArm |
| :--- | :---: | :---: | :---: | :---: | :---: |
|  | YEAR | 1994 | 1994 | 1995 | 1995 |
| MainArm | 1994 | . | 0.0756 | 0.9861 | 0.9964 |
| NorthArm | 1994 | 0.0756 | . | 0.3140 | 0.2649 |
| MainArm | 1995 | 0.9861 | 0.3140 | . | 1.0000 |
| NorthArm | 1995 | 0.9964 | 0.2649 | 1.0000 | . |

Starry flounder weight - multiple comparison table; $\operatorname{Pr}>|T|, H_{0}$ : LSMEAN $(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})$.

| REACH |  | MainArm | NorthArm | MainArm | NorthArm |
| :--- | :---: | :---: | :---: | :---: | :---: |
|  | YEAR | 1994 | 1994 | 1995 | 1995 |
| MainArm | 1994 | . | 0.0280 | 0.6733 | 0.8297 |
| NorthArm | 1994 | 0.0280 | . | 0.5688 | 0.4372 |
| MainArm | 1995 | 0.6733 | 0.5688 | . | 1.0000 |
| NorthArm | 1995 | 0.8297 | 0.4372 | 1.0000 | . |

Appendix 5. Peamouth chub age-classes used in log10 length ANCOVA to test for differences in growth rates among reaches and between years.

| Reach | Year | Age Range |
| :--- | :---: | :---: |
| Nechako | 1994 | 5 to 9 |
| Nechako | 1995 | 6 to 9 |
| Hansard | 1994 | 8 to 11 |
| Hansard | 1995 | 8 to 13 |
| Hansard | 1996 | 8 to 15 |
| Woodpecker | 1994 | 5 to 9 |
| Woodpecker | 1995 | 5 to 9 |
| Woodpecker | 1996 | 6 to 10 |
| Marguerite | 1994 | 6 to 11 |
| Marguerite | 1995 | 5 to 8 |
| Agassiz | Both | 4 to 8 |
| Mission | 1994 | 3 to 5 |
| Barnston | 1994 | 4 to 9 |
| MainArm | 1994 | 4 to 8 |
| MainArm | 1995 | 4 to 7 |
| MainArm | 1996 | 3 to 14 |
| NorthArm | 1994 | 4 to 9 |
| NorthArm | 1995 | 4 to 7 |
| N.Thompson | 1994 | 6 to 9 |
| N.Thompson | 1995 | 5 to 9 |
| Thompson | Both | 5 to 9 |

Appendix 6. Mountain whitefish age-classes used in log10 length ANCOVA to test for differences in growth rates among reaches and between years.

| Reach | Year | Age Range |
| :--- | :---: | :---: |
| Nechako | Both | 3 to 8 |
| Hansard | 1994 | 3 to 10 |
| Hansard | 1995 | 6 to 12 |
| Hansard | 1996 | 4 to 11 |
| Woodpecker | 1994 | 3 to 6 |
| Woodpecker | 1995 | 3 to 6 |
| Woodpecker | 1996 | 3 to 5 |
| Marguerite | 1994 | 2 to 5 |
| Marguerite | 1995 | 3 to 10 |
| Agassiz | 1994 | 2 to 4 |
| Agassiz | 1995 | 3 to 5 |
| N.Thompson | 1994 | 3 to 5 |
| N.Thompson | 1995 | 5 to 8 |
| Thompson | Both | 2 to 5 |

Appendix 7. Peamouth chub log10 length ANCOVA table, to test for differences in growth rates.

| Source of Variation | DF | Sum of <br> Squares | Mean <br> Square | F Value | Pr > F |
| :--- | :---: | :---: | :---: | :---: | :---: |
| LAGE | 1 | 0.2017 | 0.2017 | 282.1 | 0.0001 |
| REACH*YEAR | 21 | 0.0264 | 0.0013 | 1.8 | 0.0188 |
| LAGE*REACH*YEAR | 21 | 0.0165 | 0.0008 | 1.1 | 0.3416 |

Appendix 8. Peamouth chub age-adjusted mean lengths and 95\% confidence limits (mean age 6.03 years, CLM 5.90-6.15).

|  |  |  | Adjusted <br> Mean | 95\% Lower <br> Confidence | 95\% Upper <br> Confidence |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Reach | Year | N | Length (mm) | Limit | Limit |
| Nechako | 1994 | 51 | 224.0 | 219.1 | 229.0 |
| Hansard | 1994 | 36 | 197.1 | 189.5 | 205.1 |
| Hansard | 1995 | 40 | 200.4 | 187.0 | 214.8 |
| Hansard | 1996 | 48 | 201.0 | 189.6 | 213.1 |
| Woodpecker | 1994 | 39 | 204.2 | 200.2 | 208.3 |
| Woodpecker | 1995 | 42 | 200.5 | 195.8 | 205.2 |
| Woodpecker | 1996 | 43 | 203.2 | 197.0 | 209.6 |
| Marguerite | 1994 | 30 | 193.6 | 187.1 | 200.4 |
| Marguerite | 1995 | 21 | 190.8 | 185.1 | 196.7 |
| Agassiz | 1994 | 40 | 214.9 | 210.4 | 219.5 |
| Agassiz | 1995 | 43 | 219.1 | 213.5 | 224.8 |
| Mission | 1994 | 48 | 211.2 | 200.5 | 222.4 |
| Barnston | 1994 | 50 | 216.2 | 212.2 | 220.3 |
| MainArm | 1994 | 45 | 219.5 | 214.0 | 225.2 |
| MainArm | 1995 | 39 | 218.3 | 211.4 | 225.5 |
| MainArm | 1996 | 77 | 214.5 | 211.0 | 218.1 |
| NorthArm | 1994 | 44 | 223.5 | 218.5 | 228.6 |
| NorthArm | 1995 | 48 | 221.7 | 215.4 | 228.3 |
| N.Thompson | 1994 | 37 | 215.7 | 209.2 | 222.4 |
| N.Thompson | 1995 | 33 | 218.6 | 213.8 | 223.6 |
| Thompson | 1994 | 36 | 217.4 | 212.9 | 222.0 |
| Thompson | 1995 | 44 | 219.9 | 216.0 | 224.0 |
| When included in analysis: |  |  |  | 210.4 | 219.1 |
| Nechako | 1995 | 36 | 216.6 | 210.4 |  |

Appendix 9. Peamouth chub - results of multiple comparison test for differences in age-adjusted lengths among reaches in each year, using GT2method; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{HO}: \operatorname{LSMEAN}(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})$.

| Reach i/j |  | Nechak 0 | Hansard | Woodpecker | Marguerite | Agassiz | $\begin{aligned} & \text { issio } \\ & \text { n } \end{aligned}$ | Barnston | Main <br> Arm | North Arm | North Thomp | Thomp -SOn | ansard | Woodpecker | Marguerite | Agassiz | Main <br> Arm | North Arm | North Thomp | Thomp -SOn | sard | Woodpecker | Main <br> Arm |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  | Year | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | 1994 | -son 1994 | 1994 | 1995 | 1995 | 1995 | 1995 | 1995 | 1995 | $\begin{aligned} & \text {-son } \\ & 1995 \end{aligned}$ | 1995 | 1996 | 1996 | 1996 |
| Nechako | 1994 |  | 0.0001 | 0.0001 | 0.0001 | 0.8544 | 0.9999 | 0.9768 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.4636 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.1485 | 0.0002 | 0.4087 |
| Hansard | 1994 | 0.000 |  | 1.0000 | 1.0000 | 0.0352 | 0.9998 | 0.0081 | 0.0017 | 0.0001 | 0.0896 | 0.0042 | 1.0000 | 1.0000 | 1.0000 | 0.0029 | 0.0207 | 0.0007 | 0.0018 | 0.0002 | . 0000 | 1.0000 | 51 |
| Woodpecker | 1994 | 0.0001 | 1.0000 |  | 0.8623 | 0.1197 | 1.0000 | 0.0099 | 0.0027 | 0.0001 | 0.5180 | 0.0053 | 1.0000 | 1.0000 | 0.0605 | 0.0062 | 0.1235 | 0.0012 | 0.0018 | 0.0001 | 1.0000 | 1.0000 | 0.0443 |
| Marguerite | 1994 | 0.0001 | 1.0000 | 0.8623 |  | 0.0001 | 0.7573 | 0.0001 | 0.0001 | 0.0001 | 0.0011 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 0.0001 | 0.0002 | 0.0001 | 0.0001 | 0.0001 | 1.0000 | 0.9999 | 0.0001 |
| Agassiz | 1994 | 0.8544 | 0.0352 | 0.1197 | 0.0001 |  | 1.0000 | 1.0000 | 1.0000 | 0.9469 | 1.0000 | 1.0000 | 1.0000 | 0.0039 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9995 | 0.5426 | 1.0000 |
| Mission | 1994 | 0.9999 | 0.9998 | 1.0000 | 0.7573 | 1.0000 |  | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.1978 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 000 | 1.0000 | 00 |
| Barnston | 1994 | 0.9768 | 0.0081 | 0.0099 | 0.0001 | 1.0000 | 1.0000 |  | 1.0000 | 0.9965 | 1.0000 | 1.0000 | 0.9998 | 0.0002 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9875 | 0.1715 | 1.0000 |
| MainArm | 1994 | 1.0000 | 0.0017 | 0.0027 | 0.0001 | 1.0000 | 1.0000 | 1.0000 |  | 1.0000 | 1.0000 | 1.0000 | 0.9675 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | . 7763 | 0.0373 | 00 |
| NorthArm | 1994 | 1.0000 | 0.0001 | 0.0001 | 0.0001 | 0.9469 | 1.0000 | 0.9965 | 1.0000 |  | 1.0000 | 1.0000 | 0.5288 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.1864 | 0.0003 | 0.5962 |
| N.Thompso | 1994 | 1.0000 | 0.0896 | 0.5180 | 0.0011 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |  | 1.0000 | 1.0000 | 0.0453 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9996 | 0.8054 | 1.0000 |
| , |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Thompson | 1994 | 1.0000 | 0.0042 | 0.0053 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |  | 0.9976 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9468 | 0.0925 | 00 |
| Hansard | 1995 | 0.4636 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9998 | 0.9675 | 0.5288 | 1.0000 | 0.9976 |  | 1.0000 | 1.0000 | 0.9825 | 0.9979 | 0.8470 | 0.9850 | 0.9130 | 1.0000 | 1.0000 | 1.0000 |
| Woodpecker | 1995 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 0.0039 | 1.0000 | 0.0002 | 0.0001 | 0.0001 | 0.0453 | 0.0001 | 1.0000 |  | 0.9279 | 0.0002 | 0.0072 | 0.0001 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 0.0010 |
| Marguerite | 1995 | 0.0001 | 1.0000 | 0.0605 | 1.0000 | 0.0001 | 0.1978 | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 1.0000 | 0.9279 |  | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 1.0000 | 0.6487 | 0.0001 |
| Agassiz | 1995 | 1.0000 | 0.0029 | 0.0062 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9825 | 0.0002 | 0.0001 |  | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.8443 | 0.0629 | 1.0000 |
| MainArm | 1995 | 1.0000 | 0.0207 | 0.1235 | 0.0002 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9979 | 0.0072 | 0.0001 | 1.0000 |  | 1.0000 | 1.0000 | 1.0000 | 0.9659 | 0.3280 | 1.0000 |
| NorthArm | 1995 | 1.0000 | 0.0007 | 0.0012 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.8470 | 0.0001 | 0.0001 | 1.0000 | 1.0000 |  | 1.0000 | 1.0000 | 0.5204 | 0.0146 | 1.0000 |
| N.Thompso | 1995 | 1.0000 | 0.0018 | 0.0018 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9850 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | . | 1.0000 | 0.8467 | 0.0406 | 1.0000 |
| n |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Thompson | 1995 | 1.0000 | 0.0002 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9130 | 0.0001 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |  | 0.5915 | 0.0041 | 1.0000 |
| Hansard | 1996 | 0.1485 | 1.0000 | 1.0000 | 1.0000 | 0.9995 | 1.0000 | 0.9875 | 0.7763 | 0.1864 | 0.9996 | 0.9468 | 1.0000 | 1.0000 | 1.0000 | 0.8443 | 0.9659 | 0.5204 | 0.8467 | 0.5915 |  | 1.0000 | 0.9996 |
| Woodpecker | 1996 | 0.0002 | 1.0000 | 1.0000 | 0.9999 | 0.5426 | 1.0000 | 0.1715 | 0.0373 | 0.0003 | 0.8054 | 0.0925 | 1.0000 | 1.0000 | 0.6487 | 0.0629 | 0.3280 | 0.0146 | 0.0406 | 0.0041 | 1.0000 |  | 0.4449 |
| MainArm | 1996 | 0.4087 | 0.0251 | 0.0443 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.5962 | 1.0000 | 1.0000 | 1.0000 | 0.0010 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9996 | 0.4449 |  |

Appendix 10. Mountain whitefish $\log 10$ length ANCOVA table. To test for differences in growth rates.

| Source of Variation | DF | Sum of <br> Squares | Mean <br> Square | F Value | Pr > F |
| :--- | :---: | :---: | :---: | :---: | :---: |
| LAGE | 1 | 0.2111 | 0.2111 | 252.88 | 0.0001 |
| REACH*YEAR | 15 | 0.0435 | 0.0029 | 3.48 | 0.0001 |
| LAGE*REACH*YEAR | 15 | 0.0189 | 0.0013 | 1.51 | 0.0953 |

Appendix 11. Mountain whitefish age-adjusted mean lengths and 95\% confidence limits (mean age 4.10 years, CLM 3.95-4.26).

|  | Year | $\boldsymbol{N}$ | Adjusted <br> Mean <br> Length (mm) | 95\% Lower <br> Confidence <br> Limit | 95\% Upper <br> Confidence <br> Limit |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Neach | 1994 | 54 | 258.3 | 252.9 | 263.7 |
| Nechako | 1995 | 46 | 256.9 | 251.9 | 262.0 |
| Hansard | 1994 | 52 | 241.9 | 237.4 | 246.4 |
| Hansard | 1995 | 48 | 223.9 | 209.1 | 239.7 |
| Hansard | 1996 | 43 | 248.2 | 237.8 | 259.1 |
| Woodpecker | 1994 | 34 | 273.5 | 267.1 | 280.0 |
| Woodpecker | 1995 | 40 | 266.4 | 259.3 | 273.7 |
| Woodpecker | 1996 | 24 | 268.0 | 257.8 | 278.7 |
| Marguerite | 1994 | 24 | 277.6 | 267.9 | 287.6 |
| Marguerite | 1995 | 12 | 277.1 | 266.9 | 287.8 |
| Agassiz | 1994 | 22 | 290.3 | 275.1 | 306.3 |
| Agassiz | 1995 | 27 | 285.4 | 277.5 | 293.4 |
| N.Thompson | 1994 | 39 | 281.4 | 275.3 | 287.7 |
| N.Thompson | 1995 | 34 | 253.6 | 241.5 | 266.4 |
| Thompson | 1994 | 72 | 264.0 | 257.8 | 270.3 |
| Thompson | 1995 | 52 | 288.4 | 279.0 | 298.0 |

Appendix 12. Mountain whitefish - results of multiple comparison test for differences in age-adjusted lengths among reaches in each year, using GT2-method; Pr > $|\mathrm{T}|, \mathrm{HO}: \operatorname{LSMEAN}(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})$.

| Reach i/j | Year | Nechako$1994$ | Hansard$1994$ | Woodpecker$1994$ | Marguerite$1994$ | Agassiz$1994$ | North <br> Thomp- <br> son <br> 1994 | Thompson$1994$ | Nechako$1995$ | Hansard$1995$ | Woodpecker$1995$ | Marguerite 1995 | Agassiz$1995$ | North <br> Thomp- <br> son <br> 1995 | Thomp- <br> son <br> 1995 | Hansard$1996$ | Woodpecker$1996$ |
| :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
|  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Nechako | 1994 |  | 0.0006 | 0.0454 | 0.0739 | 0.0094 | 0.0001 | 1.0000 | 1.0000 | 0.0109 | 0.9999 | 0.1561 | 0.0001 | 1.0000 | 0.0001 | 1.0000 | 1.0000 |
| Hansard | 1994 | 0.0006 |  | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 0.0018 | 0.9762 | 0.0001 | 0.0001 | 0.0001 | 0.9999 | 0.0001 | 1.0000 | 0.0005 |
| Woodpecker | 1994 | 0.0454 | 0.0001 |  | 1.0000 | 0.9958 | 0.9999 | 0.9878 | 0.0086 | 0.0001 | 1.0000 | 1.0000 | 0.9327 | 0.5462 | 0.7060 | 0.0133 | 1.0000 |
| Marguerite | 1994 | 0.0739 | 0.0001 | 1.0000 |  | 1.0000 | 1.0000 | 0.9167 | 0.0240 | 0.0001 | 0.9998 | 1.0000 | 1.0000 | 0.3459 | 1.0000 | 0.0109 | 1.0000 |
| Agassiz | 1994 | 0.0094 | 0.0001 | 0.9958 | 1.0000 |  | 1.0000 | 0.1720 | 0.0039 | 0.0001 | 0.4648 | 1.0000 | 1.0000 | 0.0348 | 1.0000 | 0.0011 | 0.8897 |
| North | 1994 | 0.0001 | 0.0001 | 0.9999 | 1.0000 | 1.0000 |  | 0.0134 | 0.0001 | 0.0001 | 0.2240 | 1.0000 | 1.0000 | 0.0193 | 1.0000 | 0.0001 | 0.9797 |
| Thompson |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Thompson | 1994 | 1.0000 | 0.0001 | 0.9878 | 0.9167 | 0.1720 | 0.0134 |  | 0.9999 | 0.0010 | 1.0000 | 0.9781 | 0.0042 | 1.0000 | 0.0026 | 0.8013 | 1.0000 |
| Nechako | 1995 | 1.0000 | 0.0018 | 0.0086 | 0.0240 | 0.0039 | 0.0001 | 0.9999 |  | 0.0184 | 0.9800 | 0.0595 | 0.0001 | 1.0000 | 0.0001 | 1.0000 | 0.9988 |
| Hansard | 1995 | 0.0109 | 0.9762 | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 0.0010 | 0.0184 |  | 0.0005 | 0.0001 | 0.0001 | 0.3494 | 0.0001 | 0.7544 | 0.0010 |
| Woodpecker | 1995 | 0.9999 | 0.0001 | 1.0000 | 0.9998 | 0.4648 | 0.2240 | 1.0000 | 0.9800 | 0.0005 |  | 1.0000 | 0.0659 | 1.0000 | 0.0333 | 0.5293 | 1.0000 |
| Marguerit | 1995 | 0.1561 | 0.0001 | 1.0000 | 1.0000 | 1.0000 | 1.0000 | 0.9781 | 0.0595 | 0.0001 | 1.0000 |  | 1.0000 | 0.4542 | 1.0000 | 0.0201 | 1.0000 |
| Agassiz | 1995 | 0.0001 | 0.0001 | 0.9327 | 1.0000 | 1.0000 | 1.0000 | 0.0042 | 0.0001 | 0.0001 | 0.0659 | 1.0000 |  | 0.0057 | 1.0000 | 0.0001 | 0.7184 |
| North | 1995 | 1.0000 | 0.9999 | 0.5462 | 0.3459 | 0.0348 | 0.0193 | 1.0000 | 1.0000 | 0.3494 | 1.0000 | 0.4542 | 0.0057 |  | 0.0028 | 1.0000 | 1.0000 |
| Thompson |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| Thompson | 1995 | 0.0001 | 0.0001 | 0.7060 | 1.0000 | 1.0000 | 1.0000 | 0.0026 | 0.0001 | 0.0001 | 0.0333 | 1.0000 | 1.0000 | 0.0028 |  | 0.0001 | 0.4531 |
| Hansard | 1996 | 1.0000 | 1.0000 | 0.0133 | 0.0109 | 0.0011 | 0.0001 | 0.8013 | 1.0000 | 0.7544 | 0.5293 | 0.0201 | 0.0001 | 1.0000 | 0.0001 |  | 0.6758 |
| Woodpecker | 1996 | 1.0000 | 0.0005 | 1.0000 | 1.0000 | 0.8897 | 0.9797 | 1.0000 | 0.9988 | 0.0010 | 1.0000 | 1.0000 | 0.7184 | 1.0000 | 0.4531 | 0.6758 |  |

Appendix 13. Hansard reach comparison among sites -- mountain whitefish log10 length ANCOVA table, ageadjusted ( 6.92 years) mean lengths, and multiple comparison test ( $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H} 0$ : LSMEAN $(\mathrm{i})=\mathrm{LSMEAN}(\mathrm{j})$ ).

| Source of Variation | DF | Sum of Squares | Mean Square | F Value | $\boldsymbol{P r}>\boldsymbol{F}$ |
| :--- | :---: | :---: | :---: | :---: | :---: |
| LAGE | 1 | 0.076071 | 0.076071 | 125.01 | 0.0001 |
| SITE | 2 | 0.005900 | 0.002950 | 4.85 | 0.0090 |
| LAGE*SITE | 2 | 0.002989 | 0.001494 | 2.46 | 0.0891 |
| YEAR | 2 | 0.003158 | 0.001579 | 2.60 | 0.0778 |
| LAGE*YEAR | 2 | 0.003598 | 0.001799 | 2.96 | 0.0549 |
| SITE*YEAR | 1 | 0.000097 | 0.000097 | 0.16 | 0.6899 |
| LAGE*SITE*YEAR | 1 | 0.000154 | 0.000154 | 0.25 | 0.6160 |


|  | YEAR | $\boldsymbol{N}$ | Adjusted <br> Mean <br> Length $(\mathbf{m m})$ | 95\% Lower <br> Confidence <br> Limit | 95\% Upper <br> Confidence <br> Limit |
| :--- | :---: | :---: | :---: | :---: | :---: |
| SITE | 1994 | 2 | 285.0 | 263.2 | 308.6 |
| Hansard | 1994 | 57 | 271.1 | 265.8 | 276.4 |
| Dome Creek | 1995 | 12 | 265.3 | 256.1 | 274.8 |
| Dome Creek | 1996 | 27 | 271.4 | 265.6 | 277.4 |
| Dome Creek | 1995 | 48 | 241.9 | 237.0 | 246.8 |
| McBride | 1996 | 25 | 245.7 | 235.0 | 257.0 |
| McBride |  |  |  |  |  |


| SITE | HEAR | Hansard | Dome Creek | Dome Creek | Dome Creek | McBride | McBride |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| i/j | $\mathbf{1 9 9 4}$ | $\mathbf{1 9 9 4}$ | $\mathbf{1 9 9 5}$ | $\mathbf{1 9 9 6}$ | $\mathbf{1 9 9 5}$ | $\mathbf{1 9 9 6}$ |  |
| Hansard | 1994 | . | 0.9776 | 0.8046 | 0.9835 | 0.0018 | 0.0240 |
| Dome Creek | 1994 | 0.9776 | . | 0.9935 | 1.0000 | 0.0001 | 0.0016 |
| Dome Creek | 1995 | 0.8046 | 0.9935 | . | 0.9907 | 0.0002 | 0.1203 |
| Dome Creek | 1996 | 0.9835 | 1.0000 | 0.9907 |  | 0.0001 | 0.0017 |
| McBride | 1995 | 0.0018 | 0.0001 | 0.0002 | 0.0001 | . | 1.0000 |
| McBride | 1996 | 0.0240 | 0.0016 | 0.1203 | 0.0017 | 1.0000 | . |

Appendix 14. Hansard reach comparison among sites -- peamouth chub log10 length ANCOVA table.

| Source of Variation | $\boldsymbol{D F}$ | Type IV SS | Mean Square | F Value | Pr $>$ F |
| :--- | :---: | :---: | :---: | :---: | :---: |
| LAGE | 1 | 0.061151 | 0.061151 | 72.42 | 0.0001 |
| SITE | 2 | 0.001091 | 0.000545 | 0.65 | 0.5254 |
| LAGE*SITE | 2 | 0.001485 | 0.000742 | 0.88 | 0.4170 |
| YEAR | 1 | 0.000177 | 0.000177 | 0.21 | 0.6475 |
| LAGE*YEAR | 1 | 0.000301 | 0.000301 | 0.36 | 0.5514 |

Appendix 15. Starry flounder log10 length ANOVA table for age 2 years fish.

| Source of Variation | DF | Sum of <br> Squares | Mean <br> Square | F Value | Pr > F |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Model | 3 | 0.013916 | 0.004639 | 5.27 | 0.0017 |
| Error | 166 | 0.146246 | 0.000881 |  |  |
| Corrected Total | 169 | 0.160162 |  |  |  |
| Source of Variation | DF | Type III | Mean <br> Square | F Value | Pr > F |
| REACH |  | SS | Squ |  |  |
| YEAR | 1 | 0.009541 | 0.009541 | 10.83 | 0.0012 |
| REACH*YEAR | 1 | 0.001324 | 0.001324 | 1.50 | 0.2221 |

Appendix 16. Starry flounder mean lengths and 95\% confidence limits for age 2 years.

|  |  |  | Simultaneous <br> 95\% Lower <br> Confidence | Simultaneous <br> 95\% Upper <br> Confidence |
| :--- | :---: | :---: | :---: | :---: |
| Reach | $\boldsymbol{N}$ | Mean | Limit | Limit |
| NorthArm | 83 | 183.4 | 180.3 | 186.5 |
| MainArm | 87 | 176.6 | 173.7 | 179.5 |

Appendix 17. Peamouth chub lipids, condition, and HSI ANOVA tables and multiple comparison tests.
ANOVA Tables

| Source of Variation | DF | Type IV SS | Mean Square | F Value | Pr $>$ F |
| :--- | :---: | ---: | ---: | ---: | ---: |
| Liver Lipid (No Agassiz) |  |  |  |  |  |
| REACH | 9 | 1383.25 | 153.69 | 31.76 | 0.0001 |
| YEAR | 1 | 25.68 | 25.68 | 5.31 | 0.0360 |
| YEAR*REACH | 7 | 54.37 | 7.77 | 1.60 | 0.2088 |
| Muscle Lipid |  |  |  |  |  |
| REACH | 10 | 11.78 | 1.18 | 9.97 | 0.0001 |
| YEAR | 1 | 0.67 | 0.67 | 5.71 | 0.0198 |
| YEAR*REACH | 8 | 8.46 | 1.06 | 8.96 | 0.0001 |
| Mesenteric Fat |  |  |  |  |  |
| REACH | 10 | 267.44 | 26.74 | 29.18 | 0.0001 |
| YEAR | 1 | 74.29 | 74.29 | 81.06 | 0.0001 |
| YEAR*REACH | 8 | 198.03 | 24.75 | 27.01 | 0.0001 |
| Condition |  |  |  |  |  |
| REACH | 10 | 1.225692 | 0.122569 | 29.30 | 0.0001 |
| YEAR | 1 | 0.065904 | 0.065904 | 15.76 | 0.0001 |
| REACH*YEAR | 8 | 0.881708 | 0.110214 | 26.35 | 0.0001 |
| Hepatosomatic Index |  |  |  |  |  |
| REACH | 10 | 235.724619 | 23.572462 | 72.54 | 0.0001 |
| YEAR | 1 | 121.642915 | 121.642915 | 374.35 | 0.0001 |
| REACH*YEAR | 8 | 272.027364 | 34.003421 | 104.64 | 0.0001 |

Peamouth chub liver lipid levels (no Agassiz, Mission, and Barnston) -- multiple comparison test results; $\operatorname{Pr}>$ |T|, H0: LSMEAN(i) =LSMEAN(j).

| REACH | LSMean <br> Lipid (\%) | Nechako | Hansard | Wood- <br> pecker | Margue- <br> rite | Main Arm | North Arm | North <br> Thompson | Thomp-son |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |
| Nechako | 14.3 | . | 1.0000 | 0.9370 | 0.0070 | 0.0031 | 0.0001 | 0.0153 | 0.0150 |
| Hansard | 13.8 | 1.0000 | . | 0.9904 | 0.0115 | 0.0017 | 0.0001 | 0.0283 | 0.0079 |
| Woodpecker | 11.6 | 0.9370 | 0.9904 | . | 0.1606 | 0.0005 | 0.0001 | 0.5156 | 0.0018 |
| Marguerite | 5.3 | 0.0070 | 0.0115 | 0.1606 | . | 0.0001 | 0.0001 | 0.9946 | 0.0001 |
| Main Arm | 22.3 | 0.0031 | 0.0017 | 0.0005 | 0.0001 | . | 0.7361 | 0.0001 | 0.9988 |
| North Arm | 25.2 | 0.0001 | 0.0001 | 0.0001 | 0.0001 | 0.7361 | . | 0.0001 | 0.1299 |
| North Thompson | 7.6 | 0.0153 | 0.0283 | 0.5156 | 0.9946 | 0.0001 | 0.0001 | . | 0.0001 |
| Thompson | 20.7 | 0.0150 | 0.0079 | 0.0018 | 0.0001 | 0.9988 | 0.1299 | 0.0001 | . |
|  | LSMean |  |  |  |  |  |  |  |  |
| Year | Lipid (\%) | 1995 |  |  |  |  |  |  |  |
| 1994 | 14.1 | 0.0360 |  |  |  |  |  |  |  |
| 1995 | 16.0 | . |  |  |  |  |  |  |  |

Appendix 17. Peamouth chub lipids, condition, and HSI ANOVA tables and multiple comparison tests. (continued)
Peamouth chub muscle lipid levels -- multiple comparison test results; $\operatorname{Pr}>|T|, \mathrm{H} 0$ : LSMEAN $(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})$.

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j \& YE \& Nechako

1994 \& ansard
1994 \& Woodpecker

$$
1994
$$ \& \[

$$
\begin{aligned}
& \hline \begin{array}{c}
\text { Margue- } \\
\text { rite }
\end{array} \\
& 1994
\end{aligned}
$$
\] \& Agassiz

1994 \& Mission

1994 \& Barnston

1994 \& \begin{tabular}{l}
Main <br>
Arm <br>
1994

 \& 

North <br>
Arm <br>
1994
\end{tabular} \& North

Thomp-
son

1994 \& | Thomp- |
| :--- |
| son |
| 1994 | \& Nechako

1995 \& 1995 \& Woodpecker

$$
1995
$$ \& \[

$$
\begin{gathered}
\hline \begin{array}{c}
\text { Margue- } \\
\text { rite }
\end{array} \\
1995
\end{gathered}
$$
\] \& Agassiz

1995 \&  \& | North |
| :--- |
| Arm |
| 1995 | \& North Thompson 1995 \& Thompson <br>

\hline Nechako \& 1994 \& \& 0.4148 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.8005 \& 1.0000 \& 1.0000 \& 0.0001 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0987 \& 0.0041 \& 1.0000 \& 1.0000 \& 0.7340 \& 0.2330 <br>
\hline Hansard \& 1994 \& 0.4148 \& \& 1.0000 \& 0.9999 \& 0.2330 \& 0.9998 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.5499 \& 0.0550 \& 0.0550 \& 1.0000 \& 0.9998 \& 0.0001 \& 1.0000 \& 0.2272 \& 0.9656 \& 1.0000 \& . 0000 <br>
\hline Woodpecker \& 1994 \& 1.0000 \& 1.0000 \& \& 1.0000 \& 0.9984 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0005 \& 0.8821 \& 1.0000 \& 1.0000 \& 0.0003 \& 0.5499 \& 0.9981 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline Marguerite \& 1994 \& 1.0000 \& 0.9999 \& 1.0000 \& \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.9945 \& 1.0000 \& 1.0000 \& 0.0013 \& 0.2330 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9969 <br>
\hline Agassiz \& 1994 \& 1.0000 \& 0.2330 \& 0.9984 \& 1.0000 \& \& 1.0000 \& 0.9973 \& 0.5660 \& 0.9998 \& 1.0000 \& 0.0001 \& 1.0000 \& 0.9984 \& 1.0000 \& 0.1995 \& 0.0016 \& 1.0000 \& 1.0000 \& 0.5035 \& 0.1178 <br>
\hline Mission \& 1994 \& 1.0000 \& 0.9998 \& 1.0000 \& 1.0000 \& 1.0000 \& \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.9805 \& 1.0000 \& 1.0000 \& 0.0004 \& 0.1555 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9921 <br>
\hline Barnston \& 1994 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9973 \& 1.0000 \& \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.8393 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.3616 \& 0.9969 \& 1.0000 \& 1.0000 \& 0.9999 <br>
\hline MainArm \& 1994 \& 0.8005 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.5660 \& 1.0000 \& 1.0000 \& \& 1.0000 \& 0.9036 \& 0.0036 \& 0.1746 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.9724 \& 0.5561 \& 0.9999 \& 1.0000 \& 1.0000 <br>
\hline NorthArm \& 1994 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9998 \& 1.0000 \& 1.0000 \& 1.0000 \& \& 1.0000 \& 0.0001 \& 0.9331 \& 1.0000 \& 1.0000 \& 0.0002 \& 0.2430 \& 0.9997 \& 1.0000 \& 1.0000 \& 0.9990 <br>
\hline N.Thompson \& 1994 \& 1.0000 \& 0.5499 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9036 \& 1.0000 \& \& 0.0001 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0624 \& 0.0071 \& 1.0000 \& 1.0000 \& 0.8506 \& 0.3342 <br>
\hline Thompson \& 1994 \& 0.0001 \& 0.0550 \& 0.0005 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0036 \& 0.0001 \& 0.0001 \& \& 0.0001 \& 0.0005 \& 0.0001 \& 0.0001 \& 0.9656 \& 0.0001 \& 0.0001 \& 0.0172 \& 0.1178 <br>
\hline Nechako \& 1995 \& 1.0000 \& 0.0550 \& 0.8821 \& 0.9945 \& 1.0000 \& 0.9805 \& 0.8393 \& 0.1746 \& 0.9331 \& 1.0000 \& 0.0001 \& \& 0.8821 \& 0.9969 \& 0.5874 \& 0.0002 \& 1.0000 \& 1.0000 \& 0.1564 \& 0.0242 <br>
\hline Hansard \& 1995 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9984 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0005 \& 0.8821 \& \& 1.0000 \& 0.0003 \& 0.5499 \& 0.9981 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline Woodpecker \& 1995 \& 1.0000 \& 0.9998 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.9969 \& 1.0000 \& \& 0.0016 \& 0.2048 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9945 <br>
\hline Marguerite \& 1995 \& 0.0987 \& 0.0001 \& 0.0003 \& 0.0013 \& 0.1995 \& 0.0004 \& 0.0001 \& 0.0001 \& 0.0002 \& 0.0624 \& 0.0001 \& 0.5874 \& 0.0003 \& 0.0016 \& \& 0.0001 \& 0.2048 \& 0.0082 \& 0.0001 \& 0.0001 <br>
\hline Agassiz \& 1995 \& 0.0041 \& 1.0000 \& 0.5499 \& 0.2330 \& 0.0016 \& 0.1555 \& 0.3616 \& 0.9724 \& 0.2430 \& 0.0071 \& 0.9656 \& 0.0002 \& 0.5499 \& 0.2048 \& 0.0001 \& \& 0.0016 \& 0.0550 \& 0.9992 \& 1.0000 <br>
\hline MainArm \& 1995 \& 1.0000 \& 0.2272 \& 0.9981 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9969 \& 0.5561 \& 0.9997 \& 1.0000 \& 0.0001 \& 1.0000 \& 0.9981 \& 1.0000 \& 0.2048 \& 0.0016 . \& \& 1.0000 \& 0.4944 \& 0.1144 <br>
\hline NorthArm \& 1995 \& 1.0000 \& 0.9656 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.9999 \& 1.0000 \& 1.0000 \& 0.0001 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0082 \& 0.0550 \& 1.0000 \& \& 0.9992 \& 0.8506 <br>
\hline N.Thompson \& 1995 \& 0.7340 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.5035 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.8506 \& 0.0172 \& 0.1564 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.9992 \& 0.4944 \& 0.9992 \& \& 1.0000 <br>
\hline Thompson \& 1995 \& 0.2330 \& 1.0000 \& 1.0000 \& 0.9969 \& 0.1178 \& 0.9921 \& 0.9999 \& 1.0000 \& 0.9990 \& 0.3342 \& 0.1178 \& 0.0242 \& 1.0000 \& 0.9945 \& 0.0001 \& 1.0000 \& 0.1144 \& 0.8506 \& 1.0000 \& <br>
\hline
\end{tabular}

Appendix 17. Peamouth chub lipids, condition, and HSI ANOVA tables and multiple comparison tests. (continued)
Peamouth chub mesenteric fat index -- multiple comparison test results; $\operatorname{Pr}>|T|, \mathrm{HO}$ : LSMEAN $(\mathrm{i})=\mathrm{LSMEAN}(\mathrm{j})$.

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j \& YEAR \& Nechako

1994 \& Hansard
1994 \& Woodpecker

$$
1994
$$ \& \[

$$
\begin{gathered}
\hline \begin{array}{l}
\text { Margue- } \\
\text { rite }
\end{array} \\
1994
\end{gathered}
$$
\] \& Agassiz

1994 \& Mission

1994 \& Barnston

1994 \& | Main |
| :--- |
| Arm |
| 1994 | \& North Arm 1994 \& North

Thomp-
son
1994 \& Thompson 1994 \& Nechako

1995 \& Hansard

1995 \& Woodpecker

$$
1995
$$ \& \[

$$
\begin{gathered}
\hline \begin{array}{c}
\text { Margue- } \\
\text { rite }
\end{array} \\
1995
\end{gathered}
$$
\] \& Agassiz

1995 \& Main Arm

\[
1995

\] \& | North |
| :--- |
| Arm |
| 1995 | \& North Thompson 1995 \& \[

$$
\begin{gathered}
\hline \text { Thomp- } \\
\text { son } \\
\\
1995
\end{gathered}
$$
\] <br>

\hline Nechako \& 1994 \& \& 0.0577 \& 0.0003 \& 1.0000 \& 0.0001 \& 0.0006 \& 0.0027 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0002 \& 0.8498 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0002 \& 0.0001 <br>
\hline Hansard \& 1994 \& 0.0577 \& \& 0.0001 \& 0.5604 \& 0.9654 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0058 \& 0.0085 \& 0.0025 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 <br>
\hline Woodpecker \& 1994 \& 0.0003 \& 0.0001 \& \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0042 \& 0.0021 \& 0.0174 \& 1.0000 \& 0.0001 \& 0.9849 \& 1.0000 \& 1.0000 \& 0.4984 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline Marguerite \& 1994 \& 1.0000 \& 0.5604 \& 0.0001 \& \& 0.0001 \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0083 \& 0.1521 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 <br>
\hline Agassiz \& 1994 \& 0.0001 \& 0.9654 \& 0.0001 \& 0.0001 \& \& 0.0001 \& 0.0001 \& 0.3693 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 <br>
\hline Mission \& 1994 \& 0.0006 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& \& 1.0000 \& 0.0001 \& 0.0083 \& 0.0042 \& 0.0331 \& 1.0000 \& 0.0001 \& 0.9988 \& 1.0000 \& 1.0000 \& 0.2586 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline Barnston \& 1994 \& 0.0027 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 1.0000 \& \& 0.0001 \& 0.0304 \& 0.0163 \& 0.1053 \& 1.0000 \& 0.0001 \& 1.0000 \& 1.0000 \& 1.0000 \& 0.0747 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline MainArm \& 1994 \& 1.0000 \& 1.0000 \& 0.0001 \& 1.0000 \& 0.3693 \& 0.0001 \& 0.0001 \& \& 0.9742 \& 0.9900 \& 0.8849 \& 0.0001 \& 0.9987 \& 0.0143 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 <br>
\hline NorthArm \& 1994 \& 1.0000 \& 0.0058 \& 0.0042 \& 1.0000 \& 0.0001 \& 0.0083 \& 0.0304 \& 0.9742 \& \& 1.0000 \& 1.0000 \& 0.0017 \& 0.0001 \& 0.9998 \& 0.0019 \& 0.0001 \& 0.0001 \& 0.0002 \& 0.0037 \& 0.0001 <br>
\hline N.Thompson \& 1994 \& 1.0000 \& 0.0085 \& 0.0021 \& 1.0000 \& 0.0001 \& 0.0042 \& 0.0163 \& 0.9900 \& 1.0000 \& \& 1.0000 \& 0.0008 \& 0.0001 \& 0.9975 \& 0.0009 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0018 \& 0.0001 <br>
\hline Thompson \& 1994 \& 1.0000 \& 0.0025 \& 0.0174 \& 1.0000 \& 0.0001 \& 0.0331 \& 0.1053 \& 0.8849 \& 1.0000 \& 1.0000 \& \& 0.0080 \& 0.0001 \& 1.0000 \& 0.0089 \& 0.0002 \& 0.0001 \& 0.0014 \& 0.0157 \& 0.0001 <br>
\hline Nechako \& 1995 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0017 \& 0.0008 \& 0.0080 \& \& 0.0001 \& 0.9334 \& 1.0000 \& 1.0000 \& 0.5806 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline Hansard \& 1995 \& 0.0002 \& 1.0000 \& 0.0001 \& 0.0083 \& 1.0000 \& 0.0001 \& 0.0001 \& 0.9987 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 <br>
\hline Woodpecker \& 1995 \& 0.8498 \& 0.0001 \& 0.9849 \& 0.1521 \& 0.0001 \& 0.9988 \& 1.0000 \& 0.0143 \& 0.9998 \& 0.9975 \& 1.0000 \& 0.9334 \& 0.0001 \& \& 0.9477 \& 0.2059 \& 0.0001 \& 0.5541 \& 0.9841 \& 0.0559 <br>
\hline Marguerite \& 1995 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0019 \& 0.0009 \& 0.0089 \& 1.0000 \& 0.0001 \& 0.9477 \& \& 1.0000 \& 0.5215 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline Agassiz \& 1995 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0002 \& 1.0000 \& 0.0001 \& 0.2059 \& 1.0000 \& \& 0.9997 \& 1.0000 \& 1.0000 \& 1.0000 <br>
\hline MainArm \& 1995 \& 0.0001 \& 0.0001 \& 0.4984 \& 0.0001 \& 0.0001 \& 0.2586 \& 0.0747 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.5806 \& 0.0001 \& 0.0001 \& 0.5215 \& 0.9997 \& \& 0.9541 \& 0.4308 \& 1.0000 <br>
\hline NorthArm \& 1995 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0002 \& 0.0001 \& 0.0014 \& 1.0000 \& 0.0001 \& 0.5541 \& 1.0000 \& 1.0000 \& 0.9541 \& \& 1.0000 \& 1.0000 <br>
\hline N.Thompson \& 1995 \& 0.0002 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0037 \& 0.0018 \& 0.0157 \& 1.0000 \& 0.0001 \& 0.9841 \& 1.0000 \& 1.0000 \& 0.4308 \& 1.0000 \& \& 1.0000 <br>
\hline Thompson \& 1995 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0001 \& 1.0000 \& 1.0000 \& 0.0001 \& 0.0001 \& 0.0001 \& 0.0001 \& 1.0000 \& 0.0001 \& 0.0559 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& 1.0000 \& <br>
\hline
\end{tabular}

Appendix 17. Peamouth chub lipids, condition, and HSI ANOVA tables and multiple comparison tests. (continued)
Peamouth chub condition - multiple comparison test results; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{H} 0$ : LSMEAN(i) $=\mathrm{LSMEAN}(\mathrm{j})$.

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j \& YEAR \& Agassiz

1994 \& Barnston

1994 \& Hansard

1994 \& | Main |
| :--- |
| Arm |
| 1994 | \& Marguerite 1994 \& Mission \& Nechako

1994 \& | North |
| :--- |
| Arm |
| 1994 | \& North

Thomp-
son
1994 \& Thompson 1994 \& Woodpecker 1994 \& Agassiz

1995 \& Hansard

1995 \& | Main |
| :--- |
| Arm $1995$ | \& Marguerite 1995 \& Nechako

1995 \& | North |
| :--- |
| Arm |
| 1995 | \& North

Thomp-
son
1995 \& Thompson 1995 \& Woodpecker

$$
1995
$$ <br>

\hline Agassiz \& 1994 \& \& 0.000 \& 0.000 \& 0.000 \& 0.998 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.820 \& 0.000 \& 0.000 \& 0.107 \& 0.005 \& 0.000 \& 0.011 \& 0.000 \& 0.000 \& 0.000 <br>
\hline Barnston \& 1994 \& 0.000 \& \& 0.349 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.998 \& 0.251 \& 0.000 \& 0.939 \& 1.000 \& 0.001 \& 0.974 \& 1.000 \& 0.013 \& 1.000 \& 0.000 \& 1.000 <br>
\hline Hansard \& 1994 \& 0.000 \& 0.349 \& \& 0.107 \& 0.578 \& 0.417 \& 0.000 \& 0.005 \& 1.000 \& 0.000 \& 0.973 \& 0.000 \& 0.296 \& 1.000 \& 1.000 \& 0.001 \& 1.000 \& 0.175 \& 0.000 \& 1.000 <br>
\hline MainArm \& 1994 \& 0.000 \& 1.000 \& 0.107 \& \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.891 \& 0.657 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.781 \& 1.000 \& 0.003 \& 1.000 \& 0.000 \& 1.000 <br>
\hline Marguerite \& 1994 \& 0.998 \& 0.000 \& 0.578 \& 0.000 \& . \& 0.000 \& 0.000 \& 0.000 \& 0.034 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 1.000 \& 0.920 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 <br>
\hline Mission \& 1994 \& 0.000 \& 1.000 \& 0.417 \& 1.000 \& 0.000 \& . \& 0.000 \& 1.000 \& 0.999 \& 0.298 \& 0.000 \& 0.958 \& 1.000 \& 0.001 \& 0.983 \& 1.000 \& 0.019 \& 1.000 \& 0.000 \& 1.000 <br>
\hline Nechako \& 1994 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& . \& 0.002 \& 0.000 \& 0.974 \& 0.000 \& 0.292 \& 0.000 \& 0.000 \& 0.000 \& 0.016 \& 0.000 \& 0.000 \& 1.000 \& 0.000 <br>
\hline NorthArm \& 1994 \& 0.000 \& 1.000 \& 0.005 \& 1.000 \& 0.000 \& 1.000 \& 0.002 \& \& 0.165 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.175 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.982 <br>
\hline N.Thompson \& 1994 \& 0.000 \& 0.998 \& 1.000 \& 0.891 \& 0.034 \& 0.999 \& 0.000 \& 0.165 \& \& 0.000 \& 0.198 \& 0.001 \& 0.993 \& 0.934 \& 1.000 \& 0.043 \& 1.000 \& 0.963 \& 0.000 \& 1.000 <br>
\hline Thompson \& 1994 \& 0.000 \& 0.251 \& 0.000 \& 0.657 \& 0.000 \& 0.298 \& 0.974 \& 1.000 \& 0.000 \& \& 0.000 \& 1.000 \& 0.447 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.519 \& 0.446 \& 0.004 <br>
\hline Woodpecker \& 1994 \& 0.820 \& 0.000 \& 0.973 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.198 \& 0.000 \& \& 0.000 \& 0.000 \& 1.000 \& 0.999 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.004 <br>
\hline Agassiz \& 1995 \& 0.000 \& 0.939 \& 0.000 \& 1.000 \& 0.000 \& 0.958 \& 0.292 \& 1.000 \& 0.001 \& 1.000 \& 0.000 \& \& 0.992 \& 0.000 \& 0.004 \& 1.000 \& 0.000 \& 0.997 \& 0.035 \& 0.078 <br>
\hline Hansard \& 1995 \& 0.000 \& 1.000 \& 0.296 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.993 \& 0.447 \& 0.000 \& 0.992 \& \& 0.001 \& 0.951 \& 1.000 \& 0.011 \& 1.000 \& 0.000 \& 1.000 <br>
\hline MainArm \& 1995 \& 0.107 \& 0.001 \& 1.000 \& 0.000 \& 1.000 \& 0.001 \& 0.000 \& 0.000 \& 0.934 \& 0.000 \& 1.000 \& 0.000 \& 0.001 \& \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.104 <br>
\hline Marguerite \& 1995 \& 0.005 \& 0.974 \& 1.000 \& 0.781 \& 0.920 \& 0.983 \& 0.000 \& 0.175 \& 1.000 \& 0.000 \& 0.999 \& 0.004 \& 0.951 \& 1.000 \& \& 0.059 \& 1.000 \& 0.881 \& 0.000 \& 1.000 <br>
\hline Nechako \& 1995 \& 0.000 \& 1.000 \& 0.001 \& 1.000 \& 0.000 \& 1.000 \& 0.016 \& 1.000 \& 0.043 \& 1.000 \& 0.000 \& 1.000 \& 1.000 \& 0.000 \& 0.059 \& . \& 0.000 \& 1.000 \& 0.001 \& 0.742 <br>
\hline NorthArm \& 1995 \& 0.011 \& 0.013 \& 1.000 \& 0.003 \& 1.000 \& 0.019 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.011 \& 1.000 \& 1.000 \& 0.000 \& \& 0.005 \& 0.000 \& 0.569 <br>
\hline N.Thompson \& 1995 \& 0.000 \& 1.000 \& 0.175 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 1.000 \& 0.963 \& 0.519 \& 0.000 \& 0.997 \& 1.000 \& 0.000 \& 0.881 \& 1.000 \& 0.005 \& . \& 0.000 \& 1.000 <br>
\hline Thompson \& 1995 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.446 \& 0.000 \& 0.035 \& 0.000 \& 0.000 \& 0.000 \& 0.001 \& 0.000 \& 0.000 \& \& 0.000 <br>
\hline Woodpecker \& 1995 \& 0.000 \& 1.000 \& 1.000 \& 1.000 \& 0.000 \& 1.000 \& 0.000 \& 0.982 \& 1.000 \& 0.004 \& 0.004 \& 0.078 \& 1.000 \& 0.104 \& 1.000 \& 0.742 \& 0.569 \& 1.000 \& 0.000 \& . <br>
\hline
\end{tabular}

Appendix 17. Peamouth chub lipids, condition, and HSI ANOVA tables and multiple comparison tests. (continued)

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline $$
\begin{aligned}
& \hline \text { REACH } \\
& i / j
\end{aligned}
$$ \& YEAR \& Agassiz

1994 \& Barnston

1994 \& Hansard

1994 \& Main Arm 1994 \& Marguerite 1994 \& Mission

1994 \& Nechako

1994 \& North Arm 1994 \& North Thompson 1994 \& Thompson 1994 \& Woodpecker 1994 \& Agassiz

1995 \& Hansard

1995 \& Main Arm 1995 \& Marguerite 1995 \& Nechako

1995 \& | North |
| :--- |
| Arm |
| 1995 | \& North Thompson 1995 \& Thompson 1995 \& Woodpecker 1995 <br>

\hline Agassiz \& 1994 \& \& 1.000 \& 0.997 \& 0.985 \& 1.000 \& 1.000 \& 1.000 \& 0.000 \& 0.999 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 0.992 \& 0.000 \& 0.002 \& 0.000 \& 0.995 <br>
\hline Barnston \& 1994 \& 1.000 \& \& 1.000 \& 1.000 \& 1.000 \& 0.834 \& 1.000 \& 0.159 \& 1.000 \& 0.990 \& 0.978 \& 0.000 \& 0.037 \& 0.269 \& 1.000 \& 1.000 \& 0.000 \& 0.596 \& 0.000 \& 1.000 <br>
\hline Hansard \& 1994 \& 0.997 \& 1.000 \& . \& 1.000 \& 1.000 \& 0.336 \& 1.000 \& 0.637 \& 1.000 \& 0.695 \& 0.615 \& 0.000 \& 0.230 \& 0.810 \& 1.000 \& 1.000 \& 0.000 \& 0.983 \& 0.000 \& 1.000 <br>
\hline MainArm \& 1994 \& 0.985 \& 1.000 \& 1.000 \& \& 1.000 \& 0.238 \& 0.999 \& 0.745 \& 1.000 \& 0.558 \& 0.478 \& 0.000 \& 0.304 \& 0.890 \& 1.000 \& 1.000 \& 0.001 \& 0.995 \& 0.000 \& 1.000 <br>
\hline Marguerite \& 1994 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& . \& 0.883 \& 1.000 \& 0.151 \& 1.000 \& 0.996 \& 0.989 \& 0.000 \& 0.035 \& 0.256 \& 1.000 \& 1.000 \& 0.000 \& 0.573 \& 0.000 \& 1.000 <br>
\hline Mission \& 1994 \& 1.000 \& 0.834 \& 0.336 \& 0.238 \& 0.883 \& \& 1.000 \& 0.000 \& 0.391 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.975 \& 0.277 \& 0.000 \& 0.000 \& 0.000 \& 0.315 <br>
\hline Nechako \& 1994 \& 1.000 \& 1.000 \& 1.000 \& 0.999 \& 1.000 \& 1.000 \& . \& 0.000 \& 1.000 \& 1.000 \& 1.000 \& 0.000 \& 0.000 \& 0.001 \& 1.000 \& 0.999 \& 0.000 \& 0.004 \& 0.000 \& 1.000 <br>
\hline NorthArm \& 1994 \& 0.000 \& 0.159 \& 0.637 \& 0.745 \& 0.151 \& 0.000 \& 0.000 \& \& 0.569 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.929 \& 0.715 \& 1.000 \& 1.000 \& 0.000 \& 0.664 <br>
\hline N.Thompson \& 1994 \& 0.999 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 0.391 \& 1.000 \& 0.569 \& \& 0.758 \& 0.681 \& 0.000 \& 0.193 \& 0.752 \& 1.000 \& 1.000 \& 0.000 \& 0.969 \& 0.000 \& 1.000 <br>
\hline Thompson \& 1994 \& 1.000 \& 0.990 \& 0.695 \& 0.558 \& 0.996 \& 1.000 \& 1.000 \& 0.000 \& 0.758 \& \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 0.616 \& 0.000 \& 0.000 \& 0.000 \& 0.668 <br>
\hline Woodpecker \& 1994 \& 1.000 \& 0.978 \& 0.615 \& 0.478 \& 0.989 \& 1.000 \& 1.000 \& 0.000 \& 0.681 \& 1.000 \& \& 0.000 \& 0.000 \& 0.000 \& 0.999 \& 0.535 \& 0.000 \& 0.000 \& 0.000 \& 0.587 <br>
\hline Agassiz \& 1995 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 <br>
\hline Hansard \& 1995 \& 0.000 \& 0.037 \& 0.230 \& 0.304 \& 0.035 \& 0.000 \& 0.000 \& 1.000 \& 0.193 \& 0.000 \& 0.000 \& 0.000 \& \& 1.000 \& 0.604 \& 0.282 \& 1.000 \& 1.000 \& 0.000 \& 0.247 <br>
\hline MainArm \& 1995 \& 0.000 \& 0.269 \& 0.810 \& 0.890 \& 0.256 \& 0.000 \& 0.001 \& 1.000 \& 0.752 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& \& 0.977 \& 0.869 \& 0.998 \& 1.000 \& 0.000 \& 0.832 <br>
\hline Marguerite \& 1995 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 0.975 \& 1.000 \& 0.929 \& 1.000 \& 1.000 \& 0.999 \& 0.000 \& 0.604 \& 0.977 \& \& 1.000 \& 0.010 \& 0.999 \& 0.000 \& 1.000 <br>
\hline Nechako \& 1995 \& 0.992 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 0.277 \& 0.999 \& 0.715 \& 1.000 \& 0.616 \& 0.535 \& 0.000 \& 0.282 \& 0.869 \& 1.000 \& \& 0.001 \& 0.993 \& 0.000 \& 1.000 <br>
\hline NorthArm \& 1995 \& 0.000 \& 0.000 \& 0.000 \& 0.001 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 0.998 \& 0.010 \& 0.001 \& . \& 0.932 \& 0.009 \& 0.001 <br>
\hline N.Thompson \& 1995 \& 0.002 \& 0.596 \& 0.983 \& 0.995 \& 0.573 \& 0.000 \& 0.004 \& 1.000 \& 0.969 \& 0.000 \& 0.000 \& 0.000 \& 1.000 \& 1.000 \& 0.999 \& 0.993 \& 0.932 \& \& 0.000 \& 0.987 <br>
\hline Thompson \& 1995 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.000 \& 0.009 \& 0.000 \& \& 0.000 <br>
\hline Woodpecker \& 1995 \& 0.995 \& 1.000 \& 1.000 \& 1.000 \& 1.000 \& 0.315 \& 1.000 \& 0.664 \& 1.000 \& 0.668 \& 0.587 \& 0.000 \& 0.247 \& 0.832 \& 1.000 \& 1.000 \& 0.001 \& 0.987 \& 0.000 \& . <br>
\hline
\end{tabular}

Appendix 18. Mountain whitefish lipids, condition, and HSI ANOVA tables and multiple comparison tests.
ANOVA tables

| Source | DF | Type III SS | Mean Square | F Value | Pr > F |
| :--- | :---: | :---: | :---: | :---: | :---: |
| Liver Lipid |  |  |  |  |  |
| REACH | 6 | 2.03 | 0.34 | 3.54 | 0.0380 |
| YEAR | 1 | 1.17 | 1.17 | 12.32 | 0.0056 |
| YEAR*REACH | 6 | 0.68 | 0.11 | 1.20 | 0.3813 |
| Muscle Lipid (No Agassiz) |  |  |  |  |  |
| REACH | 5 | 21.25 | 4.25 | 9.04 | 0.0001 |
| YEAR | 1 | 0.05 | 0.05 | 0.10 | 0.7520 |
| YEAR*REACH | 5 | 2.07 | 0.41 | 0.88 | 0.5047 |
| Mesenteric Fat |  |  |  |  |  |
| REACH | 6 | 272.18 | 45.36 | 39.71 | 0.0001 |
| YEAR | 1 | 23.15 | 23.15 | 20.26 | 0.0001 |
| YEAR*REACH | 6 | 52.37 | 8.73 | 7.64 | 0.0001 |
| Condition |  |  |  |  |  |
| REACH | 6 | 0.879300 | 0.146550 | 25.81 | 0.0001 |
| YEAR | 1 | 0.185614 | 0.185614 | 32.68 | 0.0001 |
| REACH*YEAR | 6 | 0.607465 | 0.101244 | 17.83 | 0.0001 |
| Hepatosomatic Index |  |  |  |  |  |
| REACH | 6 | 35.889633 | 5.981606 | 32.44 | 0.0001 |
| YEAR | 1 | 4.848903 | 4.848903 | 26.30 | 0.0001 |
| REACH*YEAR | 6 | 18.667160 | 3.111193 | 16.87 | 0.0001 |



Appendix 18. Mountain whitefish lipids, condition, and HSI ANOVA tables and multiple comparison tests. (continued)

| Mountain whitefish muscle lipid levels | -- multiple comparison test; $\operatorname{Pr}>\|\mathrm{T}\|, \mathrm{HO}: \mathrm{LSMEAN}(\mathrm{i})$ |
| :--- | :---: | :---: | :---: | :---: | :---: | :---: | :---: | :---: |$=$

Mountain whitefish mesenteric fat index -- multiple comparison test results; $\operatorname{Pr}>|\mathrm{T}|, \mathrm{HO}:$ LSMEAN(i) =LSMEAN(j).

\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline \begin{tabular}{l}
REACH \\
i/j
\end{tabular} \& \& Nechako
1994 \& Hansard
1994 \& Woodpecker 1994 \& Margue-rite

1994 \& Agassiz
1994 \& North
Thomp-son
1994 \& Thomp-son
1994 \& Nechako

1995 \& Hansard

1995 \& Woodpecker 1995 \& Margue-rite
1995 \& Agassiz

1995 \& ```
Thomp-son
1995

``` & Thomp-son

1995 \\
\hline & YEAR & 1994 & 1994 & 1994 & 1994 & 1994 & 1994 & 1994 & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 \\
\hline Nechako & 1994 & & 1.0000 & 0.9908 & 1.0000 & 0.9994 & 0.2903 & 0.0001 & 0.0319 & 0.0085 & 1.0000 & 1.0000 & 0.0001 & 0.0001 & 0.0001 \\
\hline Hansard & 1994 & 1.0000 & & 0.3637 & 1.0000 & 0.7778 & 0.9744 & 0.0001 & 0.4395 & 0.1795 & 0.7833 & 0.9980 & 0.0001 & 0.0060 & 0.0001 \\
\hline Woodpecker & 1994 & 0.9908 & 0.3637 & & 1.0000 & 1.0000 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 1.0000 & 1.0000 & 0.0001 & 0.0001 & 0.0001 \\
\hline Marguerite & 1994 & 1.0000 & 1.0000 & 1.0000 & . & 1.0000 & 0.6468 & 0.0001 & 0.1688 & 0.0676 & 1.0000 & 1.0000 & 0.0001 & 0.0033 & 0.0001 \\
\hline Agassiz & 1994 & 0.9994 & 0.7778 & 1.0000 & 1.0000 & . & 0.0072 & 0.0001 & 0.0007 & 0.0002 & 1.0000 & 1.0000 & 0.0001 & 0.0001 & 0.0001 \\
\hline N.Thompson & 1994 & 0.2903 & 0.9744 & 0.0001 & 0.6468 & 0.0072 & & 0.0001 & 1.0000 & 1.0000 & 0.0009 & 0.1742 & 0.2371 & 0.9889 & 0.0009 \\
\hline Thompson & 1994 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.7093 & 0.0048 & 1.0000 \\
\hline Nechako & 1995 & 0.0319 & 0.4395 & 0.0001 & 0.1688 & 0.0007 & 1.0000 & 0.0001 & & 1.0000 & 0.0001 & 0.0431 & 0.8191 & 1.0000 & 0.0167 \\
\hline Hansard & 1995 & 0.0085 & 0.1795 & 0.0001 & 0.0676 & 0.0002 & 1.0000 & 0.0001 & 1.0000 & & 0.0001 & 0.0197 & 0.9775 & 1.0000 & 0.0590 \\
\hline Woodpecker & 1995 & 1.0000 & 0.7833 & 1.0000 & 1.0000 & 1.0000 & 0.0009 & 0.0001 & 0.0001 & 0.0001 & & 1.0000 & 0.0001 & 0.0001 & 0.0001 \\
\hline Marguerite & 1995 & 1.0000 & 0.9980 & 1.0000 & 1.0000 & 1.0000 & 0.1742 & 0.0001 & 0.0431 & 0.0197 & 1.0000 & & 0.0001 & 0.0018 & 0.0001 \\
\hline Agassiz & 1995 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.2371 & 0.7093 & 0.8191 & 0.9775 & 0.0001 & 0.0001 & & 1.0000 & 1.0000 \\
\hline N.Thompson & 1995 & 0.0001 & 0.0060 & 0.0001 & 0.0033 & 0.0001 & 0.9889 & 0.0048 & 1.0000 & 1.0000 & 0.0001 & 0.0018 & 1.0000 & & 0.7020 \\
\hline Thompson & 1995 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0009 & 1.0000 & 0.0167 & 0.0590 & 0.0001 & 0.0001 & 1.0000 & 0.7020 & . \\
\hline
\end{tabular}

Appendix 18. Mountain whitefish lipids, condition, and HSI ANOVA tables and multiple comparison tests. (continued)
Mountain whitefish condition - multiple comparison test results; \(\operatorname{Pr}>|\mathrm{T}|, \mathrm{HO}\) : LSMEAN(i) =LSMEAN(j).
\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j & YEAR & Agassiz

1994 & Hansard

1994 & \[
\begin{gathered}
\hline \begin{array}{c}
\text { Margue- } \\
\text { rite }
\end{array} \\
1994
\end{gathered}
\] & Nechako

1994 & North Thompson 1994 & \[
\begin{gathered}
\hline \text { Thomp- } \\
\text { son } \\
\\
1994
\end{gathered}
\] & Woodpecker
\[
1994
\] & Agassiz

1995 & Hansard

1995 & \[
\begin{aligned}
& \begin{array}{l}
\text { Margue- } \\
\text { rite }
\end{array} \\
& 1995
\end{aligned}
\] & Nechako

1995 & North Thompson 1995 & \[
\begin{gathered}
\hline \text { Thomp- } \\
\text { son } \\
\\
1995
\end{gathered}
\] & Woodpecker 1995 \\
\hline Agassiz & 1994 & & 1.000 & 1.000 & 0.869 & 0.002 & 0.000 & 0.000 & 0.000 & 0.002 & 0.455 & 0.768 & 0.000 & 0.000 & 0.000 \\
\hline Hansard & 1994 & 1.000 & . & 1.000 & 1.000 & 0.044 & 0.000 & 0.011 & 0.000 & 0.049 & 1.000 & 1.000 & 0.003 & 0.000 & 0.000 \\
\hline Marguerite & 1994 & 1.000 & 1.000 & & 1.000 & 0.536 & 0.000 & 0.275 & 0.000 & 0.564 & 1.000 & 1.000 & 0.126 & 0.001 & 0.000 \\
\hline Nechako & 1994 & 0.869 & 1.000 & 1.000 & . & 0.498 & 0.000 & 0.205 & 0.000 & 0.531 & 1.000 & 1.000 & 0.076 & 0.000 & 0.000 \\
\hline N.Thompson & 1994 & 0.002 & 0.044 & 0.536 & 0.498 & . & 0.000 & 1.000 & 0.000 & 1.000 & 1.000 & 0.666 & 1.000 & 0.836 & 0.002 \\
\hline Thompson & 1994 & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 & . & 0.000 & 1.000 & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 & 0.583 \\
\hline Woodpecker & 1994 & 0.000 & 0.011 & 0.275 & 0.205 & 1.000 & 0.000 & . & 0.000 & 1.000 & 1.000 & 0.320 & 1.000 & 0.978 & 0.005 \\
\hline Agassiz & 1995 & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 & 1.000 & 0.000 & & 0.000 & 0.009 & 0.000 & 0.000 & 0.086 & 1.000 \\
\hline Hansard & 1995 & 0.002 & 0.049 & 0.564 & 0.531 & 1.000 & 0.000 & 1.000 & 0.000 & & 1.000 & 0.698 & 1.000 & 0.809 & 0.001 \\
\hline Marguerite & 1995 & 0.455 & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & 0.009 & 1.000 & & 1.000 & 1.000 & 0.997 & 0.172 \\
\hline Nechako & 1995 & 0.768 & 1.000 & 1.000 & 1.000 & 0.666 & 0.000 & 0.320 & 0.000 & 0.698 & 1.000 & . & 0.129 & 0.000 & 0.000 \\
\hline N.Thompson & 1995 & 0.000 & 0.003 & 0.126 & 0.076 & 1.000 & 0.000 & 1.000 & 0.000 & 1.000 & 1.000 & 0.129 & . & 1.000 & 0.024 \\
\hline Thompson & 1995 & 0.000 & 0.000 & 0.001 & 0.000 & 0.836 & 0.000 & 0.978 & 0.086 & 0.809 & 0.997 & 0.000 & 1.000 & . & 0.983 \\
\hline Woodpecker & 1995 & 0.000 & 0.000 & 0.000 & 0.000 & 0.002 & 0.583 & 0.005 & 1.000 & 0.001 & 0.172 & 0.000 & 0.024 & 0.983 & . \\
\hline
\end{tabular}
\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j & YEAR & Agassiz

1994 & Hansard

1994 & \[
\begin{gathered}
\hline \begin{array}{c}
\text { Margue- } \\
\text { rite }
\end{array} \\
\\
1994
\end{gathered}
\] & Nechako

1994 & North
Thomp-
son
1994 & \[
\begin{gathered}
\text { Thomp- } \\
\text { son } \\
\\
1994
\end{gathered}
\] & Woodpecker
\[
1994
\] & Agassiz

1995 & Hansard

1995 & \[
\begin{aligned}
& \begin{array}{c}
\text { Margue- } \\
\text { rite }
\end{array} \\
& 1995
\end{aligned}
\] & Nechako

1995 & North
Thomp-
son
1995 & \[
\begin{aligned}
& \text { Thomp- } \\
& \text { son } \\
& \\
& 1995
\end{aligned}
\] & Woodpecker
\[
1995
\] \\
\hline Agassiz & 1994 & & 0.169 & 0.975 & 0.853 & 0.069 & 0.011 & 0.943 & 0.000 & 1.000 & 0.903 & 0.137 & 1.000 & 0.939 & 0.002 \\
\hline Hansard & 1994 & 0.169 & . & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 0.575 & 1.000 & 1.000 & 0.093 & 1.000 & 1.000 \\
\hline Marguerite & 1994 & 0.975 & 1.000 & . & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & 1.000 & 1.000 & 0.996 & 1.000 & 0.956 \\
\hline Nechako & 1994 & 0.853 & 1.000 & 1.000 & . & 1.000 & 0.986 & 1.000 & 0.000 & 1.000 & 1.000 & 1.000 & 0.885 & 1.000 & 0.611 \\
\hline N.Thompson & 1994 & 0.069 & 1.000 & 1.000 & 1.000 & . & 1.000 & 1.000 & 0.000 & 0.249 & 1.000 & 1.000 & 0.026 & 1.000 & 1.000 \\
\hline Thompson & 1994 & 0.011 & 1.000 & 1.000 & 0.986 & 1.000 & . & 0.875 & 0.000 & 0.024 & 1.000 & 1.000 & 0.001 & 0.923 & 1.000 \\
\hline Woodpecker & 1994 & 0.943 & 1.000 & 1.000 & 1.000 & 1.000 & 0.875 & . & 0.000 & 1.000 & 1.000 & 1.000 & 0.971 & 1.000 & 0.342 \\
\hline Agassiz & 1995 & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 & . & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 & 0.000 \\
\hline Hansard & 1995 & 1.000 & 0.575 & 1.000 & 1.000 & 0.249 & 0.024 & 1.000 & 0.000 & . & 1.000 & 0.481 & 1.000 & 1.000 & 0.004 \\
\hline Marguerite & 1995 & 0.903 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & . & 1.000 & 0.975 & 1.000 & 1.000 \\
\hline Nechako & 1995 & 0.137 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 0.481 & 1.000 & . & 0.067 & 1.000 & 1.000 \\
\hline N.Thompson & 1995 & 1.000 & 0.093 & 0.996 & 0.885 & 0.026 & 0.001 & 0.971 & 0.000 & 1.000 & 0.975 & 0.067 & . & 0.970 & 0.000 \\
\hline Thompson & 1995 & 0.939 & 1.000 & 1.000 & 1.000 & 1.000 & 0.923 & 1.000 & 0.000 & 1.000 & 1.000 & 1.000 & 0.970 & & 0.417 \\
\hline Woodpecker & 1995 & 0.002 & 1.000 & 0.956 & 0.611 & 1.000 & 1.000 & 0.342 & 0.000 & 0.004 & 1.000 & 1.000 & 0.000 & 0.417 & . \\
\hline
\end{tabular}

Appendix 19. Starry flounder lipid, condition, and HSI ANOVA tables.
\begin{tabular}{|lcrrrr|}
\hline Source & DF & Type III SS & Mean Square & F Value & Pr > F \\
\hline Muscle Lipid & & & & & \\
REACH & 1 & 0.02 & 0.02 & 0.52 & 0.4829 \\
YEAR & 1 & 0.04 & 0.04 & 1.11 & 0.3100 \\
YEAR*REACH & 1 & 0.03 & 0.03 & 0.69 & 0.4195 \\
Condition & & & & & \\
REACH & 1 & 0.003644 & 0.003644 & 0.52 & 0.4716 \\
YEAR & 1 & 0.102538 & 0.102538 & 14.63 & 0.0002 \\
REACH*YEAR & 1 & 0.014930 & 0.014930 & 2.13 & 0.1457 \\
Hepatosomatic Index & & & & & \\
REACH & 1 & 1.094755 & 1.094755 & 15.25 & 0.0001 \\
YEAR & 1 & 1.618140 & 1.618140 & 2.54 & 0.0001 \\
REACH*YEAR & 1 & 0.000020 & 0.000020 & 0.00 & 0.9867 \\
\hline
\end{tabular}

Appendix 20. Peamouth chub (1995) regression of mesenteric fat on log length, arcsine sqrt GSI, reach, and sex.
\begin{tabular}{|lrrrrr|}
\hline Source & DF & \begin{tabular}{r} 
Sum of \\
Squares
\end{tabular} & \begin{tabular}{r} 
Mean \\
Square
\end{tabular} & F Value & Pr > F \\
\hline Log(10) LENGTH & 1 & 3.50 & 3.50 & 4.55 & 0.0334 \\
Arcsine Sqrt GSI & 1 & 0.69 & 0.69 & 0.90 & 0.3442 \\
REACH & 8 & 113.45 & 14.18 & 18.42 & 0.0001 \\
SEX & 1 & 2.15 & 2.15 & 2.79 & 0.0954 \\
\hline
\end{tabular}
\begin{tabular}{|cccc|}
\hline R-Square & C.V. & Root MSE & FAT Mean \\
\hline 0.26 & 26.84 & 0.88 & 3.27 \\
\hline
\end{tabular}

Appendix 21. Mountain whitefish regression of mesenteric fat on log length, arcsine squareroot GSI, reach, year, and sex.
\begin{tabular}{|lrrrrr|}
\hline Source & DF & \begin{tabular}{r} 
Sum of \\
Squares
\end{tabular} & \begin{tabular}{r} 
Mean \\
Square
\end{tabular} & F Value & Pr > F \\
\hline Loo(10) LENGTH & 1 & 1.64 & 1.64 & 1.50 & 0.2218 \\
Arcsine GSI & 1 & 13.41 & 13.41 & 12.24 & 0.0005 \\
YEAR & 2 & 42.05 & 21.03 & 19.18 & 0.0001 \\
REACH & 6 & 242.19 & 40.36 & 36.83 & 0.0001 \\
SEX & 1 & 2.48 & 2.48 & 2.27 & 0.1326 \\
\hline
\end{tabular}
\begin{tabular}{|cccc|}
\hline R-Square & C.V. & Root MSE & FAT Mean \\
\hline 0.31 & 54.57 & 1.05 & 1.92 \\
\hline
\end{tabular}

Appendix 22. ANOVA and multiple comparison tables for comparisons of gonadosomatic indices among reaches and years for peamouth chub and mountain whitefish.
\begin{tabular}{|lccccc|}
\hline Source & DF & Sum of & Mean Square & F Value & Pr \(>\boldsymbol{F}\) \\
\hline Peamouth Chub & & & & & \\
REACH & 10 & 463.01 & 46.30 & 10.21 & 0.0001 \\
YEAR & 1 & 108.22 & 108.22 & 23.85 & 0.0001 \\
REACH*YEAR & 8 & 120.50 & 15.06 & 3.32 & 0.0009 \\
Mountain Whitefish & & & & & \\
REACH & 6 & 6898.38 & 1149.73 & 33.96 & 0.0001 \\
YEAR & 1 & 14.42 & 14.42 & 0.43 & 0.5142 \\
REACH*YEAR & 6 & 1634.62 & 272.44 & 8.05 & 0.0001 \\
\hline
\end{tabular}

Appendix 22. ANOVA and multiple comparison tables for comparisons of gonadosomatic indices among reaches and years for peamouth chub and mountain whitefish. Continued

Peamouth chub gonadosomatic index -- multiple comparison test results; \(\operatorname{Pr}>|\mathrm{T}|, \mathrm{H} 0\) : LSMEAN(i) =LSMEAN(j).
\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline REACH i/j & YEAR & Nechako

1994 & Hansard

1994 & Agassiz

1994 & Mission

1994 & Barnston

1994 & Main Arm
\[
1994
\] & Marguerite 1994 & \begin{tabular}{l}
North \\
Arm \\
1994
\end{tabular} & North Thompson 1994 & \begin{tabular}{l}
Thomp- \\
son \\
1994
\end{tabular} & Woodpecker
\[
1994
\] & \begin{tabular}{l}
Nechak \\
0 \\
1995
\end{tabular} & Hansard

1995 & Agassiz

1995 & \begin{tabular}{l}
Main \\
Arm \\
1995
\end{tabular} & Marguerite
\[
1995
\] & North Arm
\[
1995
\] & North Thompson 1995 & \begin{tabular}{l}
Thomp- \\
son \\
1995
\end{tabular} & Woodpecker
\[
1995
\] \\
\hline Nechako & 1994 & & 1.000 & 1.000 & 0.001 & 0.005 & 0.013 & 0.918 & 0.002 & 1.000 & 0.905 & 0.053 & 0.008 & 0.545 & 1.000 & 0.000 & 0.131 & 0.000 & 0.986 & 0.004 & 0.006 \\
\hline Hansard & 1994 & 1.000 & & 0.895 & 0.000 & 0.000 & 0.000 & 0.002 & 0.000 & 0.431 & 0.002 & 0.000 & 0.000 & 0.000 & 1.000 & 0.000 & 0.000 & 0.000 & 0.006 & 0.000 & 0.000 \\
\hline Woodpecker & 1994 & 0.053 & 0.000 & 0.374 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.853 & 1.000 & & 1.000 & 1.000 & 0.006 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 \\
\hline Marguerite & 1994 & 0.918 & 0.002 & 1.000 & 0.999 & 1.000 & 1.000 & & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.370 & 0.320 & 1.000 & 0.957 & 1.000 & 1.000 & 1.000 \\
\hline Agassiz & 1994 & 1.000 & 0.895 & & 0.016 & 0.062 & 0.134 & 1.000 & 0.022 & 1.000 & 1.000 & 0.374 & 0.084 & 0.986 & 1.000 & 0.000 & 0.518 & 0.004 & 1.000 & 0.046 & 0.066 \\
\hline Mission & 1994 & 0.001 & 0.000 & 0.016 & . & 1.000 & 1.000 & 0.999 & 1.000 & 0.092 & 0.999 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & 1.000 & 1.000 & 0.986 & 1.000 & 1.000 \\
\hline Barnston & 1994 & 0.005 & 0.000 & 0.062 & 1.000 & & 1.000 & 1.000 & 1.000 & 0.285 & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 \\
\hline MainArm & 1994 & 0.013 & 0.000 & 0.134 & 1.000 & 1.000 & & 1.000 & 1.000 & 0.496 & 1.000 & 1.000 & 1.000 & 1.000 & 0.001 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 \\
\hline NorthArm & 1994 & 0.002 & 0.000 & 0.022 & 1.000 & 1.000 & 1.000 & 1.000 & & 0.125 & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & 1.000 & 1.000 & 0.997 & 1.000 & 1.000 \\
\hline N.Thompson & 1994 & 1.000 & 0.431 & 1.000 & 0.092 & 0.285 & 0.496 & 1.000 & 0.125 & & 1.000 & 0.853 & 0.356 & 1.000 & 1.000 & 0.001 & 0.893 & 0.028 & 1.000 & 0.224 & 0.294 \\
\hline Thompson & 1994 & 0.905 & 0.002 & 1.000 & 0.999 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & & 1.000 & 1.000 & 1.000 & 0.349 & 0.339 & 1.000 & 0.965 & 1.000 & 1.000 & 1.000 \\
\hline Nechako & 1995 & 0.008 & 0.000 & 0.084 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.356 & 1.000 & 1.000 & & 1.000 & 0.001 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 \\
\hline Hansard & 1995 & 0.545 & 0.000 & 0.986 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & & 0.116 & 0.825 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 \\
\hline Woodpecker & 1995 & 0.006 & 0.000 & 0.066 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.294 & 1.000 & 1.000 & 1.000 & 1.000 & 0.001 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & \\
\hline Marguerite & 1995 & 0.131 & 0.000 & 0.518 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.893 & 1.000 & 1.000 & 1.000 & 1.000 & 0.025 & 1.000 & & 1.000 & 1.000 & 1.000 & 1.000 \\
\hline Agassiz & 1995 & 1.000 & 1.000 & 1.000 & 0.000 & 0.000 & 0.001 & 0.370 & 0.000 & 1.000 & 0.349 & 0.006 & 0.001 & 0.116 & & 0.000 & 0.025 & 0.000 & 0.578 & 0.000 & 0.001 \\
\hline MainArm & 1995 & 0.000 & 0.000 & 0.000 & 1.000 & 1.000 & 1.000 & 0.320 & 1.000 & 0.001 & 0.339 & 1.000 & 1.000 & 0.825 & 0.000 & & 1.000 & 1.000 & 0.179 & 1.000 & 1.000 \\
\hline NorthArm & 1995 & 0.000 & 0.000 & 0.004 & 1.000 & 1.000 & 1.000 & 0.957 & 1.000 & 0.028 & 0.965 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & 1.000 & & 0.839 & 1.000 & 1.000 \\
\hline N.Thompson & 1995 & 0.986 & 0.006 & 1.000 & 0.986 & 1.000 & 1.000 & 1.000 & 0.997 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.578 & 0.179 & 1.000 & 0.839 & & 1.000 & 1.000 \\
\hline Thompson & 1995 & 0.004 & 0.000 & 0.046 & 1.000 & 1.000 & 1.000 & 1.000 & 1.000 & 0.224 & 1.000 & 1.000 & 1.000 & 1.000 & 0.000 & 1.000 & 1.000 & 1.000 & 1.000 & . & 1.000 \\
\hline
\end{tabular}

Mountain whitefish gonadosomatic index -- multiple comparison test results; Pr > |T|, H0: LSMEAN(i) =LSMEAN(j).
\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline \[
\overline{\mathrm{REACH}}
\]
\[
\mathrm{i} / \mathrm{j}
\] & & Nechako & Hansard & Agassiz & Marguerite & North Thompson & Thompson & Woodpecker & Nechako & Hansard & Agassiz & Marguerite & North Thompson & Thompson & Woodpecker \\
\hline & YEAR & 1994 & 1994 & 1994 & 1994 & 1994 & 1994 & 1994 & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 \\
\hline Nechako & 1994 & & 0.771 & 1.000 & 0.000 & 0.003 & 0.000 & 0.996 & 0.000 & 0.142 & 0.968 & 0.046 & 1.000 & 0.075 & 1.000 \\
\hline Hansard & 1994 & 0.771 & & 1.000 & 0.000 & 0.000 & 0.004 & 0.002 & 0.838 & 1.000 & 0.002 & 0.983 & 0.022 & 1.000 & 0.963 \\
\hline Woodpecker & 1994 & 0.996 & 0.002 & 0.998 & 0.000 & 0.823 & 0.000 & & 0.000 & 0.000 & 1.000 & 0.000 & 1.000 & 0.000 & 0.919 \\
\hline Marguerite & 1994 & 0.000 & 0.000 & 0.000 & & 0.000 & 0.994 & 0.000 & 0.164 & 0.001 & 0.000 & 1.000 & 0.000 & 0.002 & 0.000 \\
\hline Agassiz & 1994 & 1.000 & 1.000 & & 0.000 & 0.034 & 0.001 & 0.998 & 0.102 & 0.976 & 0.984 & 0.348 & 1.000 & 0.924 & 1.000 \\
\hline N.Thompson & 1994 & 0.003 & 0.000 & 0.034 & 0.000 & & 0.000 & 0.823 & 0.000 & 0.000 & 0.999 & 0.000 & 0.309 & 0.000 & 0.001 \\
\hline Thompson & 1994 & 0.000 & 0.004 & 0.001 & 0.994 & 0.000 & & 0.000 & 1.000 & 0.083 & 0.000 & 1.000 & 0.000 & 0.162 & 0.000 \\
\hline Nechako & 1995 & 0.000 & 0.838 & 0.102 & 0.164 & 0.000 & 1.000 & 0.000 & & 1.000 & 0.000 & 1.000 & 0.000 & 1.000 & 0.001 \\
\hline Hansard & 1995 & 0.142 & 1.000 & 0.976 & 0.001 & 0.000 & 0.083 & 0.000 & 1.000 & & 0.000 & 1.000 & 0.001 & 1.000 & 0.344 \\
\hline Woodpecker & 1995 & 1.000 & 0.963 & 1.000 & 0.000 & 0.001 & 0.000 & 0.919 & 0.001 & 0.344 & 0.806 & 0.089 & 1.000 & 0.202 & \\
\hline Marguerite & 1995 & 0.046 & 0.983 & 0.348 & 1.000 & 0.000 & 1.000 & 0.000 & 1.000 & 1.000 & 0.000 & & 0.002 & 1.000 & 0.089 \\
\hline Agassiz & 1995 & 0.968 & 0.002 & 0.984 & 0.000 & 0.999 & 0.000 & 1.000 & 0.000 & 0.000 & & 0.000 & 1.000 & 0.000 & 0.806 \\
\hline N.Thompson & 1995 & 1.000 & 0.022 & 1.000 & 0.000 & 0.309 & 0.000 & 1.000 & 0.000 & 0.001 & 1.000 & 0.002 & & 0.000 & 1.000 \\
\hline Thompson & 1995 & 0.075 & 1.000 & 0.924 & 0.002 & 0.000 & 0.162 & 0.000 & 1.000 & 1.000 & 0.000 & 1.000 & 0.000 & . & 0.202 \\
\hline
\end{tabular}

Appendix 23. Peamouth chub and mountain whitefish log weight-adjusted ANCOVA, mean fecundity estimates, and \(95 \%\) confidence limits (mean weights 132 g (length 226 mm ) and 225 g ( 272 mm ), respectively).
\begin{tabular}{|lccccc|}
\hline Source & DF & \begin{tabular}{c} 
Sum of \\
Squares
\end{tabular} & \begin{tabular}{c} 
Mean \\
Square
\end{tabular} & F Value & Pr > F \\
\hline Peamouth Chub & & & & & \\
Log weight & 1 & 0.1856 & 0.1856 & 52.80 & 0.0001 \\
Reach & 2 & 0.0197 & 0.0098 & 2.80 & 0.0780 \\
Log weight*Reach & 2 & 0.0226 & 0.0113 & 3.22 & 0.0553 \\
Mountain Whitefish & & & & & \\
Log weight & 1 & 0.3137 & 0.3137 & 39.63 & 0.0001 \\
Reach & 1 & 0.0026 & 0.0026 & 0.32 & 0.5740 \\
Log weight*Reach & 1 & 0.0033 & 0.0033 & 0.42 & 0.5214 \\
\hline
\end{tabular}
\begin{tabular}{|lrcccc|}
\hline & & \begin{tabular}{c} 
LS Mean*
\end{tabular} \\
Reach & N & \begin{tabular}{c} 
95\% Lower \\
Confidence \\
Fenndity
\end{tabular} & \begin{tabular}{c} 
95\% Upper \\
Confidence \\
Limit
\end{tabular} & Limit & Group \\
\hline Peamouth Chub & & & & & \\
Hansard & 16 & 7391 & 6830 & 7997 & A \\
Woodpecker & 5 & 7226 & 6332 & 8246 & A \\
MainArm & 13 & 10752 & 9472 & 12206 & B \\
Mountain Whitefish & & & & A \\
Hansard & 23 & 3089 & 2831 & 3372 & A \\
Woodpecker & 14 & 2716 & 2424 & 3044 & \\
\hline
\end{tabular}

Appendix 24. Peamouth chub and mountain whitefish ANOVA, mean egg diameters, and 95\% confidence limits.
\begin{tabular}{|lccccc|}
\hline Source & DF & \begin{tabular}{l} 
Sum of \\
Squares
\end{tabular} & \begin{tabular}{l} 
Mean \\
Square
\end{tabular} & F Value & Pr \(>\boldsymbol{F}\) \\
\hline \begin{tabular}{l} 
Peamouth Chub \\
REACH
\end{tabular} & 2 & 235001 & 117501 & 14.34 & 0.0001 \\
\begin{tabular}{l} 
Mountain Whitefish \\
REACH
\end{tabular} & 1 & 363926 & 363926 & 25.87 & 0.0001 \\
\hline
\end{tabular}
\begin{tabular}{|c|c|c|c|c|c|}
\hline Reach & \(N\) & LS Mean Diameter \(\mu m\) & 95\% Lower Confidence Limit & 95\% Upper Confidence Limit & Group \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline Hansard & 16 & 1129 & 1072 & 1186 & A \\
\hline Woodpecker & 5 & 1090 & 988 & 1192 & A \\
\hline MainArm & 13 & 951 & 888 & 1014 & B \\
\hline \multicolumn{6}{|l|}{Mountain Whitefish} \\
\hline Hansard & 23 & 2326 & 2268 & 2383 & A \\
\hline Woodpecker & 14 & 2530 & 2456 & 2604 & B \\
\hline
\end{tabular}

Appendix 25. Results of the ANOVA to compare duplicate assessments done in 1996.
\begin{tabular}{|c|c|c|c|c|c|}
\hline Source & DF & Type III SS & Mean Square & \(F\) Value & Pr>F \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Fins Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 0.2412 & 0.2412 & 0.11 & 0.7428 \\
\hline ASSESSOR & 1 & 0.0006 & 0.0006 & 0.00 & 0.9875 \\
\hline ASSESSOR*REACH & 1 & 0.0006 & 0.0006 & 0.00 & 0.9875 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 28.2489 & 9.4163 & 1.48 & 0.2191 \\
\hline ASSESSOR & 1 & 0.5586 & 0.5586 & 0.09 & 0.7671 \\
\hline ASSESSOR*REACH & 3 & 8.9263 & 2.9754 & 0.47 & 0.7049 \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{\begin{tabular}{|c|}
\hline Skin \\
Mountain Whitefish
\end{tabular}}} \\
\hline & & & & & \\
\hline REACH & 1 & 40.1721 & 40.1721 & 3.20 & 0.0753 \\
\hline ASSESSOR & 1 & 1.2247 & 1.2247 & 0.10 & 0.7551 \\
\hline ASSESSOR*REACH & 1 & 5.6691 & 5.6691 & 0.45 & 0.5024 \\
\hline \multicolumn{6}{|l|}{Peamouth chub} \\
\hline REACH & 3 & 30.1404 & 10.0468 & 1.78 & 0.1500 \\
\hline ASSESSOR & 1 & 20.9561 & 20.9561 & 3.71 & 0.0546 \\
\hline ASSESSOR*REACH & 3 & 32.2476 & 10.7492 & 1.91 & 0.1279 \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Gills
Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 48.3408 & 48.3408 & 0.40 & 0.5304 \\
\hline ASSESSOR & 1 & 944.2150 & 944.2150 & 7.72 & 0.0061 \\
\hline ASSESSOR*REACH & 1 & 12.5702 & 12.5702 & 0.10 & 0.7489 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 1125.7557 & 375.2519 & 7.18 & 0.0001 \\
\hline ASSESSOR & 1 & 212.8726 & 212.8726 & 4.07 & 0.0442 \\
\hline ASSESSOR*REACH & 3 & 151.6909 & 50.5636 & 0.97 & 0.4078 \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Parasite \(\begin{array}{r}\text { Pr } \\ \hline \text { Mountain Whitefish }\end{array}\)}} \\
\hline & & & & & \\
\hline REACH & 1 & 31.7409 & 31.7409 & 1.60 & 0.2070 \\
\hline ASSESSOR & 1 & 3.0409 & 3.0409 & 0.15 & 0.6955 \\
\hline ASSESSOR*REACH & 1 & 3.0409 & 3.0409 & 0.15 & 0.6955 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 82.9929 & 27.6643 & 2.74 & 0.0427 \\
\hline ASSESSOR & 1 & 15.1036 & 15.1036 & 1.50 & 0.2216 \\
\hline ASSESSOR*REACH & 3 & 2.4994 & 0.8331 & 0.08 & 0.9695 \\
\hline
\end{tabular}

Appendix 25. Results of the ANOVA to compare duplicate assessments done in 1996 (continued)
\begin{tabular}{|c|c|c|c|c|c|}
\hline Source & DF & Type III SS & Mean Square & \(F\) Value & Pr \(>\) F \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Bile
Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 13.7069 & 13.7069 & 15.60 & 0.0001 \\
\hline ASSESSOR & 1 & 0.0281 & 0.0281 & 0.03 & 0.8583 \\
\hline ASSESSOR*REACH & 1 & 0.1170 & 0.1170 & 0.13 & 0.7156 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 48.5249 & 16.1750 & 21.26 & 0.0001 \\
\hline ASSESSOR & 1 & 0.5205 & 0.5205 & 0.68 & 0.4086 \\
\hline ASSESSOR*REACH & 3 & 0.9165 & 0.3055 & 0.40 & 0.7520 \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Liver
Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 9.7267 & 9.7267 & 0.49 & 0.4846 \\
\hline ASSESSOR & 1 & 0.4960 & 0.4960 & 0.03 & 0.8745 \\
\hline ASSESSOR*REACH & 1 & 20.4960 & 20.4960 & 1.03 & 0.3107 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 20540.3401 & 6846.7800 & 80.97 & 0.0001 \\
\hline ASSESSOR & 1 & 41.2777 & 41.2777 & 0.49 & 0.4851 \\
\hline ASSESSOR*REACH & 3 & 337.4715 & 112.4905 & 1.33 & 0.2639 \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Fat
Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 0.8443 & 0.8443 & 1.60 & 0.2081 \\
\hline ASSESSOR & 1 & 1.0393 & 1.0393 & 1.96 & 0.1628 \\
\hline ASSESSOR*REACH & 1 & 5.2615 & 5.2615 & 9.95 & 0.0019 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 11.8619 & 3.9540 & 5.69 & 0.0008 \\
\hline ASSESSOR & 1 & 1.3104 & 1.3104 & 1.89 & 0.1702 \\
\hline ASSESSOR*REACH & 3 & 3.6416 & 1.2139 & 1.75 & 0.1564 \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Spleen
Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 463.5223 & 463.5223 & 6.52 & 0.0115 \\
\hline ASSESSOR & 1 & 58.4615 & 58.4615 & 0.82 & 0.3658 \\
\hline ASSESSOR*REACH & 1 & 58.4615 & 58.4615 & 0.82 & 0.3658 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 61.5938 & 20.5313 & 1.18 & 0.3176 \\
\hline ASSESSOR & 1 & 18.9227 & 18.9227 & 1.09 & 0.2980 \\
\hline ASSESSOR*REACH & 3 & 53.6991 & 17.8997 & 1.03 & 0.3803 \\
\hline
\end{tabular}

Appendix 25. Results of the ANOVA to compare duplicate assessments done in 1996. (continued)
\begin{tabular}{|c|c|c|c|c|c|}
\hline Source & DF & Type III SS & Mean Square & \(F\) Value & Pr \(>\) F \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{Hindgut
Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 21.4170 & 21.4170 & 3.65 & 0.0576 \\
\hline ASSESSOR & 1 & 11.2461 & 11.2461 & 1.92 & 0.1678 \\
\hline ASSESSOR*REACH & 1 & 46.8016 & 46.8016 & 7.98 & 0.0053 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 1258.9408 & 419.6469 & 30.37 & 0.0001 \\
\hline ASSESSOR & 1 & 1.1666 & 1.1666 & 0.08 & 0.7715 \\
\hline ASSESSOR*REACH & 3 & 19.2557 & 6.4186 & 0.46 & 0.7071 \\
\hline \multicolumn{6}{|l|}{\begin{tabular}{l}
Kidney
Mountain Whitefish \\
Mountain Whitefish
\end{tabular}} \\
\hline REACH & 1 & 1836.8016 & 1836.8016 & 10.51 & 0.0014 \\
\hline ASSESSOR & 1 & 171.0526 & 171.0526 & 0.98 & 0.3239 \\
\hline ASSESSOR*REACH & 1 & 171.0526 & 171.0526 & 0.98 & 0.3239 \\
\hline Peamouth Chub & \multicolumn{5}{|l|}{All values=30} \\
\hline \multicolumn{6}{|l|}{\multirow[t]{2}{*}{HAI
Mountain Whitefish}} \\
\hline & & & & & \\
\hline REACH & 1 & 151.9412 & 151.9412 & 0.33 & 0.5641 \\
\hline ASSESSOR & 1 & 481.6744 & 481.6744 & 1.06 & 0.3049 \\
\hline ASSESSOR*REACH & 1 & 1831.1380 & 1831.1380 & 4.03 & 0.0464 \\
\hline \multicolumn{6}{|l|}{Peamouth Chub} \\
\hline REACH & 3 & 16712.5925 & 5570.8642 & 27.73 & 0.0001 \\
\hline ASSESSOR & 1 & 610.9875 & 610.9875 & 3.04 & 0.0818 \\
\hline ASSESSOR*REACH & 3 & 1092.7397 & 364.2466 & 1.81 & 0.1439 \\
\hline
\end{tabular}

Appendix 26. Peamouth chub, mountain whitefish, and starry flounder HAI ANOVA and multiple comparison test tables.
\begin{tabular}{|lccccc|}
\hline Source & DF & Type IV SS & Mean Square & F Value & Pr > F \\
\hline Peamouth Chub & & & & & \\
REACH & 11 & 67819 & 6165 & 22.71 & 0.0001 \\
YEAR & 3 & 42975 & 14325 & 52.76 & 0.0001 \\
YEAR*REACH & 13 & 27062 & 2082 & 7.67 & 0.0001 \\
Mountain Whitefish & & & & & \\
REACH & 6 & 52983 & 8830 & 21.91 & 0.0001 \\
YEAR & 3 & 20835 & 6945 & 17.23 & 0.0001 \\
YEAR*REACH & 8 & 16747 & 2093 & 5.19 & 0.0001 \\
Starry Flounder & & & & & \\
REACH & 1 & 4192 & 4192 & 12.55 & 0.0005 \\
YEAR & 1 & 647 & 647 & 1.94 & 0.1650 \\
YEAR*REACH & 1 & 6199 & 6199 & 18.56 & 0.0001 \\
\hline
\end{tabular}

Peamouth Chub - Multiple comparison table; \(\operatorname{Pr}>|T|, \mathrm{H} 0\) : LSMEAN(i) \(=\) LSMEAN(j).
\begin{tabular}{lcccccccccccccc}
\hline REACH & & Nechako & Hansard \\
i/j & & \begin{tabular}{c} 
Wood- \\
pecker
\end{tabular} & \begin{tabular}{c} 
Marg- \\
uerite
\end{tabular} & Agassiz & & Mission & Barnston Main Arm & North & \begin{tabular}{c} 
North \\
Thomp-
\end{tabular} & \begin{tabular}{c} 
Thomp- \\
son
\end{tabular} \\
& & & & & & & & & & & & & & \\
Arm
\end{tabular}

Appendix 26. Peamouth chub, mountain whitefish, and starry flounder HAI ANOVA and multiple comparison test tables. (continued)
\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|}
\hline \multirow[t]{2}{*}{\[
\begin{gathered}
\text { REACH } \\
\mathrm{i} / \mathrm{j}
\end{gathered}
\]} & & Nechako & Hansard & Wood-pecker & Marg-uerite & Agassiz & Main Arm & North Arm & North Thomp-son & Thomp-son \\
\hline & YEAR & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 & 1995 \\
\hline Nechako & 1994 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 \\
\hline Hansard & 1994 & 0.0001 & 0.0001 & 0.2437 & 0.6033 & 0.0001 & 0.0771 & 0.0001 & 0.0001 & 0.0001 \\
\hline Woodpecker & 1994 & 0.0001 & 0.0001 & 0.0001 & 0.0007 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 \\
\hline Marguerite & 1994 & 0.0001 & 0.0001 & 0.9998 & 1.0000 & 0.0002 & 0.9674 & 0.0010 & 0.0001 & 0.0001 \\
\hline Agassiz & 1994 & 0.0001 & 0.0001 & 0.9819 & 0.9990 & 0.0001 & 0.7752 & 0.0003 & 0.0001 & 0.0001 \\
\hline Mission & 1994 & 1.0000 & 1.0000 & 0.1078 & 0.9248 & 1.0000 & 0.3164 & 1.0000 & 1.0000 & 0.9797 \\
\hline Barnston & 1994 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.0052 \\
\hline MainArm & 1994 & 0.0135 & 0.0004 & 1.0000 & 1.0000 & 0.0192 & 1.0000 & 0.0815 & 0.0014 & 0.0001 \\
\hline NorthArm & 1994 & 1.0000 & 1.0000 & 0.9993 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.0264 \\
\hline N.Thompson & 1994 & 0.7145 & 0.0744 & 1.0000 & 1.0000 & 0.8085 & 1.0000 & 0.9907 & 0.2043 & 0.0001 \\
\hline Thompson & 1994 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.0017 \\
\hline Nechako & 1995 & . & 1.0000 & 0.7752 & 1.0000 & 1.0000 & 0.9819 & 1.0000 & 1.0000 & 0.2437 \\
\hline Hansard & 1995 & 1.0000 & & 0.0912 & 0.8974 & 1.0000 & 0.2750 & 1.0000 & 1.0000 & 0.9917 \\
\hline Woodpecker & 1995 & 0.7752 & 0.0912 & & 1.0000 & 0.8593 & 1.0000 & 0.9957 & 0.2437 & 0.0001 \\
\hline Marguerite & 1995 & 1.0000 & 0.8974 & 1.0000 & & 1.0000 & 1.0000 & 1.0000 & 0.9913 & 0.0005 \\
\hline Agassiz & 1995 & 1.0000 & 1.0000 & 0.8593 & 1.0000 & . & 0.9942 & 1.0000 & 1.0000 & 0.1681 \\
\hline MainArm & 1995 & 0.9819 & 0.2750 & 1.0000 & 1.0000 & 0.9942 & . & 1.0000 & 0.5837 & 0.0001 \\
\hline NorthArm & 1995 & 1.0000 & 1.0000 & 0.9957 & 1.0000 & 1.0000 & 1.0000 & . & 1.0000 & 0.0504 \\
\hline N.Thompson & 1995 & 1.0000 & 1.0000 & 0.2437 & 0.9913 & 1.0000 & 0.5837 & 1.0000 & & 0.7752 \\
\hline Thompson & 1995 & 0.2437 & 0.9917 & 0.0001 & 0.0005 & 0.1681 & 0.0001 & 0.0504 & 0.7752 & . \\
\hline Hansard & 1996.1 & 0.6499 & 1.0000 & 0.0001 & 0.0025 & 0.5157 & 0.0001 & 0.2043 & 0.9907 & 1.0000 \\
\hline Woodpecker & 1996.1 & 0.3095 & 0.0190 & 1.0000 & 1.0000 & 0.3923 & 1.0000 & 0.7898 & 0.0577 & 0.0001 \\
\hline MainArm & 1996.1 & 1.0000 & 1.0000 & 0.0079 & 0.3959 & 1.0000 & 0.0324 & 1.0000 & 1.0000 & 1.0000 \\
\hline MainArm Sep & 1996.1 & 1.0000 & 1.0000 & 0.1596 & 0.9584 & 1.0000 & 0.4221 & 1.0000 & 1.0000 & 0.9806 \\
\hline Hansard & 1996.2 & 0.9957 & 1.0000 & 0.0001 & 0.0184 & 0.9820 & 0.0003 & 0.7752 & 1.0000 & 1.0000 \\
\hline Woodpecker & 1996.2 & 0.9956 & 0.3971 & 1.0000 & 1.0000 & 0.9990 & 1.0000 & 1.0000 & 0.7296 & 0.0001 \\
\hline MainArm & 1996.2 & 1.0000 & 1.0000 & 0.9950 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.1225 \\
\hline MainArm Sep & 1996.2 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.0108 \\
\hline
\end{tabular}
\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|}
\hline \multirow[t]{2}{*}{\[
\begin{gathered}
\text { REACH } \\
\text { i/j }
\end{gathered}
\]} & & Hansard & Wood-pecker & Main Arm & Main Arm Sep & Hansard & Wood pecker & Main Arm & Main Arm Sep \\
\hline & YEAR & 1996.1 & 1996.1 & 1996.1 & 1996.1 & 1996.2 & 1996.2 & 1996.2 & 1996.2 \\
\hline Nechako & 1994 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 \\
\hline Hansard & 1994 & 0.0001 & 0.8328 & 0.0001 & 0.0001 & 0.0001 & 0.1068 & 0.0001 & 0.0001 \\
\hline Woodpecker & 1994 & 0.0001 & 0.0003 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 \\
\hline Marguerite & 1994 & 0.0001 & 1.0000 & 0.0001 & 0.0001 & 0.0001 & 0.9815 & 0.0014 & 0.0185 \\
\hline Agassiz & 1994 & 0.0001 & 1.0000 & 0.0001 & 0.0001 & 0.0001 & 0.8364 & 0.0004 & 0.0058 \\
\hline Mission & 1994 & 1.0000 & 0.0227 & 1.0000 & 1.0000 & 1.0000 & 0.4479 & 1.0000 & 1.0000 \\
\hline Barnston & 1994 & 0.0278 & 0.9979 & 0.9990 & 1.0000 & 0.2196 & 1.0000 & 1.0000 & 1.0000 \\
\hline MainArm & 1994 & 0.0001 & 1.0000 & 0.0001 & 0.0009 & 0.0001 & 1.0000 & 0.0935 & 0.5406 \\
\hline NorthArm & 1994 & 0.1193 & 0.8833 & 1.0000 & 1.0000 & 0.5992 & 1.0000 & 1.0000 & 1.0000 \\
\hline N.Thompson & 1994 & 0.0001 & 1.0000 & 0.0062 & 0.1326 & 0.0001 & 1.0000 & 0.9898 & 1.0000 \\
\hline Thompson & 1994 & 0.0101 & 1.0000 & 0.9674 & 1.0000 & 0.0947 & 1.0000 & 1.0000 & 1.0000 \\
\hline Nechako & 1995 & 0.6499 & 0.3095 & 1.0000 & 1.0000 & 0.9957 & 0.9956 & 1.0000 & 1.0000 \\
\hline Hansard & 1995 & 1.0000 & 0.0190 & 1.0000 & 1.0000 & 1.0000 & 0.3971 & 1.0000 & 1.0000 \\
\hline Woodpecker & 1995 & 0.0001 & 1.0000 & 0.0079 & 0.1596 & 0.0001 & 1.0000 & 0.9950 & 1.0000 \\
\hline Marguerite & 1995 & 0.0025 & 1.0000 & 0.3959 & 0.9584 & 0.0184 & 1.0000 & 1.0000 & 1.0000 \\
\hline Agassiz & 1995 & 0.5157 & 0.3923 & 1.0000 & 1.0000 & 0.9820 & 0.9990 & 1.0000 & 1.0000 \\
\hline MainArm & 1995 & 0.0001 & 1.0000 & 0.0324 & 0.4221 & 0.0003 & 1.0000 & 1.0000 & 1.0000 \\
\hline NorthArm & 1995 & 0.2043 & 0.7898 & 1.0000 & 1.0000 & 0.7752 & 1.0000 & 1.0000 & 1.0000 \\
\hline N.Thompson & 1995 & 0.9907 & 0.0577 & 1.0000 & 1.0000 & 1.0000 & 0.7296 & 1.0000 & 1.0000 \\
\hline Thompson & 1995 & 1.0000 & 0.0001 & 1.0000 & 0.9806 & 1.0000 & 0.0001 & 0.1225 & 0.0108 \\
\hline Hansard & 1996.1 & . & 0.0001 & 1.0000 & 1.0000 & 1.0000 & 0.0001 & 0.3963 & 0.0512 \\
\hline Woodpecker & 1996.1 & 0.0001 & . & 0.0013 & 0.0364 & 0.0001 & 1.0000 & 0.7933 & 0.9996 \\
\hline MainArm & 1996.1 & 1.0000 & 0.0013 & & 1.0000 & 1.0000 & 0.0592 & 1.0000 & 0.9998 \\
\hline MainArm Sep & 1996.1 & 1.0000 & 0.0364 & 1.0000 & . & 1.0000 & 0.5620 & 1.0000 & 1.0000 \\
\hline Hansard & 1996.2 & 1.0000 & 0.0001 & 1.0000 & 1.0000 & & 0.0007 & 0.9370 & 0.3260 \\
\hline Woodpecker & 1996.2 & 0.0001 & 1.0000 & 0.0592 & 0.5620 & 0.0007 & . & 1.0000 & 1.0000 \\
\hline MainArm & 1996.2 & 0.3963 & 0.7933 & 1.0000 & 1.0000 & 0.9370 & 1.0000 & . & 1.0000 \\
\hline MainArm Sep & 1996.2 & 0.0512 & 0.9996 & 0.9998 & 1.0000 & 0.3260 & 1.0000 & 1.0000 & . \\
\hline
\end{tabular}

Appendix 26. Peamouth chub, mountain whitefish, and starry flounder HAI ANOVA and multiple comparison test tables. (continued)
Mountain Whitefish - Multiple comparison table; \(\operatorname{Pr}>|\mathrm{T}|, \mathrm{HO}: \operatorname{LSMEAN}(\mathrm{i})=\operatorname{LSMEAN}(\mathrm{j})\).
\begin{tabular}{|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|c|}
\hline \multirow[t]{2}{*}{\[
\overline{\mathrm{REACH}}
\]
i/j} & \multicolumn{3}{|r|}{Nechako Hansard} & \multirow[t]{2}{*}{Woodpecker
\[
1994
\]} & \multirow[t]{2}{*}{Marguerite 1994} & \multirow[t]{2}{*}{Agassiz
\[
1994
\]} & \multirow[t]{2}{*}{North Thompson 1994} & \multirow[t]{2}{*}{\begin{tabular}{c} 
Thomp- \\
son
\end{tabular}
1994} & \multicolumn{2}{|l|}{Nechako Hansard} & \multirow[t]{2}{*}{Woodpecker 1995} & \multirow[t]{2}{*}{Marguerite 1995} & \multirow[t]{2}{*}{Agassiz
\[
1995
\]} & \multirow[t]{2}{*}{North Thompson 1995} & \multirow[t]{2}{*}{\[
\begin{gathered}
\hline \begin{array}{c}
\text { Thomp- } \\
\text { son }
\end{array} \\
\\
\hline 1995 \\
\hline
\end{gathered}
\]} & \multirow[t]{2}{*}{Hansard
\[
1996.1
\]} & \multirow[t]{2}{*}{Woodpecker 1996.1} & \multirow[t]{2}{*}{Hansard
\[
1996.2
\]} & \multirow[t]{2}{*}{Woodpecker
\[
1996.2
\]} \\
\hline & YEAR & 1994 & 1994 & & & & & & 1995 & 1995 & & & & & & & & & \\
\hline Nechako & 1994 & & 0.0008 & 0.0001 & 0.6890 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 & 0.0001 \\
\hline Hansard & 1994 & 0.0008 & & 0.0114 & 1.0000 & 1.0000 & 0.0001 & 0.6264 & 0.2756 & 0.0001 & 0.0001 & 0.3824 & 0.0010 & 0.0001 & 0.0001 & 0.0065 & 1.0000 & 1.0000 & 0.8520 \\
\hline Woodpecker & 1994 & 0.0001 & 0.0114 & & 0.0017 & 1.0000 & 0.9951 & 1.0000 & 1.0000 & 0.4975 & 0.8685 & 1.0000 & 1.0000 & 0.1195 & 0.4027 & 1.0000 & 1.0000 & 0.9690 & 1.0000 \\
\hline Marguerite & 1994 & 0.6890 & 1.0000 & 0.0017 & & 0.9683 & 0.0001 & 0.1079 & 0.0378 & 0.0001 & 0.0001 & 0.0717 & 0.0002 & 0.0001 & 0.0001 & 0.0010 & 0.8182 & 0.9244 & 0.2146 \\
\hline Agassiz & 1994 & 0.0001 & 1.0000 & 1.0000 & 0.9683 & & 0.2797 & 1.0000 & 1.0000 & 0.0441 & 0.1184 & 1.0000 & 0.9916 & 0.0097 & 0.0339 & 1.0000 & 1.0000 & 1.0000 & 1.0000 \\
\hline N.Thompson & 1994 & 0.0001 & 0.0001 & 0.9951 & 0.0001 & 0.2797 & & 0.0452 & 0.4289 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.0664 & 0.0038 & 0.6108 \\
\hline Thompson & 1994 & 0.0001 & 0.6264 & 1.0000 & 0.1079 & 1.0000 & 0.0452 & & 1.0000 & 0.0019 & 0.0099 & 1.0000 & 0.9667 & 0.0002 & 0.0012 & 1.0000 & 1.0000 & 1.0000 & 1.0000 \\
\hline Nechako & 1995 & 0.0001 & 0.2756 & 1.0000 & 0.0378 & 1.0000 & 0.4289 & 1.0000 & & 0.0435 & 0.1543 & 1.0000 & 1.0000 & 0.0059 & 0.0308 & 1.0000 & 1.0000 & 1.0000 & 1.0000 \\
\hline Hansard & 1995 & 0.0001 & 0.0001 & 0.4975 & 0.0001 & 0.0441 & 1.0000 & 0.0019 & 0.0435 & & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.9375 & 0.0053 & 0.0001 & 0.1104 \\
\hline Woodpecker & 1995 & 0.0001 & 0.0001 & 0.8685 & 0.0001 & 0.1184 & 1.0000 & 0.0099 & 0.1543 & 1.0000 & & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.9988 & 0.0198 & 0.0008 & 0.2939 \\
\hline Marguerite & 1995 & 0.0001 & 0.3824 & 1.0000 & 0.0717 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.9996 & 1.0000 \\
\hline Agassiz & 1995 & 0.0001 & 0.0010 & 1.0000 & 0.0002 & 0.9916 & 1.0000 & 0.9667 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & & 0.9930 & 1.0000 & 1.0000 & 0.9038 & 0.3636 & 1.0000 \\
\hline N.Thompson & 1995 & 0.0001 & 0.0001 & 0.1195 & 0.0001 & 0.0097 & 1.0000 & 0.0002 & 0.0059 & 1.0000 & 1.0000 & 1.0000 & 0.9930 & & 1.0000 & 0.5005 & 0.0007 & 0.0001 & 0.0222 \\
\hline Thompson & 1995 & 0.0001 & 0.0001 & 0.4027 & 0.0001 & 0.0339 & 1.0000 & 0.0012 & 0.0308 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & & 0.8878 & 0.0037 & 0.0001 & 0.0840 \\
\hline Hansard & 1996.1 & 0.0001 & 0.0065 & 1.0000 & 0.0010 & 1.0000 & 1.0000 & 1.0000 & 1.0000 & 0.9375 & 0.9988 & 1.0000 & 1.0000 & 0.5005 & 0.8878 & & 0.9996 & 0.8560 & 1.0000 \\
\hline Woodpecker & 1996.1 & 0.0001 & 1.0000 & 1.0000 & 0.8182 & 1.0000 & 0.0664 & 1.0000 & 1.0000 & 0.0053 & 0.0198 & 1.0000 & 0.9038 & 0.0007 & 0.0037 & 0.9996 & & 1.0000 & 1.0000 \\
\hline Hansard & 1996.2 & 0.0001 & 1.0000 & 0.9690 & 0.9244 & 1.0000 & 0.0038 & 1.0000 & 1.0000 & 0.0001 & 0.0008 & 0.9996 & 0.3636 & 0.0001 & 0.0001 & 0.8560 & 1.0000 & & 1.0000 \\
\hline Woodpecker & 1996.2 & 0.0001 & 0.8520 & 1.0000 & 0.2146 & 1.0000 & 0.6108 & 1.0000 & 1.0000 & 0.1104 & 0.2939 & 1.0000 & 1.0000 & 0.0222 & 0.0840 & 1.0000 & 1.0000 & 1.0000 & \\
\hline
\end{tabular}
\begin{tabular}{lccccc}
\multicolumn{6}{l}{ Starry Flounder - Multiple comparison table; \(\operatorname{Pr}>|\mathrm{T}|, \mathrm{HO}:\) LSMEAN \((\mathrm{i})=\) LSMEAN \((\mathrm{j})\). } \\
\hline REACH & & Main & North & Main & North \\
i/j & & Arm & Arm & Arm & Arm \\
& YEAR & 1994 & 1994 & 1995 & 1995 \\
\hline MainArm & 1994 &. & 0.9951 & 0.2079 & 0.0039 \\
NorthArm & 1994 & 0.9951 &. & 0.5512 & 0.0005 \\
MainArm & 1995 & 0.2079 & 0.5512 &. & 0.0001 \\
NorthArm & 1995 & 0.0039 & 0.0005 & 0.0001 &. \\
\hline
\end{tabular}

\section*{Plates}

\section*{PLATE I. SPLEEN HISTOLOGY}

Photo 1. Normal spleen tissue from peamouth chub (WOOPC1). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 2. Spleen tissue with Myxobolus infection (arrow) from a peamouth chub (MCLPC17). H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 3. Post-mortem autolysis (arrow) in spleen tissue from a peamouth chub (NORPC55). H\&E. Bar \(=50 \mu \mathrm{~m}\).

Photo 4. Spleen melanosis in mountain whitefish (HANMW17. H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 5. Hemosiderin (hemo) in spleen from mountain whitefish (HANMW20). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 6. Lipoma found in the spleen of a mountain whitefish (NECMW57). H\&E. Bar=100 \(\mu \mathrm{m}\).

\section*{Plate I}


\section*{PLATE II. HINDGUT HISTOLOGY}

Photo 7. Normal hindgut tissue from mountain whitefish (MCLMW30). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 8. Hindgut tissue with Myxobolus infection (myx) from peamouth chub (NORPC37). H\&E. Bar \(=100 \mu \mathrm{~m}\).

Photo 9. Post-mortem autolysis (arrow) in hindgut tissue from peamouth chub (NORPC55). H\&E. \(B a r=100 \mu \mathrm{~m}\).

Photo 10. Inflammation (inf) of the hindgut from mountain whitefish (NECMW13). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 11. Inflammatory nodule (arrow) of the hindgut from mountain whitefish (MCLMW40). H\&E. Bar \(=100 \mu \mathrm{~m}\).

Photo 12. Inflammation with fibrosis of the hindgut from mountain whitefish (MCLMW20). H\&E. \(B a r=100 \mu \mathrm{~m}\).

\section*{Plate II}


\section*{PLATE III. KIDNEY HISTOLOGY}

Photo 13. Normal kidney tissue from mountain whitefish (MCLMW60). H\&E. Bar=100 mm .

Photo 14. Normal kidney tissue with Myxobolus (arrow) infection from peamouth chub (WOOPC01). H\&E. \(B a r=100 \mu \mathrm{~m}\).

Photo 15. Post-mortem autolysis (arrow) in kidney tissue from peamouth chub (NORPC55). H\&E. Bar=50 m .

Photo 16. Kidney melanosis in mountain whitefish (NECMW40). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 17. Lipid infiltration, large empty vacuolar spaces, of the kidney in mountain whitefish (AGAMW33). \(\mathrm{H} \& E . \mathrm{Bar}=100 \mu \mathrm{~m}\).

Photo 18. Renal xenomas, characterized by the thin kidney tubules, in peamouth chub (AGAPC40). H\&E. \(B a r=100 \mu \mathrm{~m}\).

\section*{Plate III}


\section*{PLATE IV. LIVER HISTOLOGY}

Photo 19. Normal liver tissue from peamouth chub (WOOPC01). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 20. Encapsulated Myxobolus infection associated with the liver tissue of peamouth chub (HANPC20). H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 21. Post-mortem autolysis (arrow) in liver tissue from peamouth chub. H\&E. Bar=50 \(\boldsymbol{\mu}\).

Photo 22. Vacuolation of hepatocytes due to glycogen storage in peamouth chub. H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 23. Hepato cellular steatosis in peamouth chub (NECPC01), characterized by numerous smooth edge, clear vacuoles within the hepatocytes. \(\mathrm{H} \& E . \mathrm{Bar}=50 \mu \mathrm{~m}\).

Photo 24. Hydropic vacuolation (arrows) of the hepatocytes in mountain whitefish (WALMW25). Histologically, affected cells possess a clear cytoplasm, small compact nuclei, and are markedly vacuolated. H\&E. Bar=50 1 m.

\section*{Plate IV}


\section*{PLATE V. LIVER HISTOLOGY}

Photo 25. Small inflammatory focus in the liver tissue of peamouth chub (HANPC35). H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 26. Large inflammatory focus in the liver tissue of mountain whitefish (WALMW13). H\&E.
\(B a r=100 \mu \mathrm{~m}\).

Photo 27. Small granuloma in the liver tissue of mountain whitefish (HISMW33(=WALMW11)). H\&E. \(B a r=100 \mu \mathrm{~m}\).

Photo 28. Necrosis (nec) and an associated encapsulated helminth (helm) in the liver tissue of peamouth chub (NECPC44). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 29. Hepatocellular coagulation necrosis (nec) in mountain whitefish (WALMW34) has a focal distribution affecting a large proportion of the liver and accompanied by fibrination. H\&E. \(B a r=100 \mu \mathrm{~m}\).

Photo 30. Liver cirrhosis in peamouth chub (MAIPC53), characterized by a mottled appearance. H\&E. \(B a r=100 \mu \mathrm{~m}\).

\section*{Plate V}


\section*{PLATE VI. LIVER HISTOLOGY}

Photo 31. Toxic insult to the liver tissue in mountain whitefish (NECMW37). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 32. Toxic insult to the liver tissue in mountain whitefish (NECMW37). H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 33. Pyknosis (arrows) in the liver tissue of the mountain whitefish (HANMW57). H\&E. Bar=30 mm .

Photo 34. Pyknosis (arrows) in the liver tissue of the mountain whitefish (WALMW17). H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 35. Nuclear pleomorphism and megalocytic hepatosis (arrows), characterized by an increase in nuclear to cytoplasm ratio, in peamouth chub (MCLPC25). H\&E. Bar=30 \(\mu \mathrm{m}\).

Photo 36. Nuclear pleomorphism and megalocytic hepatosis (arrows) in peamouth chub (MCLPC25). Note nuclei devoid of any nuclear content (lower arrow). H\&E. Bar=30 \(\mu \mathrm{m}\).

\section*{Plate VI}


\section*{PLATE VII. GILL HISTOLOGY}

Photo 37. Normal gill lamellae from peamouth chub (WOOPC02). H\&E. Bar=100 Hm .

Photo 38. Normal gill lamellae from peamouth chub (WALPC37) with encapsulated Myxobolus (myx) and external trematode (trem) infection. H\&E. Bar \(=100 \mu \mathrm{~m}\).

Photo 39. Post-mortem autolysis of gill tissue in peamouth chub (NORPC55). H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 40. Aneurysms (an) and hyperplasia (ha) of the secondary gill lamellae in mountain whitefish (HANMW25). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 41. Moderate hyperplasia of the secondary gill lamellae in peamouth chub (HANPC25). H\&E. \(B a r=100 \mu \mathrm{~m}\).

Photo 42. Severe hyperplasia resulting in fusion of the secondary gill lamellae in peamouth chub (HANPC10). H\&E. Bar=100 \(\mu \mathrm{m}\).

\section*{Plate VII}


\section*{PLATE VIII. GILL HISTOLOGY}

Photo 43. Encapsulated microsporidian (mic) by the secondary gill lamellae of peamouth chub (NECPC25). H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 44. Dermocystidium (derm) associated with the secondary lamellae of peamouth chub (NECPC45). \(H \& E . B a r=100 \mu \mathrm{~m}\).

Photo 45. Amoebae associated with the secondary lamellae of mountain whitefish (HISMW33(=WALMW11)). H\&E. Bar=50 \(\mu \mathrm{m}\).

Photo 46. Trichophyra infection (arrows) of mountain whitefish (WALMW53) gills. H\&E. Bar=100 Hm .

Photo 47. External trematodes found in peamouth chub (WALPC25) gills. H\&E. Bar=100 \(\mu \mathrm{m}\).

Photo 48. Large fluke found infecting and causing hyperplasia of the secondary lamellae of a mountain whitefish (NECMW01). H\&E. Bar=100 \(\mu \mathrm{m}\).

\section*{Plate VIII}


PLATE IX. NECROPSY: GONADS, LESIONS, GILLS, AND PARASITES

Photo 49. Abnormally immature ovaries in female mountain whitefish from Marguerite (MAR04B).

Photo 50. Swollen, immature ovaries in female mountain whitefish from Marguerite (MAR01B).

Photo 51. Normal ovaries, with parasite cysts, in female mountain whitefish from the Nechako River (NEC14B).

Photo 52. Large skin lesion on mountain whitefish from the Nechako River (NEC19B).

Photo 53. Hemorrhaging over opercules on mountain whitefish from the Thompson River (WAL32B).

Photo 54. "Frayed" gills on mountain whitefish from Hansard (HAN59A).

Photo 55. Ligula intestinalis in peamouth chub from the Thompson River (WAL06A).

Photo 56. Ligula intestinalis and peamouth chub from the Thompson River (WAL03A).

\section*{Plate IX.}


\section*{PLATE X. NECROPSY OF LIVERS AND KIDNEYS}

Photo 57. Normal liver in a peamouth chub from Hansard (HAN12A).

Photo 58. Fatty (coffee-with-cream) liver in a peamouth chub from Woodpecker (96WOOPC36).

Photo 59. Black discoloured liver in a peamouth chub from Woodpecker (96WOOPC35).

Photo 60. Grey discoloured liver in a peamouth chub from Woodpecker (96WOOPC35).

Photo 61. Light pink kidney (normal?) in a peamouth chub from Hansard (HAN12A).

Photo 62. Darkly discoloured kidney in a peamouth chub from Marguerite (MAR01A).

Photo 63. Mottled, fatty kidney in a mountain whitefish from the Thompson River (WAL39B).

Photo 64. Urolithic kidney in a mountain whitefish from the Nechako River (NEC29B).

\section*{Plate \(\mathbf{X}\).}
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[^0]:    ${ }^{1}$ from van den Berg et al. (1998)

[^1]:    ${ }^{1)}$ Tissues collected upstream of the Mission Reach in 1994 and all tissues collected in 1995 were analyzed at the National Laboratory for Environmental Testing, Burlington, Ontario.
    ${ }^{2)}$ Tissues collected at Mission and downstream in 1994 were analyzed by Quanta Trace, Burnaby, B.C.

