AN INFORMATION REVIEW OF FOUR NATIVE SPORTFISH SPECIES IN WEST-CENTRAL ALBERTA

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Prepared for

FOOTHILLS MODEL FOREST
Hinton, Alberta
and the
FISHERIES MANAGEMENT AND ENHANCEMENT PROGRAM

by
R.L. & L. ENVIRONMENTAL SERVICES LTD.
17312 - 106 Avenue
Edmonton, Alberta
T5S 1H9
Phone: (403) 483-3499

in association with
Lilley Environmental Consulting
Sherwood Park, Alberta

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EXECUTIVE SUMMARY

The Foothills Model Forest (FMF) Area is located in the foothills of west-central Alberta; it encompasses an area of approximately 2 500 000 ha and 3300 km of streams and rivers, exclusive of Jasper National Park (Rothwell and O'Neil 1994; Canadian Forestry Service 1996). The Foothills Model Forest is one of ten large-scale working model forests in Canada that comprise the Canadian Model Forest Program.

As part of a decision support framework, the Foothills Model Forest is developing a Watershed Assessment Model (WAM) to integrate fisheries, aquatic habitat, and hydrological values into the process. The primary goal of this study was to assemble existing information on four native sportfish species in the foothills region, namely rainbow trout (Athabasca River strain; *Onchorynchus mykiss*), Arctic grayling (*Thymallus arcticus*), bull trout (*Salvelinus confluentus*), and mountain whitefish (*Prosopium williamsoni*). The information was to include habitat requirements by life stage, and sensitivity to impacts arising from land use activities.

A literature search of scientific journals, reports (published and unpublished) and books yielded 398 citations related to the four target species or to the impacts of forest harvesting on aquatic environments. Many of these sources have been synthesized and summarized in this report. The habitat requirements summarized herein, provide both a regional (within the geographic range or Alberta specific) overview of the habitat requirements of the various life stages of the four target species as well as information specific to the FMF region. Habitat requirements of the four target species are well understood and documented throughout their geographic ranges. Less is known about specific requirements for populations found in the FMF region. In particular requirements for egg development and juvenile rearing were found to be lacking for all of the species of concern. Recommendations were provided for establishing research priorities to fill existing, data gaps.

Detailed habitat information from 27 reports was entered into a computer database which may be used to support future habitat modelling. In addition, the citations accumulated during the preparation of the report were entered into the computer database.

Existing habitat suitability models were also reviewed and summarized. Habitat suitability models were located for all the target species. Models for rainbow trout, mountain whitefish and Arctic grayling were readily available. The bull trout habitat suitability model was still being developed during the preparation phase of this report and could not be evaluated. Of the available models, only a modified Arctic grayling model has been utilized in Alberta. The other models require field testing in order to determine their applicability to Alberta in general and more specifically to the fish populations in the FMF. The potential for using the available models in the Foothills Model Forest region was assessed, deficiencies in the data required to run the models were identified and recommendations for using the models were provided.

To gain a better understanding of the habitat parameters that contribute to the suitability of Alberta foothills streams for rainbow trout, simple statistical tests were utilized to compare relative abundance (catch-per-unit-effort [CPUE] data) to selected habitat attributes in a representative sample of the studied streams in the Athabasca River drainage. Stream gradient, elevation and percentage of high quality runs/pools, were positively correlated with rainbow trout CPUE; while channel depth, variability of flows following spawning, drainage basin area and conductivity were negatively correlated.

Finally the impacts of land use activities, including linear developments (e.g., roads, transmission lines and pipelines) and timber harvest, on fish and fish habitat were summarized. When possible, concerns specific to the four target species or fish habitat in general in the Foothills Model Forest region were highlighted and discussed. Of the linear developments reviewed, road construction has the greatest potential to impinge upon the aquatic environment. Timber harvesting in a watershed can also affect aquatic habitats by increasing sediment loading into streams via surface soil erosion, mass-soil failures and stream bank erosion.

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1.0 INTRODUCTION

The Foothills Model Forest Area is located in the foothills of west-central Alberta; it encompasses an area of approximately 2 500 000 ha and 3300 km of streams and rivers in the area outside of Jasper National Park (Rothwell and O'Neil 1994; Canadian Forestry Service 1996). The forests in the area have been actively managed over the past 38 years (Rothwell and O'Neil 1994). The Foothills Model Forest is one of ten large-scale working model forests in Canada that comprise the Canadian Model Forest Program. The Canadian Model Forest Network Program is partnered with the International Model Forest Program. The Foothills Model Forest is sponsored by Weldwood of Canada, Canadian Forestry Service, Alberta Environmental Training Center (Hinton, AB), Alberta Environmental Protection and Jasper National Park.

As part of its development of a decision support system, the Foothills Model Forest is developing a Watershed Assessment Model (WAM) to integrate fisheries, aquatic habitat, and hydrological values into the process. The development of the WAM will assist managers in "maintaining the integrity of aquatic ecosystems and associated hydrological values and thus supporting a viable fisheries resource" (Foothills Model Forest 1995). The goal of this study was to assemble existing information on four native sportfish species in the foothills region, rainbow trout (Athabasca River strain; *Onchorynchus mykiss*), Arctic grayling (*Thymallus arcticus*), bull trout (*Salvelinus confluentus*), and mountain whitefish (*Prosopium williamsoni*). The information was to include habitat requirements by life stage, and sensitivity to impacts arising from land use activities.

Foothills Model Forest contracted R.L. & L. Environmental Services Ltd. to carry out the following tasks:

- Establish an up-to-date database of existing information on habitat requirements of four native sportfish species in west-central Alberta. The database will identify key habitat needs by species, lifestage, watershed location, and time of the year. It will also support the future development of critical habitat models for rainbow trout (Athabasca River strain), Arctic grayling, mountain whitefish and bull trout;

- Develop a focussed, comprehensive, easy to use database of aquatic information including key habitat needs common to most species. This information will be used to help evaluate land management alternatives, assess cumulative effects and environmental impacts, and identify management and inventory priorities;

- Using the existing information sources, bring together information on rainbow trout (Athabasca River strain), bull trout, Arctic grayling and mountain whitefish from Alberta sources and from transferable sources outside Alberta. Record information on habitat requirements, life-requisite needs, sensitivity to land use impacts, past research and existing habitat suitability models; and

- Develop a relational database of habitat information as part of the Watershed Assessment Model of the Foothills Model Forest. The database will include data input, reports, standard formats for references and will qualify data by source, date and evaluation of worth based on agreed upon criteria.

As noted above, the goal of this project was the preparation of a database which contained detailed habitat information for the four fish species of interest to the study. To supplement the database, a summary report identifying the habitat requirements and other relevant information for the target species was also prepared. The report summarizes the habitat requirements for each lifestage (spawning, rearing, adult feeding and holding, and overwintering) of the four target species in the Foothills Model Forest area as well as elsewhere in Alberta. When regional or Alberta information was not available, data from other regions (e.g., B.C., Montana, etc.), which was deemed by the authors of this report to be relevant or applicable to the Foothills Model Forest area, was included in the discussion. This information is presented in Section 3.0.

Several types of habitat models, including Habitat Suitability Index (HSI) and Habitat Quality Index (HQI) models, were also examined as part of the project. Overviews of these models have been included in Section 4.0. A discussion of some of the data deficiencies and limitations of these types of habitat models in relation to potential use as a management tool is included. In an effort to overcome these limitations, an alternative approach using available data and simple statistical tests is presented in Section 5.0.

The study also reviewed the potential impacts of various land-use activities on fish and fish habitat (Section 6.0). Where possible, specific impacts on the habitat requirements of the four target species were addressed.

2.0 METHODOLOGY

2.1 INFORMATION SEARCH

In order to identify potential sources of information relevant to the four target species extensive use was made of available information sources, which included:

- The Gate; - this allows the user to search 10 Alberta government libraries as well as the library holdings at the University of Alberta.
- Alberta Environmental Protection, Fish and Wildlife Library;
- Northern River Basins Study, Project Report No. 17; and
- Foothills Model Forest library.

The use of these information sources allowed many different scientific journals to be searched. Journals that were searched by the use of CD-ROM's included: Canadian Journal of Fisheries and Aquatic Sciences, Transactions of the American Fisheries Society, North American Journal of Fisheries Management, Science, Nature, Reviews in Fish Biology and Fisheries, Reviews in Fisheries Science, and Copeia. Other CD-ROM's that were searched included:

- Environmental Periodicals Bibliography NISC;
- Wildlife Review and Fisheries Review;
- Environmental Abstracts;
- AQUAREF;
- Life Sciences;
- Biological Abstracts;
- U.S. Fish and Wildlife Reference Service [via the Internet]; and
- ABSEARCH, which is a citation database for Transactions of the American Fisheries Society, Fisheries, North American Journal of Fisheries Management and the Canadian Journal of Fisheries and Aquatic Sciences.

The keywords used to search the above sources are provided in Table 2.1. The nomenclature for several of the target species have changed over the years (Table 2.2), therefore the former names were also included in the search. The currently accepted name, the former common name used by the source, and the Latin name were incorporated into the database. For several older references that dealt with Dolly Varden (*Salvelinus malma*), the name has been left as Dolly Varden because it was unclear from the information whether these fish were bull trout or Dolly Varden.

The information search phase of the study was conducted from mid-October to mid-December 1995, although relevant information that became available during the review phase was incorporated into the document.

Table 2.1 Literature Search Keywords.

Primary Term	Secondary Term
Common or scientific name and	Alberta British Columbia

	habitat
	forestry
	Idaho
	Montana
	Washington
Names of key authors	
Stream crossings	
Fish* and	habitat
Forest* and	fish
	habitat
	sediment
	water quality
Logging	
Riparian and	fish*
Road and	fish*
Sediment* and	fish*
Silt and	fish*
	forest*

*signifies a truncated or wildcard search term.

Table 2.2 Other common and scientific names for target species.

Current Common Name	Previous Common or Scientific Names	Current Scientific Name	Date of Change	Source
Arctic grayling	*T. montanus* Milner *T. signifer*	*Thymallus arcticus*	1958	Scott and Crossman 1973; Hubert et al. 1985
Bull trout	Dolly Varden (*S. malma)*	*Salvelinus confluentus*	1978	Skeesick 1989
Mountain whitefish	*Coregonus williamsoni* *Prosopium oregonium*	*Prosopium williamsoni*	1947	Scott and Crossman 1973
Rainbow trout	*Salmo gairdneri* Richardson	*Onchorynchus mykiss*	1988	Kendall 1988

2.2 DEVELOPMENT OF RELATIONAL DATABASE

The literature search described above yielded 398 citations related to the four target species or to the impacts of land use activities on aquatic environments. These citations were incorporated as a data table within the computer database and assigned keyword search terms.

To develop the detailed habitat data, information from selected references was entered into the habitat information data table. The information entered varied depending upon the emphasis of the source; however, material information generally included species, lifestage, habitat (stream type, substrate water velocity, etc.). The process of reviewing each article or report, determining its relevance to the Foothills Model Forest area and then entering the data into the computer was conducted from mid-December 1995 to the end of January 1996.

Several criteria were established for selecting the sources for entry into the habitat database. These were:

- Was the data specific to the Foothills Model Forest region?

- If not from the study area, was the data transferable because of similarities in geography, climate, stream setting, etc.?
- Was it quality information (e.g., scientific study design, statistically defensible results)?

The transferability and quality of data was assessed by the study team based on their experience in the study area and with the species of interest.

Detailed habitat information from 27 reports was entered into the database; these reports yielded 143 individual records. The habitat data entered (where available) included:

- species of fish,
- life stage of fish,
- stream order,
- stream gradient,
- water velocity,
- channel width,
- channel depth,
- discharge,
- substrate,
- habitat type,
- water quality,
- groundwater sources,
- migration barriers,
- instream cover, and
- riparian cover.

Information regarding the sensitivity of a species or life stage to land use impacts was also entered into the habitat database. Other information, such as fish population estimates, length frequency, age structure, growth characteristics and dietary information was noted and included in the database so that interested parties could return to the sources.

The detailed habitat information was stored in a relational database that was developed in Microsoft Access (Ver. 2.0). In the database, the habitat data table is linked with a bibliographic data table so that complete bibliographic citations can be extracted with the detailed habitat information. The database provides users with data entry screens and the ability to search the database by species or keywords. The information from such searches can be examined either as flat files (similar to spreadsheets), or in a report format, either on the computer screen or as a printout. Keywords were assigned to each entry in the two data tables. The keywords and their definitions are listed in Table 2.3.

Table 2.3 List of keywords used in database.

Keyword	Definition
Abundance	Used as a keyword to refer to references likely to provide an indication of the relative abundance of fish species or an estimate of fish populations (e.g., mark and recapture studies).
Age, Length, or Weight	Keywords which indicate that detailed information on these statistics is provided in the paper for one of the species of interest.
Angling	Covers discussion of the implication of sportfishing on the fish populations.
Contamination	Deals with the addition of contaminants to the aquatic environment - usually anthropogenic in origin - fine particles (i.e., sediment) are not included.
Water storage	Indicates that the effects of impoundments and dams on the aquatic environment are discussed.
Flow regime	Covers aspects of natural and altered stream flow patterns.
Linear development	Includes non-'logging' disturbances such as pipelines, roads, and power-lines.
Catch-per-unit-effort (CPUE)	Indicates that detailed information was provided about sampling methods and catch per unit of effort for these methods.
Diet	Indicates that the study referred to the feeding habits and food preferences.
Groundwater	Used when reference made to the relevance or importance of groundwater to the fish species in any life stage. This keyword was selected because of the importance of groundwater to the overwintering and spawning success of some fish species.
Habitat	Used when an information source had detailed information about the habitat. It encompasses information on substrate, stream channel width, depth, current velocity, discharge, instream cover, and riparian cover and is used if there was substantial information on these aspects in the reference.
Spawning, Incubation, Rearing, Feeding, and Overwintering	These terms are used to refer to the life stage of the species that is discussed. They are usually used in conjunction with habitat, for example, to indicate that the paper contained information on the rearing habitat of juvenile mountain whitefish. **Rearing** refers to the behaviour and habitat use of fry and juveniles, part of which obviously includes feeding. The keyword **feeding** was reserved for papers which clearly discuss adult behaviour, whether this was feeding or a description of the adult use of instream cover. This distinction is not always clear in every paper and the keyword applied was discretionary depending on which life stages seemed to be the main emphasis of a paper.
Inventory	Refers to a study that was conducted primarily to determine presence or absence and distribution. Such studies are not likely to contain the type of habitat information being sought in this study; therefore, few references in the database are inventories.
Location	Location names (province, state) describe the location or locations where field studies were undertaken. In the case of literature reviews or discussion papers, a location keyword would not be used.
Logging	This term is used to cover all aspects of timber harvesting operations, including patch cutting, clear cutting, buffer strips, roads, woody debris when discussed in the context of forest harvesting, and the implications for the aquatic environment. If the result is increased sediment loading, **sediment** would also be used as a keyword. If the implications of logging on stream temperature are discussed, then **temperature** would also be used as a keyword.
Movement	Used to indicate information sources which discuss fish migrations for any life stage. Most often movement is used in conjunction with another keyword to indicate the life stage involved. For example, a study may have involved an investigation of the spawning migrations of Arctic grayling.
Sediment	Used in a broad sense to include variations such as sedimentation, silt, siltation.
Species	Keywords are given for the four species of interest to this study and not for other species that may be discussed in the study or found in inventories.
Logging, Sediment, Angling, Contamination, Water storage, Flow regime, and Linear development	Suggest information about these various environmental changes or impacts. In some cases, one or more of these impacts may be the focus of discussion in the paper; in other papers it may only be a mention, for example, that overfishing is believed to be significant cause of reduced populations.
Water quality and Temperature	Terms given as keywords if these aspects of the aquatic environment are a significant aspect of the paper's discussion.

2.3 ANALYSIS OF HABITAT-FISH RELATIONSHIPS IN THE ATHABASCA DRAINAGE FOOTHILLS STREAMS

Habitat preference data directly applicable to the FMF region was generally lacking. For this reason information on the four target species was drawn from investigations and literature reviews conducted outside the study area. To provide a better understanding of fish-habitat relationships in Athabasca drainage foothills streams, data was analyzed (available in published reports) from 16 representative streams in the McLeod River, Berland River, Sakwatamau River and Freeman River watersheds. The analyses focussed on native populations of rainbow trout (Athabasca strain) because of data availability and time restrictions. Although the information presented would not be directly applicable to the other target species (bull trout, Arctic grayling, mountain whitefish) it was felt that the process might be transferable (i.e., it would be beneficial to conduct a similar evaluation for the other target species, as well as a more comprehensive analyses of rainbow trout).

Rainbow trout CPUE data (determined by electrofishing) were compared to a wide range of habitat variables in order to determine and statistically define correlations. The information was also used to develop a typical profile for a stream or stream reach inhabited by Athabasca strain rainbow trout.

3.0 REVIEW OF SPECIES HABITAT REQUIREMENTS

The following sections describe the habitat requirements of four key sportfish species (i.e., rainbow trout, bull trout, mountain whitefish, and Arctic grayling) in the Foothills Model Forest area. When available, information specific to Alberta (and the Foothills Model Forest region in particular) has been utilized. Information from other geographical areas has been used to supplement the review.

3.1 RAINBOW TROUT

Habitat suitability indices for rainbow trout have been developed by the U.S. Fish and Wildlife Service (Raleigh et al. 1984). However, as these authors point out, there are wide variations in the life history patterns of rainbow trout and in the habitat selected. The authors divide rainbow trout into three basic ecological forms: anadromous steelhead trout, stream resident rainbow trout, and lake or reservoir dwelling rainbow trout. They provide somewhat different indices for the different forms. They note that it is important to recognize that there is a genetic or hereditary basis for each ecological form and that each form may react differently to environmental stimuli.

Carl et al. (1994) reported that rainbow trout in the upper Athabasca River are genetically distinct from rainbow trout west of the Continental Divide. In conducting the present review, none of the papers distinguished between Athabasca strain rainbow trout and other strains with respect to their particular study area. This may be due to the difficulty in distinguishing resident Athabasca strain rainbow trout from other strains (i.e., stocked populations). Therefore, information on stream or river-dwelling rainbow trout in Alberta presented in the database and discussed here has not been differentiated into Athabasca strain and other strains.

Raleigh et al. (1984) characterized optimal riverine habitat for rainbow trout as: clear, cold water; a silt-free rocky substrate in riffle-run areas; an approximately 1:1 pool to riffle ratio; with areas of slow, deep water; well-vegetated stream banks; abundant instream cover; and relatively stable water flow, temperature regimes and stream banks.

Table 3.1 provides a summary of habitat data for the geographic range of rainbow trout, as well as for the Athabasca drainage of Alberta.

Table 3.1 Summary of habitat use by life requisite function for rainbow trout.

Life Requisite Function	Habitat Component	Geographic Range in North America*	Foothills Model Forest and other Relevant Transferrable Information
Spawning	Spawning habitat	small streams[a]	riffle/run (50%), flats (30%), pools (20%)[d]
	Preferred temperature	7.2 - 13.3°C[a]	commenced at 6 - 8°C[b]; 2 - 3°C[d]
	Preferred depth	0.15 - 2.5 m[a]	0.15 m[b]; <1.0 m[c]
	Preferred substrate	0.04 - 100 mm[a]	if <12 mm egg deposition at maximum depth, if 25-75 mm eggs found near the surface; 12% fines[b]
	Preferred velocity	0.30 - 0.90 m/s[a]	average 0.32 m/s (ranged from 0.12-0.69 m/s)[b]
Egg Development	Temperature tolerance	2 - 20°C[a]	no data (n.d.)
	Upper limit of optimal incubation	11°C[a]	n.d.
	Recommended oxygen requirement	>5.3 mg/L[a]	n.d.
	Minimum oxygen concentration	>4.3 mg/L[a]	n.d.
	Range of incubation time	18 - 102 days[a]	peak emergence in 55 days at 10°C, 95 days at 6°C[b]
	Recommended current velocity	0.02 m/s (intragravel velocity)[a]	<0.556 cubic meters/s[b]
	Substrate	gravel with < 5% fines[a]	
Rearing	Temperature tolerance	0 - 24°C[a]	n.d.
	Optimum temperature for growth	10 - 14°C[a]	n.d.
	Recommended oxygen concentration	>7 mg/L[a]	n.d.
	Lower lethal oxygen concentration	3 mg/L[a]	n.d.
	Habitat type preference	margins of lakes and streams[a]	most numerous in the mouth of small tributary streams[c]
	Depth preference	0.3 - 1.2 m[a]	in water shallower than where adults were found[c]
	Preferred current velocity	0.08 - 0.20 m/s[a]	n.d.
	Substrate	cobble, boulder, rubble[a]	n.d.
	Cover	cobble, woody debris[a]	n.d.
Adult Holding and Feeding	Temperature tolerance	0 - 28°C[a]	n.d.
	Optimum temperature for growth	10 - 14°C[a]	n.d.
	Recommended oxygen concentration	>7 mg/L if <15°C[a]; >9 mg/L if >15°C[a]	n.d.
	Lower lethal oxygen concentration	3 mg/L[a]	n.d.
	Habitat preference	lakes and streams[a]	n.d.
	Depth preference	variable depending on water temperature[a]	<1.0 m[c]
	Preferred current velocity	0.20 - 0.30 m/s[a]	n.d.
	Substrate	cobble, boulders[a]	sites dominated by medium size (64-255 mm) cobble[c]
	Cover	debris, boulders, light intensity[a]	n.d.
Overwintering	Young of the year & juveniles Cover	no data (n.d.) n.d.	macrophytes, cutbanks, emergent vegetation along stream margins[e]
	Substrate	n.d.	100-400 mm diameter[f]
	Cover	n.d.	“deeper” water[f]

*Information covers entire range of species in North America; when applying to FMF area, consideration should be given to the fact that rainbow trout are at the northern end of their range (e.g. temperature tolerance for adult feeding/holding will be considerably lower than 28°C).

[a] Ford et al. 1995; [d] O’Neil and Hildebrand 1986.

[b] Sterling 1986; [e] Riehle and Griffith 1993;

[c] R.L. & L. 1994a; [f] Raleigh et al. 1984.

3.1.1 Spawning and Egg Development

Raleigh et al. (1984) noted that the optimal spawning substrate for rainbow trout is different depending on the size of the spawners. For the model which these authors developed, the optimal spawning gravel was 15 to 60 mm in diameter for spawners less than 50 cm fork length; for larger fish, gravel and cobble up to 100 mm in diameter was considered optimal. This has implications to the Foothills Model Forest study area in Alberta where Dietz (1971) noted that rainbow trout in tributaries to the McLeod River are slow growing and reach small adult size. In Wampus and Deerlick creeks, spawning was observed to occur in loose gravel, 52 mm in diameter or smaller (Dietz 1971). Sterling (1986) also reported that none of the spawning gravel used in these creeks was larger than 50 mm. The spawning areas had convex bottom contours, that according to Dietz (1971) maximized interchange of surface and intragravel water ensuring oxygen supply to the eggs. Sterling (1986) reported that redds (spawning beds) were found in pool-riffle transition areas: these likely offer the same benefits of improved water flow through gravel.

It should be noted that the definitions of substrate type and size used in the literature are variable, which often makes comparisons difficult. Unless otherwise stated, we have assumed that the definitions used correspond to the classification system defined by Alberta Fish and Wildlife Phase II Fisheries Assessment guidelines (Table 3.2).

Table 3.2 Alberta Fish and Wildlife substrate classification.

Substrate type	Substrate size (mm)
Fines	<2
Small gravel	2-16
Large gravel	17-64
Cobble	65-256
Boulders	>256

In the Crowsnest River, Alberta, redds were usually located either peripheral to faster and deeper habitat types or on the inside curvature of river bends. A majority of the redds observed were situated at, or immediately downstream of, velocity breaks between shallow riffles and shallow runs (O’Neil and Hildebrand 1986).

Raleigh et al. (1984) indicated that optimal spawning gravel conditions included a minimal amount of fines (less than 5 percent); greater than 30 percent fines are assumed to result in low survival of embryos and yolk-sac fry. Hitchcock (1988) reported that in Beaver Creek, Montana, rainbow trout recruitment was poor, due to the high level of fines (greater than 20 percent) found throughout the study area. An exception to this occurred in 1982, when a 2400 percent increase in Age-0+ recruitment (1588 fish vs 63 fish in 1981) was observed. This was thought to have resulted from flushing of fines associated with a 1 in 10 year spring runoff event. In Tri-Creeks, spawning substrates had a mean particle diameter of 8 mm. Approximately 75 percent of spawning substrates were less than 25.4 mm in diameter and fines (< 0.841 mm) were generally less than 12 percent (Sterling 1986). O’Neil (1981) observed that spawning rainbow trout in Jarvis Creek, Alberta utilized gravel substrates that were 20-60 mm in diameter. A slightly smaller size range of substrate (16-32 mm) was observed to be used in the Crowsnest River (O’Neil and Hildebrand 1986).

Dietz (1971) noted scouring of typical spawning areas in Wampus, Deerlick, and Eunice creeks by a flood event in June, 1969. Only spawning areas in 'atypical' locations survived this freshet. The impact of scouring was exacerbated by the shallow egg deposition by rainbow trout. The eggs of larger fish, spawning in larger gravels, were more likely to survive scouring. Sterling (1990) indicated that the substrates in spawning areas in the Tri-Creeks study streams moved at discharges that exceeded 0.556 m^3/s and that the shallowness of egg deposition also made them vulnerable to changes in stream discharge. If spawning occurred at higher than normal flows, redds may have been located where they would become dewatered during lower flows; alternatively, spawning at low flows may leave eggs where they will be scoured at higher flows. Sterling (1990) indicated that a freshet of similar magnitude as that reported earlier by Dietz (1971) occurred again in 1980. Sterling (1990) also indicated that “any differences in population abundance due to altered habitat characteristics following logging are presumed to fall within, and are therefore masked by, the impact of the two flood events in 1969 and 1980”.

Raleigh et al. (1984) indicated that the temperature range during spawning was 2 to 14°C and the optimal average maximum water temperature for embryo development was 7 to 10°C. Dietz (1971) found that spawning coincided with an 8°C rise in water temperature over a one week period and occurred during the first 10 days of June in 1969. Sterling (1986) reported that the initiation of spawning coincided with water temperatures reaching 6 to 8°C. O’Neil (1981) observed similar temperatures for spawning in Jarvis Creek in early June, when the water temperature was 5 to 6°C. O’Neil and Hildebrand (1986) documented spawning in the Crowsnest River to occur earlier than most other authors, with redd building initiated in mid-April. During this period the water temperatures fluctuated with the air temperature and ranged from 0 to 8°C.

For the habitat suitability model, the optimal average velocity over spawning areas during embryo development was reported to be 30 to 70 cm/s (Raleigh et al. 1984). Sterling (1986) reported water velocities of 32.2 cm/s over spawning areas and water depths of 0.15 m. Similar velocities of 70 cm/s were reported by O’Neil (1981) for the Jarvis Creek population and 51 to 100 cm/s for the Crowsnest River (O’Neil and Hildebrand 1986). The optimal water depth for spawning is 0.17 m to 2.4 m according to the Raleigh et al. (1984) suitability index. O’Neil and Hildebrand (1986) reported that redds in the Crowsnest River were found in water depths of 0.2 to 0.5 m.

Sowden and Power (1985) have shown that the interstitial dissolved oxygen (DO) levels in the redds is crucial for embryo survival. They documented that there was no survival when DO in the redds was below 4.3 mg/L and that survival was negligible until mean DO concentrations exceeded 5.2 mg/L. Sowden and Power (1985) developed a linear relationship that described the embryo survival based on the mean DO concentration and the groundwater velocity in the redds.

Fry remain in the gravel for about 2 weeks after hatching and emerge 45 to 75 days after egg fertilization depending on water temperature (Raleigh et al. 1984). In Tri-Creeks, fry escapement from the gravels occurred in late July and early August approximately 10 to 14 days following hatching (Sterling 1986). The time period required for egg development, hatching and fry escapement was negatively correlated with water temperature. Escapement from spawning gravels averaged 33 percent and was weakly correlated with interstitial dissolved oxygen but not with substrate quality (substrate ranged from 0.1 to 50 mm in diameter, geometric mean of 8 mm). Interstitial oxygen varied from 0.5 to 13.4 mg/L (65-105% oxygen saturation) during incubation. Low dissolved oxygen levels were persistent at some sites but most sites had dissolved oxygen concentrations of 80 percent saturation (Sterling 1986).

3.1.2 Rearing

Only 8 percent of newly emerged fry survived to Age-1+ in the Tri-Creeks study. Survivorship was inversely correlated with estimated fry densities in August of the year. Yearling abundance was inversely correlated with the occurrence of discharges greater than 0.566 m^3/s during the incubation period for the cohort (Sterling 1986).

When moving to rearing areas, rainbow trout fry exhibit distinct movement patterns that may be genetically controlled downstream to a larger river or lake, upstream from an outlet river to a lake, or local dispersion to areas of low velocity and cover (Raleigh et al. 1984). Raleigh et al. (1984) reported that fry preferred shallower water and slower velocities than other life stages. Fry tolerate velocities up to 30 cm/s, but prefer velocities of less than 8 cm/s. Streams where pool areas contribute 40 to 60 percent of total stream area provided optimal habitat. Cover provided by aquatic vegetation, debris piles, and interstices is critical. Young rainbow trout live in shallower water and closer to escape cover than older trout. Few fry are found more than 1 m away from cover. Larger juveniles use instream cover (interstices, woody debris) more than undercut banks and overhanging vegetation (Raleigh et al. 1984).

Raleigh et al. (1984) reported that exhibited fry preference for rearing in still water; the preferred mean water column velocity bing 0 to 15 cm/s. The water depth preferred by fry is 30 to 45 cm. Juveniles prefer depths greater than 60 cm. Juveniles prefer sites where cover exceeds 15 percent of available stream area and where depths during low-water periods are more than 15 cm and velocities greater than 15 cm/s.

Optimal instream cover for fry and juveniles had more than 10 percent of the substrate in the size range of 100 to 400 mm (Raleigh et al. 1984). Streubel and Griffith (1993) examined the abundance of age 1 and older rainbow trout in Fall River, Idaho in relation to characteristics of pockets created by boulders. They found that the most suitable habitat was that in which the maximum water depth was greater than 0.5 m. Their research suggested that, in some of the study reaches, the Fall River had more suitable pockets of habitat available for trout than were being utilized.

The optimal temperature preferred by fry was reported to be 12 to 16°C and 10 to 18°C for juveniles (Raleigh et al. 1984). Fernet (1984) monitored rainbow trout fry drift in Ware Creek, Threepoint Creek, and the Sheep River, Alberta. He reported extensive downstream movement during July and August, 1983 with virtually all of this movement occurring between sunset and sunrise. Fry movements were evaluated in relation to water temperatures and discharge rates, but no uniform relationship between these variables and measured fry drift were detected.

3.1.3 Adult Feeding and Holding

Raleigh et al. (1984) reported that there was a definite relationship between the annual flow regime and the quality of trout habitat; the most critical period being during the lowest flow periods of summer and winter. An average annual base flow during the late summer or winter of more than 50 percent of the average annual flow provides high quality holding habitat according to Raleigh et al. (1984). The optimum mean water column velocity for adults is between 15 to 60 cm/s. The optimum average depth of the thalweg (line connecting deepest points in channel) during the late growing season low water period was greater than 30 cm when the stream widths were less than 5 m and greater than 45 cm when the stream widths exceeds 5 m (Raleigh et al. 1984). Average maximum water temperatures of between 12 and 18°C, during the warmest period of the year, were preferred. Average minimum dissolved oxygen levels exceeding 7 mg/L during the late growing season, low water period (at water temperatures less than 15°C) were considered to be optimum conditions in the suitability model (Raleigh et al. 1984). Annual maximal or minimal pH levels falling between 6.6 and 7.8 were reported to be characteristic of good quality habitat for rainbow trout (Raleigh et al. 1984).

Raleigh et al. (1984) indicated that optimum habitat for adult rainbow trout existed when the percentage of pools during the late growing season, low water period was 40 to 65 percent of the stream area; more than 30 percent of the pool area was composed of large, deep pools (if pool ≤5 m wide greatest pool depth ≥1.5 m; if pool >5 m wide, pool depth ≥2 m) which provided low velocity resting areas; and more than 30 percent of the stream bottom was obscured by depth, turbulence, overhanging banks and vegetation, or debris. A contribution of rubble and small boulders exceeding 50 percent of the substrate type in riffle/run areas was considered to be optimal for food production. In spring areas, aquatic vegetation could also be a source of food production. There should be limited amounts of gravel, large boulders or bedrock in food production areas and the percentage of fines (i.e., less than 3 mm diameter) in riffle/run areas during average summer flows should be less than 10 percent (Raleigh et al. 1984).

A percent instream cover of 25% during the late growing season, low-water period (at depths greater than 15 cm and velocities less than 15 cm/s) was considered to be optimum for adult rainbow trout holding and feeding in the suitability index model developed by Raleigh et al. (1984). These authors also indicated that the average percent vegetational ground cover and canopy closure along the stream bank during the summer (i.e., to provide allochthonous input) should exceed 150 on a Vegetation Index Scale[1] to achieve optimum conditions. The average percent rooted vegetation and stable rocky ground cover along the stream bank should exceed 75 percent. Although not an issue in most places in Alberta, Raleigh et al. (1984) suggested that 50 to 75 percent of the stream area should be shaded between 10 AM and 2 PM unless the maximum water temperature was less than 18°C.

It is not clear how critical water temperature is for adult fish. Matthews et al. (1994) conducted a study of daily rainbow trout movements in pools located in the North Fork of the American River, California using temperature

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[1] VI=2(%of shrubs along stream bank)+1.5(%of grasses along stream bank)+ % of trees along stream bank.

sensitive radio telemetry tags. It was observed that rainbow trout did not show a preference for cooler areas of the pools. Daily temperatures varied from 12 to 19.3°C and rainbow trout could be found in areas where the water temperature was 19.3°C even though other areas of the pool had water temperatures that were considerably cooler (14.5°C). Matthews et al. (1994) also observed that rainbow trout were much more active at night than during the day; the trout were often observed holding under rocks or ledges during the day.

In a controlled experiment, Pert and Erman (1994) observed adult rainbow trout utilization of stream habitat under different discharge regimes (1.6 to 5.1 m^3/s). They observed that as discharge and water velocities increased, the area of rainbow trout utilization around a focal point also increased. At the highest discharge levels, trout were found in the deepest water and were closely associated with the bottom. Pert and Erman (1994) also observed that rainbow trout were closely associated with boulders at all discharges and that some individuals were highly territorial (always found in the same area regardless of discharge rates). Based on their observations Pert and Erman (1994) cautioned that “It is likely that few individuals of territorial fish occupy the optimal habitat. Interpreting the most frequently used microhabitat for a population is probably incorrect and could result in erroneous predictions of available habitat based on instream flow assessment models”.

In the Foothills Model Forest region, Sterling (1978) reported that stream gradients in the study sections of the three streams examined during the Tri-Creeks project varied from 1.9 to 3.5 percent. Channels varied in width from 1.8 to 4.6 m, depths were 12.7 to 25.4 cm in riffles, and instream cover was provided by undercut banks and submerged logs. Stream banks were well vegetated and additional cover was provided by overhanging vegetation.

R.L. & L. (1995a) noted that in the Athabasca River most rainbow trout and bull trout were captured in upper sections on the mainstem river and that species diversity tended to be highest in downstream sections. In a study conducted in Idaho, Platts (1979) noted that as stream order (1 to 5) increased, the width, depth, and percent of channel containing rubble increased, while channel gradient, channel elevation, and percent of channel composed of gravel decreased. The number of fish species, summer water space (depth times surface area) for fish, and the total number of fish also increased in the higher order streams.

3.1.4 Overwintering

The onset of winter brings about changes to rainbow trout behaviour and their habitat use. Winter can also affect trout survival by changing the physical habitat. The most significant changes are reduced water temperature, ice formation, and snow cover (Marcus et al. 1990). Changes in trout behaviour are triggered by low temperatures. Prior to the onset of winter, adult rainbow trout tend to move into deeper water (Raleigh et al. 1984). Riehle and Griffith (1993) also noted changes in behaviour of young-of-the-year rainbow trout in Silver Creek, a partially spring-fed stream in Idaho. They found that in late September, the young-of-the-year aggregated briefly during the day and then began to conceal themselves in macrophyte beds, under cutbanks, and in submerged sedges and grasses along streambanks. This behavioural change happened as temperatures dropped below 8°C.

During the winter feeding habitats change, with periods of peak feeding changing from the afternoon and evening to mainly at night in October. The number of trout visible was greatest 30 to 60 minutes after sunset and about 30 minutes before sunrise. As a result, trout no longer fed on chironomids which were abundant in October in the daytime drift but not at night (Riehle and Griffith 1993). The authors suggested that Silver Creek rainbow trout young-of-the-year experienced a metabolic deficit during the winter. This deficit began in late September and coincided with the change in hiding and feeding behaviour.

Resident fry usually overwinter in shallow areas of low velocity near the stream margin, with rubble being the principal cover. The optimal size of substrate used as winter cover by fry and small juveniles ranges from 100 to 400 mm in diameter and should exceed 10 percent of the total substrate in wintering areas (Raleigh et al. 1984). The presence of more than 10 percent fines in riffle-run areas reduces the value of these areas as cover for fry and small juveniles. Young salmonids occupy different habitats in winter than in summer, with log jams and rubble being important winter cover (Raleigh et al. 1984). As in the summer, the winter average annual base flow exceeding 50 percent of the average annual daily flow was considered to be optimal for overwintering (Raleigh et al. 1984).

3.2 BULL TROUT

Bull trout were formerly considered to be synonymous with Dolly Varden (*Salvelinus malma*). In 1978, Cavander suggested that there were two distinct species, bull trout and Dolly Varden (in Haas and McPhail 1991). Bull trout were historically found from northern California to approximately 60°N and from approximately 114°W to the Pacific Coast (Haas and McPhail 1991). In Alberta, bull trout occupied all major Eastern Slope drainages and were found in greatest abundance in the upper portions of watersheds. In Alberta, approximately 20 000 km of stream habitat and 12 000 hectares of lake habitat are occupied by bull trout (Berry 1994).

Bull trout populations are very sensitive to overfishing, habitat changes and competition from exogenous trout species (Berry 1994). This sensitivity to impacts has resulted in the localized extirpation of bull trout in northern California; bull trout are also facing extirpation in Nevada (Haas and McPhail 1991). In Alberta, the range of bull trout has been reduced mainly as a result of overfishing. Currently, a management and recovery plan has been put in place to restore bull trout populations (Berry 1994).

There are two distinct types of bull trout populations, adfluvial and fluvial. Adfluvial populations spend a majority of their life cycle in lakes and migrate into tributary streams to spawn. Fluvial populations inhabit rivers and move into smaller tributary streams to spawn. In Alberta the fluvial bull trout can be divided into two groups; migratory and stream resident. Migratory populations fit the classical definition of fluvial (i.e., reside in large rivers and migrate into smaller tributary streams to spawn). The stream resident populations inhabit smaller streams throughout the year and are generally smaller in size than the classical fluvial fish. Adfluvial fish tend to be larger than those from fluvial populations, reaching weights of 2 to 18 kg (Pratt 1992). Pratt (1992) noted that most of the information available on the habitat and life history of bull trout is primarily for adfluvial populations. Although much of the research conducted on bull trout has been for adfluvial populations, some of this information, particularly spawning substrate and juvenile habitat use is transferable to the Foothills Model Forest region. For example, bull trout from fluvial populations in the Athabasca River attain sizes similar to adfluvial populations and thus should be able to utilize similar sizes of substrate for redd construction. Until more research is done to gather site specific information in the Foothills Model Forest region, information for adfluvial populations will have to be relied upon to provide an understanding of bull trout habitat use. Table 3.3 provides a tabular summary of bull trout habitat characteristics.

3.2.1 Spawning and Egg Development

Kitano et al. (1994) investigated bull trout spawning in the upper Flathead River, Montana; they reported that bull trout females (350 to 700 mm total length) constructed redds in the downstream ends of pools, where fine gravel was prevalent. Also in the Flathead River, Shepard and Graham (1982) determined that most bull trout redds were constructed in substrates with a diameter ranging from 6 mm to 50 mm range. Eggs were encountered at depths of up to 15 cm in the substrate at these locations. A mean water depth of 0.28 m and a mean water velocity of 0.31 m/s were recorded.

In Washington streams, Underwood et al. (1995) determined that substrate sizes in the bowl of the redd ranged between 16 and 130 mm diameter (n = 74); in the tail of the redd the diameter ranged between 2 and 60 mm (n = 35). In the Wigwam River, British Columbia, Oliver (1985) noted that females selected redd sites in areas of shallow depths, low surface velocity, and within 2.5 metres of the streambank. These redds were excavated in fine and coarse gravels containing high silt content. Other studies generally indicated low levels of silt or sand in the substrate. Ford et al. (1995) have recently summarized the characteristics of bull trout spawning habitat (Table 3.4).

Table 3.3 Summary of habitat use by life requisite function for bull trout.

Life Requisite Function	Habitat Component	Geographic Range in North America*	Foothills Model Forest and other Relevant Transferrable Information
Spawning	Spawning habitat	small streams[a]	no data (n.d.)
	Preferred temperature	<9.0°C, most intense at 5-6°C[b]	n.d.
	Preferred depth	0.15 - 0.84 m[a]	0.25 m, 0.8 m maximum[c]
	Preferred substrate	cobble, gravel[a]; 20% cobble, 50% large gravel, 30% fines	same as noted above for adults[c]
	Preferred velocity	0.25 - 0.65 m/s[a]	0.19 m/s in riffle/run[c]
	Groundwater in spawning areas	yes, may mitigate harsh winter temperatures and anchor ice[b]	yes, groundwater 1-3°C warmer than mainstem in winter[c]
Egg Development	Temperature tolerance	0 - 8°C[a]	n.d.
	Gradient	no data (n.d.)	n.d.
	Optimal incubation temperature	2 - 4°C[a]	n.d.
	Recommended oxygen concentration	9.5 mg/L[a]	n.d.
	Range of incubation time	34 - 125 days[a]	n.d.
	Recommended current velocity	below level causing gravel scour[a]	n.d.
Rearing	Temperature tolerance	0 - 18°C[a]	8 - 9°C[f]

	Optimum temperature for growth	<12°C[a]	n.d.
	Gradient	n.d.	1.2 to 6.7 m/km[d]
	Recommended oxygen concentration	7.75[a]	n.d.
	Lower lethal oxygen concentration	n.d.	n.d.
	Habitat type preference	pools[a]	n.d.
	Depth preference	<1.0 m[a]	n.d.
	Preferred current velocity	<0.5 m/s[a]	n.d.
	Substrate	cobble and boulder[a]	gravel, cobble, boulder[d]
	Cover	cobble and fine debris[a]	woody debris, overhanging vegetation, overhanging bank[d]
Adult Holding and Feeding	Temperature tolerance	0 - 12.8°C[a]; 9 - 15°C[b]	n.d.
	Gradient	10 - 20%[b]	n.d.
	Recommended oxygen concentration	7.75 mg/L[a]	n.d.
	Habitat preference	lakes and large rivers[a]; deep pools[b]	60% riffle/run, 30% flat and 10% pools[c]
	Depth preference	varies, up to 18 m[a]	0.75 - 1.5 m[d]
	Preferred current velocity	n.d.	0.09 - 0.19 m/s in riffle/run[c]
	Substrate	n.d.	majority large cobble with localized patches of very coarse gravel and small cobbles[c]
	Cover	depth[a]; woody debris, cutbanks and debris jams[b]	n.d.
Overwintering	Groundwater sources	n.d.	important[c]
	Preferred depth	n.d.	>0.75 m deep pools[e]

*Information covers entire range of species in North America.

[a] Ford et al. 1995;

[b] Skeesick 1989;

[c] Hildebrand 1985;

[d] R.L. & L. 1994a;

[e] Pisces 1993;

[f] Halstead 1984.

Table 3.4 Bull trout spawning habitat characteristics.

Drainage	**Mean water depth (m)**	**Mean water velocity (m/s)**	**Streambed Composition (%)**				**Depth of egg deposition (m)**
			Cobble and larger	**Large gravel**	**Small gravel**	**Sand**	
Flathead River, Montana	0.28 (0.15-0.35)	0.29 (0.24-0.61)	18	30	39	13	0.1-0.2
Sawmill Springs, Alberta Timber Creek, Alberta	0.24 0.58	0.52 0.44	54	1214	7270	109	0.03-0.18
Mackenzie Cr., Upper Arrow Lakes, B.C.		0.57-0.64	0	31	61	8	0.10-0.16
Wigwam Cr. and Ram Cr., Kootenay River, B.C.	0.34	0.43	20		50	30	0.17-0.25
Meadow Cr., Kootenay Lake, B.C.	0.73-0.84	0.04-0.61	29	59		12	0.1
Line Cr., B.C.	0.21	0.28	0.76				

Source: Ford et al. 1995.

Few studies in Alberta have specifically examined bull trout redds, but bull trout are known to spawn in streams with substrates composed primarily of cobble (75 to 300 mm diameter) (Pisces 1993), and cobble and gravel substrates (Halstead 1984, R.L. & L. 1994a). In a study of potential bull trout spawning habitat in the Alberta east slope streams, streams rated as having high to excellent potential included those with a majority of the substrate in the medium gravel to small cobble range (8 to 128 mm); of particular value were systems with contributions of very coarse gravel (32 to 64 mm) (Hildebrand 1985). Bull trout redds in the Clearwater River system in Alberta featured a range of materials varying from sand to coarse gravel and rubble, but the predominant substrate was fine gravels (2 to 14 mm diameter) and gravel (15 to 33 mm) (Allan 1980).

Graham et al. (1981) found stream order and D_{90} (the diameter of substrate material which was larger than 90 percent of all streambed material) to be the best combination of variables for predicting which stream reaches would contain redds. The use of these two variables correctly classified 58 percent of the stream reaches into frequency of use categories (no-redds, low-redd frequency, high-redd frequency) (Graham et al. 1981).

Underwood et al. (1995) measured water velocity in and near bull trout redds in three streams in southeast Washington. Water velocities and depths in the bowl of the redds averaged from 15.2 to 23.5 cm/s and 21 to 24 cm, respectively. Velocities in the tail of the redds were 33.4 to 40.4 cm/s and depths were 12 to 18 cm. Beside the redds, velocities were 23.4 to 32.2 cm/s and depths were 13 to 14 cm. Gradients in these streams ranged between 2.6 and 3.3 percent.

R.L. & L. (1992) examined 25 bull trout redds located in the West Castle River and Mill Creek, Alberta. The 12 redds observed in the West Castle River were located in areas with average water depths of 0.30 m and velocities of 19 cm/s. Substrate composite consisted of 55.4% pebbles, 19.6% larges (i.e. cobbles and boulders), 17.1% gravel and 7.9% sand. Redds were on average 1.54 m in length and 0.61 m wide. Similar findings were reported for the redds examined in Mill Creek.

Overall, streams preferred by bull trout for spawning have a good mix of pools, riffles and runs (Environmental Management Associates 1981, Halstead 1984, Hildebrand 1985, Pisces 1993). Within these streams, redds tend to be located in runs or in the tail end of plunge pools, although riffles and cascades were also used.

Areas of groundwater influence (i.e., seeps or springs) are preferred spawning reaches for bull trout (Graham et al. 1981, Hildebrand 1985, Shepard 1985, Underwood et al. 1995). Side channel development also seems to have a positive relationship with redd frequency (Graham et al. 1981).

Shepard (1985) noted an inconsistent relationship between the number of redds and overhanging bank cover, but in general, more available cover was correlated with more redds. Underwood et al. (1995) noted that redds were located, on average, about 97 cm from cover and 14 cm from the stream bank. The preferred instream cover was provided by undercut banks, overhead cover, fallen logs, and root wads. Hildebrand (1985) and Pisces (1993) also noted the importance of cover provided by woody debris, boulder gardens, overhanging banks and overhanging/submerged vegetation. Bull trout not actively involved in spawning were observed closely associated with instream cover (Graham et al. 1981).

The timing of spawning is variable throughout the geographic range of bull trout. In British Columbia, bull trout can be observed entering spawning streams as early as July with peak movements occurring in August; spawning takes place in September and October (Ford et al. 1995). In the Chowade River, B.C., upstream movements were completed by 28 August 1994. The peak of downstream movements following spawning occurred between 8 and 24 September 1994 (R.L. & L. 1994b). In the Clearwater River, Alberta, the majority of bull trout spawners appeared in the streams between 21 August and 10 September 1992 (Rhude and Rhen 1995). Halstead (1984) reported that spawning was completed by 28 August 1983 in streams near Grande Prairie, Alberta.

Hatching was complete in the Clearwater River system by the end of January. Fry remained in the gravel for 100 to 145 days and emerged at a length of 20.75 mm to 23.6 mm. Fry emergence peaked in early April (Allan 1980). Shepard et al. (1984) determined that bull trout embryo survival and subsequent fry emergence success were inversely correlated with the percentage of substrate smaller than 6.4 mm in diameter in the streambed in Coal Creek, Montana.

3.2.2 Rearing

After hatching it takes the newly emergent fry approximately three weeks to attain neutral buoyancy. During this period of time fry hold position near the substrate to avoid being swept downstream. After neutral buoyancy is achieved, fry rear in low velocity backwaters and side channels. In these areas, bull trout fry prefer substrate interstices or habitat within 0.30 m of the bottom. Fry are associated with cover such as cobble, boulders and fine debris, where the average water velocity is 0.09 m/s (Ford et al. 1995). Larger juveniles also utilize large woody debris in locations where the water depth is less than 0.5 m (Ford et al. 1995).

Saffel and Scarnecchia (1995) examined habitat usage by Age-0 and Age-1 bull trout in four Idaho streams. Age-0 bull trout used riffles, runs and pools in equal proportions; however, in all of these habitat types, most (88%) of the Age-0 fish were associated with the channel margins. In contrast, Age-1 bull trout exhibited a preference for pools and avoided riffles. Distribution in the stream channel also differed with 91% of Age-1 bull trout found in the main channel. Density of bull trout was positively correlated with the number of pools/100 m of stream. The authors also observed that maximum summer water temperature influenced the density of juvenile bull trout. Juveniles were located in highest densities (3.9 to 11.2 fish/100 m^2) in reaches where summer maximum water temperatures ranged from 7.8 to 13.9°C, whereas lowest densities (<1.0 fish/100 m^2) were recorded when maximum summer temperatures were between 18.3 and 23.3°C.

Pratt (1985) investigated habitat preferences of juvenile bull trout in the Flathead River, Montana. He reported that juveniles (40 to 200 mm in fork length) distributed themselves along the stream bottom and sought slow water (10 cm/s) in association with submerged cover. Unembedded substrate was preferred. He also noted that areas of preferred water velocity were produced in small pockets (i.e., interstitial spaces amongst clean cobble). Pratt (1985) cautioned that due to the localized distribution of these low velocity pockets, describing mean velocities using traditional field methods may not adequately assess available bull trout habitat.

Woody debris jams also may be used for cover, but these debris jams should be unconsolidated in order to allow flow-through (Underwood et al. 1995). According to the authors, juveniles prefer slow water created by boulders and woody debris. Juvenile bull trout select areas closer to instream cover and to the streambed than other salmonid species, such as *O. mykiss* and *O. tshawytscha*. Underwood et al. (1995) noted differences in habitat preferences between Age-0+ and juveniles. Age 0+ fish preferred plunge and scour pools but also used riffle and cascade habitat; juveniles preferred plunge and scour pools. In their second year of sampling they determined that juveniles used mostly runs. The difference in habitat selection between years may have been due to differences in sampling technique (electrofishing in 1991 and snorkelling in 1992). The gradients of the streams in Washington studied by Underwood et al. (1995) ranged from 2.6 to 3.3 percent. Stream widths were 4.3 to 7.5 m and pool depths 0.53 to 0.77 m. Stream velocities ranged between 34 and 46.9 cm/s, although fish were generally inhabiting the slower areas.

Juvenile bull trout in the upper Flathead River were present in areas with water temperatures in the 5 to 12°C range. Stream reaches with warmer temperatures were not used. Bull trout were found at the highest densities in streams with cold spring influence and closed forest canopy (Pratt 1985).

Shepard et al. (1984) reported a significant positive correlation between substrate score and densities of juvenile bull trout (fish longer than 75 mm) in 26 Swan River (Flathead River drainage) tributary reaches. Oliver (1985) observed that the highest densities of juvenile char, in the Wigwam River in British Columbia, were usually in portions of the stream dominated by a rubble-boulder bed material and characterized by rolling to broken flows.

In several Alberta river systems (i.e., North Wolf Creek, Brazeau River, Athabasca River), it has been observed that the preferred habitat types were predominately riffles, runs or flats interspersed with pools and cascades (Halstead 1984, Pisces 1993, R.L. & L. 1994a). Pisces (1993) indicated that cascade habitat provided good rearing habitat for juvenile bull trout. Halstead (1984) reported that water temperatures in North Wolf Creek, Alberta were 8 to 9°C in August.

Juvenile bull trout in Alberta have been reported in streams with substrates composed of predominately gravel, cobble and boulder (Halstead 1984, Pisces 1993, R.L. & L. 1994a). The cover in these streams was provided by woody debris, overhanging vegetation, overhanging banks, and boulder gardens, although the latter type generally provided less of the available cover according to Pisces (1993). The gradient in streams inhabited by juvenile bull trout was in the 1.2 to 6.7 m/km range (Halstead 1984, R.L. & L. 1994a). Based on studies in the Liard River drainage in British Columbia Stewart et al. (1982) reported that small (<300 mm) Dolly Varden char (i.e., bull trout) used the upper reaches of Adsett Creek, a second order tributary, while larger char inhabited in the lower reaches.

3.2.3 Adult Feeding and Holding

Adult bull trout have been recorded in many Alberta East Slope tributaries. These streams were typically low to moderate gradient systems characterized by gravel, cobble and boulder substrates (Halstead 1984, Pisces 1993, R.L. & L. 1994a). Adults tend to occupy deeper and faster waters than juveniles. The cover in these streams was provided by woody debris, overhanging vegetation, and overhanging banks (Pisces 1993). Preferred habitat was a mix of runs, flats and riffles interspersed with pools. R.L. & L. (1994a) noted that adult bull trout in the upper reaches of the Athabasca River were inhabiting pools deeper than 1.5 metres with large boulder cover.

Reported correlations between habitat characteristics and bull trout abundance have been quite variable. A study in Idaho found that the abundance of bull trout decreased with increasing stream order and suggested that the analysis demonstrated the value of headwater streams for this species (Platts 1979). Stewart et al. (1982) reported that the abundance of bull trout in Adsett Creek, B.C. was positively correlated with the abundance of prey (Arctic grayling and mountain whitefish). Halstead (1984) reported that bull trout presence increased with channel width variability and decreased with elevated turbidity, bicarbonate ion concentrations, and percent deciduous riparian vegetation. Bull trout abundance also increased with stream slope and stream valley slumping, and was negatively related to the occurrence of flood plains and gravel bars.

3.2.4 Overwintering

Hildebrand (1985) surveyed potential bull trout spawning areas in the east slopes region of Alberta. The study focussed on streams with open water during the winter (due to groundwater input) and known from previous studies to support abundant populations of bull trout. The study also included an assessment of the suitability of these streams for overwintering. Although no bull trout were observed in these areas, several sites were rated as having high to excellent potential for overwintering. These potential overwintering areas were featured by springs or groundwater seepage which were sufficient to keep sections of the stream open during the winter and contribute to winter baseflows. These areas also were characterized by deep pools (>0.75 m deep) with abundant cover provided by submerged bankside debris and undercuts banks. Dissolved oxygen concentrations were between 11 and 12 mg/L. Similar areas in tributaries to the Brazeau River, Alberta were also rated as providing good overwintering conditions (Pisces 1993).

Allan (1980) reported that all adult bull trout had moved out of spawning area streams by 1 October and that adults were not observed in any spawning tributary between November and July. R.L. & L. (1993) reported similar findings in the upper reaches of the Athabasca River, Alberta. In this study, radio tagged adult bull trout moved upstream during late summer into the Jasper National Park portion of the Athabasca River drainage; this was assumed to be a spawning-related migration. By the end of October, the fish had returned downstream to the same area of the Athabasca River where they were initially caught and tagged in spring 1992; the fish remained relatively static at this location until March (when the study ended) and it was concluded that this was an overwintering area.

3.3 MOUNTAIN WHITEFISH

In an extensive review of literature on mountain whitefish, Northcote and Ennis (1994) noted substantial gaps in the essential life history and biological understanding of mountain whitefish, which, to the authors, was surprising given the widespread distribution of the species. For example, Northcote and Ennis (1994) noted that there was an absence of detailed information on the spawning behaviour and requirements of mountain whitefish. Apart from a few isolated studies on temperature and oxygen effects on egg development, little is known about the effects of other environmental parameters on hatching and larval emergence. Evidently more information is needed on the timing of fry emergence, behaviour, and early habitat requirements (Northcote and Ennis 1994). Table 3.5 provides a summary of habitat requirements for the different lifestages of mountain whitefish.

Table 3.5 Summary of habitat use by life requisite function for mountain whitefish.

Life Requisite Function	Habitat Component	Geographic Range in North America*	Foothills Model Forest and other Relevant Transferrable Information
Spawning	Gradient		0.9 to 1.2 m/km[e]
	Preferred temperature	>6.0°C[a]; 2°C (ranges from 0 - 8°C)[b]	ranged from 0 - 8°C[c, d]
	Preferred depth	0.1 - 1.0 m[a]	ranged from 0.31 - 0.64 m[c]
	Preferred substrate	rubble/gravel/rocks; 50 - 100 mm diameter[b]	small rocks (50 - 100 mm) and very large rocks and rubble (300-500 mm)[c]
	Preferred velocity	0.89 - 1.02 m/s[a]	1.03 m/s (ranged from 0.63 - 1.55 m/s)[c, d]
Egg Development	Temperature tolerance	0.0 - 12.0°C[a]	no data (n.d.)
	Upper limit of optimal incubation temperature	6.0°C[a]	n.d.
	Minimum oxygen concentration	8 mg/L[a]; moderate reduction in embryo survival below 35% saturation at 4°C[b]	n.d.
	Range of incubation time	36-127 days[a]; 444 degree days[b]	48 - 71 days[d]
	Recommended current velocity	in areas where the velocity will not displace spawning bed material[a]	n.d.
Rearing	Temperature tolerance	0.0 - 20.6°C[a]	n.d.
	Optimum temperature for growth	9.0 - 12.0°C[a]	n.d.
	Recommended oxygen concentration	>5.63 mg/L[a]	n.d.
	Lower lethal oxygen concentration	3.98 mg/L[a]	n.d.
	Habitat type preference	lakes, rivers, tributaries[a]; fringe or tails of pools over rock or rubble bottom; backwater areas[b]	fringes and tails of pools[c]
	Depth preference	<3.0 m[a]	<0.25 m[c]
	Preferred current velocity	slow to moderate[a]; low velocity areas along river margins; young-of-the-year will use protected side pools of small tributaries[b]	shallow backwaters[c]; side channels[f]
	Substrate	sand/gravel[a]	sand, gravel, rock and rubble[c]
	Cover	cutbanks/woody debris/aquatic vegetation[a]	n.d.

	Turbidity tolerance	<10.0 mg/L[a]	n.d.
Adult Holding and Feeding	Gradient	0.59 to 1.46%[g]	n.d.
	Temperature tolerance	0.0 - 20.6°C[a]	0 - 18°C[c]
	Optimum temperature for growth	9.0 - 12.0°C[a]	n.d.
	Recommended oxygen concentration	7.75 mg/L[a]	n.d.
	Short term minimum oxygen concentration	4.25 mg/L[a]	n.d.
	Habitat preference	rivers, streams[a]	pools, within 20 cm of the bottom[c, d]
	Depth preference	<3.0 m[c]	1.25 m in pools[c, d]
	Preferred current velocity	moderate to fast[a]; 0.13 m/s[b]	n.d.
	Substrate	gravel/cobble[a]	bedrock and boulder[c]
	Cover	cutbanks/woody debris/aquatic vegetation[a]	n.d.
	Turbidity tolerance	<10.0 mg/L[a]	highest turbidity 5.2 NTU[c]
Overwintering	Habitat preference	shallow backwaters and shallows along river margins[c]	pools with groundwater sources[h]
	Preferred depth	no data (n.d.)	<1.0 m[i]
	Substrate	n.d.	coarse substrate and boulder gardens[i]

*Information covers entire range of species in North America.

[a] Ford et al. 1995;

[b] Northcote and Ennis 1994;

[c] Thompson 1974;

[d] Thompson and Davies 1976;

[e] R.L. & L. 1994a;

[f] R.L. & L. 1995a;

[g] Stewart et al. 1982;

[h] Hildebrand 1985;

[i] Pisces 1993.

3.3.1 Spawning and Egg Development

Mountain whitefish are autumn spawners. They are generally nocturnal spawners; little else has been reported about their spawning behaviour. Mountain whitefish do not construct redds; they are broadcast spawners and can use a wide range of habitats for spawning. Although populations residing in mainstem rivers during the summer commonly move upstream into small tributaries to spawn in the fall, there is evidence of mainstem spawning, as well as tributary spawning. Lake-dwelling populations usually move into tributary streams to spawn, but in some cases spawn within the lakes. Stream spawning populations use rubble and gravel riffle areas for spawning with particle size ranging from coarse rubble to fine gravel (Northcote and Ennis 1994).

Pisces (1993) obtained indirect evidence that mountain whitefish spawned in the Brazeau River, Alberta and in some of its tributaries (i.e., based on capture of Age-0 fish). The Brazeau River was characterized by predominate cobble and boulder substrates with some gravel, bedrock, and few fines. The river was dominated by runs, riffles and rapids; several runs exceeded 1 m in depth; and a few deep pools were noted. Instream cover was low, and was usually provided by boulder substrate.

R. L. & L. (1994a) captured large numbers of mountain whitefish in spawning condition at sites on the Athabasca River. Adults were most abundant in areas with a large to medium substrate, a moderate gradient (0.9 to 1.2 m/km) and habitat dominated by shallow riffle areas interspersed between deep runs. They were concentrated in deep runs (deeper than 1.5 metres) over, or just downstream of, cobble substrates adjacent to armoured or depositional bank habitat.

Fernet and Helwig (1992) recovered mountain whitefish eggs from the substrate using an airlift pump. They reported a preference for spawning substrate ranging from large gravel (32 to 64 mm diameter) to small cobble (64 to 128 mm diameter); no eggs were found in smaller or larger substrate material. Environmental Management Associates (1981) suggested, based on the presence of young-of-the-year fish, that mountain whitefish spawned in Bolton Creek in areas where the substrate was 65 to 85 percent rubble (larger than 10 cm diameter) and 5 to 20 percent gravel (0.3 to 9.9 cm diameter). Halstead (1984) caught ripe male and female mountain whitefish on October 7 to 10, 1983 in similar habitats: 60 percent rubble and 40 percent gravel with a thin layer of silt. O'Neil et al. (1982a) observed that mountain whitefish in the Oldman River showed preference for loosely compacted substrates, with limited sedimentation of interstitial spaces; this assessment was based on densities of eggs collected with kicknets. Substrate sizes with highest densities were small cobble (50 to 100 mm) and large cobble (100 to 200 mm).

Preferred depths for spawning in streams and rivers were in the range of 0.13 to 1.22 m (Northcote and Ennis 1994). O'Neil et al. (1982a) reported depth preferences of 0.6 to 0.8 m for mountain whitefish in the Oldman River. Halstead (1984) reported spawning activity in water depths of 0.26 to 0.40 m.

Halstead (1984) recorded stream velocities in spawning areas of between 0.11 and 0.13 m/s (stream width 2 m and water depth 0.26 to 0.40 m). O'Neil et al. (1982a) reported considerably higher water velocities for spawning in the Oldman River. They reported that high quality spawning areas (i.e., those with highest densities of eggs) had surface water velocities ranging from 0.7 to 1.1 m/s. However, velocities within 0.10 m of the bottom (i.e., locations that may better reflect actual mountain whitefish spawning requirements) were considerably lower (0.4 to 0.6 m/s).

Spawning typically occurs between early September and into November, although some populations may spawn as late as January to mid-February, depending on the location (Northcote and Ennis 1994). Thompson and Davies (1976) reported that spawning in Sheep Creek, Alberta occurred between the 29 September and 18 October, with the majority of spawning activity occurring from 4 to 10 October.

In the upper Sheep River, Alberta, mountain whitefish began to concentrate in mid-September; large groups (30 to 200 fish) were located in a few pools deeper than 2 metres. The fish then dispersed to downstream spawning areas, a migration that corresponded with a drop in the minimum water temperature from 6 to 2°C during a four day period (Thompson 1974). Spawning took place in shallow and fast midstream areas over rock and rubble (5 to 50 cm diameter). There was no substrate in the spawning areas smaller than 5 cm although there was some gravel in the interstices among the rocks. The number of spawning fish in any particular area usually was between 2 and 20 (Thompson 1974).

Water temperatures at spawning time were reported to range between 0 and 11°C, but were usually in the 3 to 5°C range (Northcote and Ennis 1994). Water temperature in the Sheep River at the time of spawning was 0 to 8°C, water turbidity was 2.2 JTU and pH was 8.1 (Thompson 1974). Low temperatures are required for successful incubation of eggs and high oxygen levels are critical for successful development and survival of young (Westworth 1992). In

North Wolf Creek, springs were a main source of flow, providing open water during the winter, and water temperatures were 2 to 4°C on 7 to 10 October 1983 (Halstead 1974).

3.3.2 Rearing

Over the geographic range of mountain whitefish egg hatching has been reported to occur in the early spring, as early as March in southern locations to early April farther north (Northcote and Ennis 1994). The exact timing of egg development and emergence has not been determined; however, emergence usually coincides with ice break-up (Northcote and Ennis 1994). Formation of anchor ice and low winter discharge (20% of normal flow) in the winter of 1973-74 caused significant egg mortality in Sheep River, Alberta (Thompson and Davies 1976).

The hatch and emergence of mountain whitefish in the Sheep River in April and May was followed by an immediate downstream dispersal, in which the fry drifted passively until shallow (5 to 20 cm deep) backwater holding areas were found; young-of-the-year remained in these areas from June to August (Davies and Thompson 1976). According to Northcote and Ennis (1994), newly emerged fry may use protected side pools, and may form schools. Later in the summer and early fall, fry inhabit side channel riffle habitat. Larval mountain whitefish dominated the catch at all sites on the upper Athabasca River within Jasper National Park in the spring 1994 (R.L. & L. 1995a). These areas may be important nursery areas for young-of-the-year mountain whitefish. Habitats with the highest densities were shallow, low velocity runs and pools along the mainstream margin. Until they reach about 55 mm in length, young mountain whitefish remain in these areas. By mid-August they may move away from the lateral habitat and into deeper sections of the stream. There is relatively little specific information on the habitat use of yearling fish. They appear to use largely mainstem riffles and runs but are sometimes found in pools (Northcote and Ennis 1994).

R. L. & L (1994a) captured young-of-the-year mountain whitefish in October in side channel areas of the lower Snake Indian River and Snaring River, tributaries to the upper Athabasca River. At another sampling site on the Athabasca River, they were most abundant over silt substrate in areas that contained woody debris.

According to Fernet and Helwig (1992), mountain whitefish young-of-the-year (less than 100 mm fork length in September) showed a preference for large cobble (128 to 256 mm diameter) and small boulder (256 to 762 mm diameter) substrates and indicated no preference for substrates smaller than medium gravel (8 to 32 mm). However, Thompson (1974) found young-of-the-year in water less than 25 cm deep over fine gravel, sand or mud substrates.

Juveniles (100 to 250 mm fork length) showed a preference for habitats with large boulders (i.e., greater than 762 mm in diameter). They also demonstrated a strong preference for small boulder habitats and a secondary preference for medium gravels (Fernet and Helwig 1992). Thompson (1974) also reported that juveniles longer than 55 mm tended to move into faster water and were found over rock and rubble substrates, often in the fringes and tail ends of pools, where there was a noticeable increase in water velocity. Environmental Management Associates (1981) found young-of-the-year in Bolton Creek where 65 to 85 percent of the substrate was larger than 10 cm diameter.

The habitat was a good mix of riffles, runs, and pools with about 1.5 to 10 percent of the stream area consisting of pools deeper than 0.9 metres. Pisces (1993) noted similar habitat in the Brazeau River, where deep runs and riffles over cobble and boulder substrates provided high quality habitat for juveniles.

R.L. & L. (1994a) reported that juvenile mountain whitefish were abundant in the Athabasca River and in the Lower Snaring and Snake Indian rivers where they preferred predominately cobble substrates adjacent to armoured banks. Juveniles were found in similar habitats as the adults; however, they preferred shallower water, usually less than 1 metre deep. They were also abundant at sampling sites at the mouths of tributaries.

3.3.3 Adult Feeding and Holding

In the spring, mountain whitefish move from overwintering areas (often in the deeper mainstem river) to feeding sites in creeks and tributaries, and then to summer feeding sites in the rivers, as small streams become too warm or flows become low (Westworth 1992).

In some streams, older age classes prefer pools at depths greater than 90 cm, in others they are found more often in mainstem runs and riffles. Mature fish may inhabit the upper reaches of rivers from late spring until autumn, but generally overwinter in deep pools in the lower reaches (Northcote and Ennis 1994).

Mountain whitefish adults (fork length exceeding 250 mm) showed a strong preference for small and large boulder habitats, and made little use of areas with smaller substrates (Fernet and Helwig 1992).

In Adsett Creek, a second-order tributary in the Liard River, adult mountain whitefish were abundant in locations where the substrate consisted of 10 to 25 percent sand, 20 to 60 percent gravel, 10 to 30 percent cobble, and 5 to 40 percent boulder. The stream was characterized by numerous, steep riffles and well-defined, deep runs and riffle sections. Instream cover was provided by small log jams and overhangs. The density of mountain whitefish adults in the stream was 0.2 to 0.3 fish/100 m^2 (Stewart et al. 1982). The authors suggested that mountain whitefish were more abundant in the lower reaches of the stream because these reaches offered more favourable physical habitat which provided protection from predators and extreme summer conditions. In addition, the lower reaches provided a more abundant or diverse benthic community, the primary food source for mountain whitefish.

Pisces (1993) reported captured mountain whitefish from the Brazeau River and several tributaries where the substrate was predominately cobble and boulder. Adults were caught almost exclusively from run habitats. The water velocity was moderately high and instream cover was low; cover was provided by woody debris and the coarse substrate. O’Neil and Hildebrand (1986) observed similar habitat preferences in the Castle and Oldman rivers. In these systems, mountain whitefish exhibited the strongest preference for moderately deep (i.e., 0.75 to 1.0 m) runs, run/boulder gardens, and rapid/boulder gardens, although they were also associated with shallow (i.e., <0.75 m) run/boulder gardens.

Renewable Resources (1971) reported that mountain whitefish in the Pembina River, Alberta showed a definite preference for riffles with transparent water, high current velocity and a clean rubble substrate. These investigators suggested that mountain whitefish were less abundant in the lower reaches of the Pembina River where high turbidity levels and siltation reduced the quality of riffle habitats.

Thompson (1974) observed adult mountain whitefish in the Sheep River during September, 1973. Groups of between 3 and 50 fish were recorded in pools deeper than 1 m and with widths up to 4.9 m wide. While holding, the fish generally remained within 2 to 10 cm of the bottom. The stream bottom in these areas was about 40 percent boulder and 60 percent bedrock; the substrate was described as being typical of pools situated in narrow, deep gorges which were scoured by spring runoff and heavy rains.

R.L. & L. (1994a) reported that adult mountain whitefish in the Athabasca River were most abundant in moderate to low gradient reaches with predominately cobble to boulder substrates. Most often adults were found in water deeper than 1.5 metres adjacent to armoured and depositional banks where cover was provided by boulders.

Stewart et al. (1982) reported that Adsett Creek had the following characteristics:

- stream gradient: 0.59 to 1.46 percent;
- channel width: 6.9 to 8.9 m.;
- channel depth: 0.29 to 0.49 m.;
- discharge: 1.09 to 1.61 m^3/s in May but below detection limits at times in August and September; slightly basic, dissolved oxygen: 9 to 11 mg/L;
- turbidity high during the spring freshet in May (48 to 75 JTU) but falling to 7 to 17 JTU through August;
- total suspended solids usually below detection; and
- water temperature rising to a maximum of 19.5°C in July.

Thompson (1974) reported turbidities in the Sheep River of 5.2 NTU between June and November with higher (unmeasured) levels during spring runoff. Water temperatures reached a maximum of 18°C with diurnal fluctuations of up to 9°C.

3.3.4 Overwintering

Westworth (1992) reported that, after spawning, mountain whitefish move downstream to the lower reaches of rivers to overwinter. Age-0+ and Age-1+ fish probably overwinter in shallow backwaters or the shallows along the river margins or pools (Thompson 1974).

Hildebrand (1985) observed numerous mountain whitefish juveniles and sub-adults in pools in Beaver Creek, a tributary to the Berland River in March 1985. Parts of this creek were kept ice free by groundwater inflow from an extensive network of seep-hole springs and surface seepage. There were numerous deep pools (0.75 to 1.0 m deep) in the creek with substrates composed of silt with the occasional boulders. The ice-free site was predominately low gradient, flat habitat (60 percent) with runs (20 percent) and pools (20 percent). The water temperature in the mainstem was 0.5°C, whereas in the spring inflow it was 2.0°C. Dissolved oxygen levels were 11.7 mg/L in the mainstem and the pH was 6.8. The stream was an entrenched, meandering channel flowing through a wide, open muskeg flat. The banks were about 2 m high, steep, and mainly stable with extensive grass and shrub cover. Instream cover was provided by bank undercuts, overhanging vegetation and deep water.

Pisces (1993) noted the presence of good quality overwintering habitat for mountain whitefish in the Brazeau River. They characterized the habitat as primarily riffles, rapids and runs, some with water depths of more than one metre, over predominately cobble and boulder substrate. Cover was provided primarily by boulder gardens and coarse substrates.

3.4 ARCTIC GRAYLING

Hubert et al. (1985), Armstrong (1986) and Northcote (1995) have summarized information on the habitat requirements of Arctic grayling. Other investigators have presented their findings on individual populations in Alberta, British Columbia, Northwest Territories, Saskatchewan and Yukon. Table 3.6 provides a summary of the habitat requirements for the different lifestages of Arctic grayling.

3.4.1 Spawning and Egg Development

Adult grayling leave their overwintering areas and enter bog-fed or unsilted rapid runoff rivers to spawn, but also spawn in mainstem rivers, in large and small tributaries to rivers and lakes, in intermittent streams, and in lakes - usually at the mouths of tributaries (Armstrong 1986). Only in large, unsilted, rapid runoff rivers do grayling spawn in the same river in which they overwinter. Migrations to these areas are associated with a rise in water temperature to about 1°C (Armstrong 1986, Bishop 1967). Machniak and Bond (1979) reported that spawning migrations in the Steepbank River watershed were well under way, by the time ice conditions on the river were suitable for sampling, in mid April. Movements into spawning areas were almost complete by the end of April, although small numbers of fish were captured moving upstream throughout May. Bond and Machniak (1979) indicated that in the Muskeg River, the spawning migration was under way by the first sampling date of April 28 and continued until early June, although the bulk of the migration occurred prior to the middle of May.

O’Neil et al. (1982b) monitored an upstream spawning migration of Arctic grayling in Hartley Creek, a tributary of the Muskeg River (Athabasca River drainage) during spring 1981. In total, 904 Arctic grayling were recorded moving upstream during the period 2 May - 18 May. The majority (90%) were captured between 3 May and 11 May; the water temperature (daily maximum) ranged from 6 to 10°C during this period. This population had travelled from the Athabasca River where overwintering occurs, prior to being intercepted at the trap installed in Hartley Creek.

Hildebrand (1981) recorded upstream spawning movements of Arctic grayling in Dismal Creek, a tributary of the Pembina River (Athabasca Drainage). The spawning run occurred between 25 April and 4 May 1981 with peak movements (80% of grayling capture) taking place between 30 April and 3 May. The trap was located 40 km upstream of the Dismal Creek-Pembina River confluence; water temperatures during the trapping period ranged from 0 to 5.5°C.

Table 3.6 Summary of habitat use by life requisite function for Arctic grayling.

Life Requisite Function	**Habitat Component**	**Geographic Range in North America***	**Foothills Model Forest and other Relevant Transferrable Information**
Spawning	Spawning habitat	small gravelly tributaries[a]	no data (n.d.)
	Preferred temperature	7 - 10°C[a]	6 - 10°C[c]
	Preferred depth	variable[a]; 0.10 - 0.40 m[b]	0.15 - 0.42 m[d]
	Preferred substrate	gravel (<20% sand)[a]; coarse gravel (20-40 mm); 10-20% fines, 30-80% gravel, 10-50% cobble/boulder[b]	medium gravel to small cobble (9 to 128 mm)[d]
	Preferred velocity	0.3 - 1.5 m/s[a]	0.35 to 0.55 m/s[d]
Egg Development	Temperature tolerance	2 - 16°C[a]	n.d.
	Optimal incubation temperature	6 - 10°C[a]	n.d.
	Range of incubation time	8 - 32 days[a]; 156 - 268 degree days[b]	n.d.
	Recommended current velocity	<0.3 m/s[a]	n.d.
	Substrate	gravel/boulder (<20% sand)[a]; coarse gravel (20 - 40 mm), 10-20% fines, 30 - 80% gravel and 10-50% boulder[b]	n.d.
	Cover	1-3 cm into the gravel[a]	medium gravel to small cobble (0.9 to 12.8 cm)[d]
Rearing	Temperature tolerance	2 - 24.5°C[a]	n.d.
	Optimum temperature for growth	10 - 12°C[a]	n.d.
	Recommended oxygen concentration	no data (n.d.)	n.d.
	Lower lethal oxygen concentration	1.4 mg/L[a]	n.d.
	Habitat type preference	streams[a]; back channels, side channels, riffles[b]	n.d.
	Depth preference	<0.50 m[a]	n.d.
	Preferred current velocity	<0.5 m/s[a]	n.d.
	Substrate	gravel/cobble/sand[a]	gravel, rubble, boulders[e]
	Cover	boulders[a]	aquatic vegetation, overhanging vegetation[e]
	Turbidity Tolerance	<50 mg/L[a]	n.d.
Adult Holding and Feeding	Temperature tolerance	1 - 20°C[a]	n.d.
	Optimum temperature for growth	10°C[a]	n.d.
	Recommended oxygen concentration	n.d.	n.d.
	Lower lethal oxygen concentration	2.0 mg/L[a]	n.d.
	Habitat preference	rivers/lakes[a]	upstream end of pools below riffles[f]
	Depth preference	<10 m[a]	1 m[f]
	Preferred current velocity	0.2 - 0.8 m/s[a]	n.d.
	Substrate	gravel/rocks/boulders[a]	n.d.
	Cover	boulders[a]	n.d.
	Turbidity Tolerance	effects noted after short term exposure to >100 mg/L[a]	n.d.
Overwintering	Groundwater	important for maintaining ice-free areas[f]	n.d.
	Preferred depth.	<1.2 m[g]	n.d.

	Preferred velocity	>0.15 m/s[g]	n.d.
	Preferred habitat	pools[g, h]	n.d.

*Information covers entire range of species in North America.

[a] Ford et al. 1995;

[b] Northcote 1995;

[c] O'Neil et al. 1982b;

[d] R.L. & L. 1996;

[e] Halstead 1984;

[f] Bond and Machniak 1979.

[g] Hubert et al. 1985;

[h] Stewart et al. 1982.

In Sundance Creek, a tributary of the McLeod River, a small number of adult Arctic grayling were recorded moving upstream, between 27 April and 9 May 1993 (R.L. & L. 1995d); water temperatures (daily maximums) varied from 4 to 12°C during this period. The fish trap was located approximately 1 km upstream from the McLeod River.

Arctic grayling spawning movements were monitored in the House River (tributary of the Athabasca River) and a 3rd order tributary of the House River (West Bear Creek) in spring 1995 (R.L. & L. 1996). The peak of upstream movements past fish traps on the House River and West Bear Creek occurred on 6 May (mean water temperature of 7.2°C). The fish trap on West Bear Creek was located 2.6 km upstream of the House River; the House River trap was located approximately 123 km upstream of its confluence with the Athabasca River. The investigators were not able to determine whether the Arctic grayling intercepted at the traps had originated from overwintering areas in the mainstem House River or had travelled from the Athabasca River.

Most authors reported spawning in the May to mid-June period (Armstrong 1986, Bishop 1967, Butcher et al. 1981, Pendray 1983, Stewart et al. 1982, Ward 1951) although spawning has been recorded in late April and early July (Armstrong 1986). R.L. & L. (1995d) collected fertilized Arctic grayling eggs in Sundance Creek, a tributary of the McLeod River, on 12 May 1993. Based on the upstream movement pattern and gonad maturity of adults during the spring and the state of development of the eggs, it was concluded that spawning occurred during the second week of May. Water temperature in the stream ranged from 8 to 12°C (daily maximum) during the spawning period.

Spawning in western Alaska seems to begin when the water temperature is 4°C (Armstrong 1986) and in the Yukon and northern B.C. at 5°C (Pendray 1983, Stewart et al. 1982). Hildebrand (1981) monitored the Arctic grayling spawning migration in Dismal Creek, a tributary of the Pembina River (Athabasca River Drainage) spawning in this system commenced in early May. Spawning in the House River occurred in early May when the water temperature was between 6 to 10°C (R.L. & L. 1996). Stuart and Chislett (1979) recorded spawning in the Sukunka River drainage (northeastern B.C.) in early June when water temperatures were between 5 and 10.5°C.

The age of attainment of sexual maturity varies with location. Grayling in the Upper Big Hole River (Liknes 1981) and in the southern Athabasca drainage (Ward 1951) matured at Age-3. Butcher et al. (1981) reported that grayling in the Liard River drainage matured at Age-4+; Bishop (1967) and Pendray (1983) indicated that the majority of grayling were mature at ages 6 or 7; Bishop (1967) stated that fish 6 to 9 years old made up 93.5 percent of the spawning runs in tributaries to Great Slave Lake, N.W.T. In Alberta Arctic grayling populations, sexual maturity appears to be attained at Age-3, with a variable proportion of the males entering the spawning population at Age-2 (O'Neil et al. 1982b, Bond and Machniak, 1977, Bond and Machniak, 1979, R.L. & L. 1996, Ward 1951). Spawning runs in these systems were generally made up of fish in the Age-2 to Age-7 cohorts.

Hubert et al. (1985) described suitable spawning substrate as being gravel and rubble in the 10 to 200 mm diameter range. Furthermore, to provide optimum conditions, these authors determined that at least 20 percent of the substrate should be of this size and that the amount of fines (<3 mm) in spawning areas and during embryo development should be less than 10 percent. In the Fond du Lac River, Saskatchewan, fine and coarse gravel (5 to 76 mm) were selected for spawning (Kratt 1977). Armstrong (1986) reported that in Alaskan rivers and streams, Arctic grayling spawn in riffle areas with pea-sized gravel (0.075 to 38.1 mm). He also reported spawning in slow, shallow backwater areas, in lakes over substrates ranging from large rubble to vegetated silt, and among sedges over an organic bottom in a nearly stagnant pond, and over mud in a slough (Armstrong 1986). Butcher et al. (1981) recorded spawning in pools and glides below culverts in British Columbia streams. However, given an alternative, only gravel areas were used: pure mud, silt or clay were not chosen as spawning areas (Bishop 1967).

Ward (1951) noted that redds were not constructed and that female Arctic grayling deposited their eggs in the interstices between stones (75 to 250 mm diameter) or in the gravel areas (15 mm to 25 mm diameter). According to Stewart et al. (1982), Arctic grayling preferred stable, coarse gravel (20 to 40 mm in diameter) and spawning took place at the tails of long runs, particularly those preceding localized steep gradient sections. Halstead (1984) captured spent grayling in Calahoo Creek, Alberta in areas where substrate was comprised of 19 to 35 percent gravel, 20 to 34 percent rubble, 22 to 49 percent boulder and 9 to 12 percent fine sediment. R.L. & L. (1996) described microhabitat conditions at several Arctic grayling spawning sites (verified by collection of fertilized eggs) in the House River and tributaries. They noted considerable variability between individual sites with respect to substrate size and composition, although there was greater utilization of substrates in the medium gravel to small cobble (9 to 128 mm diameter) size range. In Hartley Creek, spawning was confirmed in a wide range of coarse substrates, but egg densities were highest in areas featuring loosely to moderately compacted material in the 15 to 200 mm diameter size range (O’Neil et al. 1982b).

According to Hubert et al. (1985), preferred spawning depths were deeper than 0.30 m and had mean column velocity (average at 0.6 of the water depth) between 0.25 and 0.50 m/s. In a review of the Alaskan literature, Armstrong (1986) indicated that spawning in riffle areas took place in 0.20 to 0.70 m deep water, with current velocities ranging from 0.30 to 1.50 m/s. In Providence Creek, N.W.T., Arctic grayling spawned over gravel substrates at depths of 0.90 m and velocities of 0.70 m/s; the site was located in a deeper part of the stream that was just below a riffle feeding area (Bishop 1967). In Adsett Creek, British Columbia, Arctic grayling spawned in areas with depths of 0.10 to 0.44 m and velocities varying from 50 to 100 cm/s (Stewart et al. 1982). Butcher et al. (1981) reported spawning in pools as deep as 2.25 m in tributaries to the Liard River. Kratt (1977) reported a much wider range of depths and velocities for spawning in northern Saskatchewan. He recorded spawning at depths between 0.1 and 1.3 m, with most spawning fish being concentrated very close to shore where the water depth was less than 1.0 m. Water velocities in spawning areas ranged from 0 to 1.40 m/s. R.L. & L. (1996) summarized the depth and velocity characteristics at 11 fertilized egg collection sites on the House River and tributaries. Most sites were located in shallow water (i.e., mean depths at sites ranged between 15 and 42 cm), and average nose velocities (velocities taken 10 cm off the bottom) were typically between 35 and 55 cm/s. With respect to site selection, there was a preference for the “break” areas between a run and downstream (or laterally) situated riffle. They noted that one site was located immediately below a breached beaver dam.

3.4.2 Rearing

Embryonic development in Arctic grayling is rapid; at a water temperature of 8°C embryos became eyed in 14 days and hatched in 18 days (Armstrong 1986). Wojcik (1955) reported that grayling eggs hatched in 8 days at an average temperature of 15.5°C. Bishop (1967) determined that artificially fertilized Arctic grayling eggs hatched in 13.7 days at an average water temperature of 9.1°C. Kratt (1977) recorded longer incubation periods (25-32 days) in the Fond du Lac River, Saskatchewan. He attributed this to the lower water temperatures in the area. Kratt and Smith (1977) indicated that Arctic grayling eggs hatched in 186.24 degree-days. Alevins remain in the gravel 3 or 4 days after hatching (Armstrong 1986 and Kratt 1977) and young Arctic grayling began to feed as late as nine days after hatching (Bishop 1967). Stuart and Chislett (1979) observed fry leaving the substrate 24 days after spawning.

Armstrong (1986) reported that fry are helpless in the water current for about 2 weeks after hatching and are susceptible to being washed downstream by flooding. After hatching young Arctic grayling remained for several weeks in quiet water areas, including "very shallow riffles between rocks at the lower end of gravel bars, backwaters, in side channels and in quiet side pools, and brushy or grassy areas of adjacent sloughs". Later in the summer they became solitary and moved into deeper water. Butcher et al. (1981) reported a passive downstream movement of fry in Geddes Creek; within 2 weeks of emergence no more fry were caught in the creek. Kratt (1977) also reported passive downstream movements following emergence. He noted that fry could be found inhabiting calm sheltered areas along the stream banks; however, fingerlings had moved to more swiftly flowing water by mid-August.

Stewart et al. (1982) reported similar behaviour and noted that 42 days after emergence, fry were inhabiting virtually all available habitat. The substrate in the reach with the greatest abundance of rearing grayling was: 10 to 30 percent sand and silt; 20 to 60 percent gravel; 10 to 40 percent cobble; and 5 to 40 percent boulder. There were numerous steep riffles, and well-defined, abundant deep runs and riffle pockets. Instream cover was provided by a few small log jams and overhanging banks.

Stuart and Chislett (1979) reported that in the Sukunka River watershed there was a general movement of fry from backwater rearing areas into riffles. They indicated that the fry showed a preference for shallow riffles with silt-free gravel substrate.

Hubert et al. (1985) indicated that areas of depth between 0.09 and 0.60 m and mean column velocity less than 0.15 m/s were optimal for newly emergent fry. Fry less than 5 cm in length preferred cobble (64 to 250 mm diameter) and boulder (250 to 400 mm diameter) substrates. These areas were preferably located downstream from the spawning areas and had areas of backwater and side channels with a current velocity less than 0.15 m/s and more than 30 percent of the area as pool habitat (Hubert et al. 1985). Juveniles (20 to 38 cm in length) preferred similar habitat to young-of-the-year but with water depths exceeding 0.18 m and water velocity less than 0.18 m/s (Hubert et al. 1985). Stewart et al. (1982) reported depths of 0.29 to 0.53 m in sections of Adsett Creek where juvenile Arctic grayling were abundant.

According to Hubert et al. (1985), the optimal water temperature for rearing of fry and juveniles was between 8 and 16°C. Stewart et al. (1982) reported maximum water temperatures in Adsett Creek of 19.5°C in July. Dissolved oxygen levels were 9 to 11 mg/L and the total suspended solids were usually below detection limits; turbidity was highest (48 to 75 JTU) during the spring freshet.

Pendray (1983) estimated rearing populations and densities in 18 kilometres of usable channel length in Lynx Creek, Yukon at 11,000 fish or 0.12 fish/m^2. He suggested the relatively high abundance of Arctic grayling in Lynx Creek, compared to other adjacent creeks, may have been the result of several factors: proximity to suitable overwintering areas, channel alterations due to placer mining in other, nearby streams; and higher turbidity and water temperatures in the other streams. The lower reaches of Lynx Creek had a gradient of 0.85 percent, an average width of 5 to 6 m, a discharge of more than 3.4 m^3/s during high flows in May, very low turbidity (12 mg/L during run-off in May), and temperatures that in late June reached a maximum of 12°C. The stream had a meandering, stable channel and 60 to 70 percent pool/glide habitat.

Halstead (1984) reported Arctic grayling young in Calahoo Creek at densities of 10 to 59.4 fish/100 m^2 in September. This stream had a gravel, rubble, boulder substrate, abundant aquatic vegetation covering 16 to 18 percent of the stream bottom, and 33 to 42 percent of the riparian cover as understory. Temperature was 8°C, pH 8.0 to 8.2, turbidity 5 to 11 JTU, and total suspended solids 13.6 to 27.6 mg/L.

3.4.3 Adult Feeding and Holding

According to Hubert et al. (1985), adult Arctic grayling preferred deeper water (up to 1.36 m in depth) and current velocities less than 0.22 m/s. They also reported that the average maximum water temperature during the warmest period of the year in large streams and rivers inhabited by adult Arctic grayling was between 7.5 and 16°C and the minimum dissolved oxygen levels during the summer, low-flow period exceeding 6 mg/L.

Bond and Machniak (1979) reported anecdotal information reported by anglers, that Arctic grayling in the Muskeg River were found in areas of water up to 1 m in depth, with moderate water velocities over sand substrates, and were often associated with macrophyte beds. They also reported that in the canyon portion of the Muskeg river, Arctic grayling were frequently found near the upstream end of pools, just below riffles. Stuart and Chislett (1979) reported definite preference for pools by Age-2+ Arctic grayling.

Liknes (1981), in studies in the Upper Big Hole River and its tributaries, reported that Arctic grayling were most abundant (35 fish in the 25 to 29 cm length range per kilometre) in a section of the river that had the following characteristics: mean depth 0.28 m, mean width 12.21 m, mean velocity 0.21 m/s, and a gradient of 0.29 percent. Suitable spawning substrates were present, aquatic vegetation was abundant, as was cover in the form of overhanging vegetation, debris, and undercut banks. The substrate in this reach consisted of 84 percent rubble and gravel, 15 percent fines, and 1 percent boulder. The mean riffle width was 10.83 m and pool width was 8.95 m. The pool -

riffle periodicity was 6.3 and the pool to riffle ratio was 1.5:1. Water temperature in the stream ranged from 0 to 24°C (April 21, 1979 to September 1, 1979) with a mean temperature of 13.7°C, pH of 7.22 to 7.64 and dissolved oxygen of 7.7 to 10.2 mg/L, averaging 8.7 mg/L (Liknes 1981).

Migrations of adults to feeding areas from overwintering or spawning areas may depend on age, type of system in which they overwinter, and the type of stream in which they spawn. Arctic grayling that migrate to feed in unsilted, rapid-runoff rivers tend to go to different areas: juveniles to the lower reaches, the sub-adults (ages 4, 5, and sometimes 6) to the middle portions and post-spawning adults to the upper reaches. Arctic grayling probably return to the same river annually to feed (Armstrong 1981). Hughes and Reynolds (1994) noted a similar distribution of Arctic grayling in interior Alaska streams (i.e., the size of Arctic grayling increased with distance travelled upstream). While they could not provide a definitive reason why larger Arctic grayling preferred upstream areas, their experiments did indicated that all sizes of Arctic grayling prefer positions in the headwaters, but that large fish defend these areas, forcing the smaller ones to occupy positions farther downstream. Unlike many other studies, Kratt (1977) did not demonstrate much seasonal movement; although, tag and recapture studies indicated that Arctic grayling moved down into Black Lake to overwinter and returned to the river in the spring to feed.

3.4.4 Overwintering

Early in the fall, young-of-the-year and adult Arctic grayling usually leave the headwater areas, small streams, and tributaries, and enter larger rivers, lakes, or spring fed areas for overwintering. They leave the bog-fed tributaries, most of which freeze solid or dry up in the winter (Armstrong 1986). Most also leave spring-fed tributaries, possibly because of the formation of frazil ice in these areas. Machniak and Bond (1979) reported that downstream migrations in the Steepbank River began in early October and were much more concentrated than the upstream migration in the spring, lasting only approximately three weeks. The peak of downstream activity occurred within four days of the onset of migration. In the Arctic, overwintering areas contain extensive groundwater sources because the rest of the stream may freeze completely to the bottom. The distance Arctic grayling migrate to reach overwintering areas varies considerably from one system to another (Armstrong 1986).
Bond and Machniak (1979) indicated that in both the Muskeg and Steepbank rivers young-of-the-year remain in the tributaries for their first winter and do not join the migrant population until the autumn of their second year. Both young-of-the-year and juveniles were reported to overwinter in Sundance Creek (R.L. & L. 1993).

Arctic grayling overwinter in large streams with pools more than 1.2 metres deep and current velocities less than 0.15 m/s. Arctic grayling will also overwinter in spring-fed smaller streams in reaches that do not freeze solid and in which the dissolved oxygen levels exceed 1.0 mg/L (Hubert et al. 1985).

Stewart et al. (1982) believed that yearling and Age-2 fish overwintered in pools created by beaver dams on Adsett Creek, which were 6.9 to 8.9 m wide (mean wetted width), 0.29 to 0.53 cm deep in most areas, and had a discharge below detection limits at times in August and September. Environmental Management Associates (1981) suggested that Arctic grayling overwinter in Bolton Creek, which was about 15 m wide and 0.35 m deep with 2 to 10 percent of the stream area being pools deeper than 0.9 m. The substrate in this stream was 60 to 85 percent rubble of a size greater than 10 cm in diameter.

4.0 REVIEW OF EXISTING HABITAT SUITABILITY MODELS

One of the tasks of the present study was to “identify existing habitat models for the four target species and evaluate these for adaptation to the Alberta environment.” The following sections will describe models which are currently available for the target species. Following the discussion of habitat models, information on alternatives to habitat modelling will be presented.

4.1 HABITAT SUITABILITY INDEX (HSI) MODELS

The development of HSI models, by the U.S. Fish and Wildlife Service, was initiated in the late 1970's and continued until the late 1980's. The HSI models were developed as part of the Habitat Evaluation Procedures (HEP), a planning and evaluation technique developed by the U.S. Fish and Wildlife Service that focussed on the habitat requirements of fish and wildlife species. The HSI models provide tools to evaluate the impacts on fish and wildlife from changes in water or land use practices (Schamberger et al. 1982).

Initially, all of the HSI models were published as part of the “blue book” series of reports produced by the Biological Service Program and Division of Ecological Services, of the U.S. Fish and Wildlife Service. In the early 1990's the National Biological Service was directed to stop producing HSI models and to allocate more effort to field trial testing of the models. Although some models continue to be produced, they are no longer published by the U.S. Fish and Wildlife Service. Consequently, any new models developed, or the results of field validation trials, are scattered throughout various scientific journals (Jim Terrell, National Biological Service, U.S. Geological Service, Fort Collins, Colorado, pers. com. February 1996).

The models contain references to numerous literature sources and whenever possible, relationships were derived from site specific population and habitat models. Models provide an index of habitat suitability on a numerical scale (0.0 to 1.0, where 0 indicates unsuitable habitat and 1 indicates optimum habitat) (Terrell et al. 1982). The models are based on the assumption that a positive relationship exists between the index and habitat carrying capacity (Schamberger et al. 1982).

The models can be simple or sophisticated depending on the extent and specificity of data available to support the variables and relationships in the model. The HSI is generally presented in three formats (Schamberger et al. 1982):

- graphic - the structure of the model and sequential aggregations of the variables in the HSI are presented;
- word - the model relationships and the assumed relationships between variables, components and other HSI’s are discussed; and
- mathematical - the model relationships discussed in the word description are presented in mathematical form.

According to Schamberger et al. (1982) it is important that these models be considered as a hypothesis of species-habitat relationships rather than positive proof of cause and effect relationships. While the model performance may have been evaluated and found to be adequate in specific locations, they may not be transferable to other situations and must be used carefully. Terrell et al. (1982) indicated that HSI models can be used as presented or modified according to need. However, the assumptions, limitations, accuracy, and data requirements of each HSI model need to be evaluated relative to the objectives and constraints of the individual Habitat Evaluation Procedures (HEP) application prior to selecting a particular HSI model. Habitat Suitability Index models must consist of habitat variables (Terrell et al. 1982):

- whose importance to the species to be evaluated can be documented;
- that are quantifiable; and
- whose values can be measured or predicted under various habitat conditions in the present and if required, in the future to meet project goals.

The three types of HSI models developed by the U.S. Fish and Wildlife Service can also be described as (Terrell et al. 1982):

- regressions models which predict a measurable response, such as standing crop or harvest from the environmental variables;
- mechanistic models that describe the suitability index ratings for individual habitat variables and sum the ratings into the HSI that is based on hypothesized causal relationships between variables and habitat suitability; and
- descriptive models that assign HSI values based on the presence or absence of specific habitat parameters.

The existing models for rainbow trout and Arctic grayling which are relevant to this study are mechanistic models. More details regarding the development of descriptive models, as well as the other two types of models can be found in Terrell et al. (1982).

Terrell et al. (1982) acknowledged that one of the difficulties with HSI models is the difficulty in measuring all of the habitat variables in the models or measuring the variables in the indicated frequency. These authors suggested that this problem can be overcome by modifying the measurement technique. As an example, they suggested that if the model requires mean monthly turbidity, which can be expensive to determine, the users of the model may determine that one or only a few turbidity measurements may be sufficient. Another suggestion was to use available data from similar waterbodies nearby.

The development of the HSI models was to have been a two part process. The first part was to develop the HSI models, as published in the blue book series, which summarized all the known relationships for a particular species. The second phase was intended to examine the relationships and attempt to simplify the models by selecting the key habitat parameters and deciding when to collect the information. This second phase was not completed due to the directive for the National Biological Service to begin field testing the HSI models, (Terrell, pers. com., February 1996).

4.1.1 Rainbow Trout

Raleigh et al. (1984) indicated that the HSI model for rainbow trout was intended for application throughout the geographic range of rainbow trout. There are five components to the HSI model (Figure 4.1): adult (C_A); juvenile (C_J); fry (C_F); embryo (C_E) and other (C_O). Each of the lifestage components contains habitat variables which are specific to that component. The component C_O contains the habitat variables which are common to all life stages of development (i.e., water quality and food supply).

The model employs a limiting factor procedure which assumes that model variables with suitability indices in the average to good range (i.e., >0.4 and <1.0) can be compensated for by higher suitability indices of other related model variables and components. However, variables with suitability indices of ≤0.4 cannot be compensated and as such become limiting factors (Raleigh et al. 1984). A complete description of the reasoning and assumptions for inclusion of the different variables within each life stage component was provided by Raleigh et al. (1984). How the model is utilized to determine an HSI rating is described in Appendix A1.

The model is intended to provide a relative indication of habitat suitability for individual life stages, composite life stages or for the species as a whole. The intent is not to predict standing crops (i.e., numbers or kg of fish/hectare). The factors which limit standing crop (i.e., interspecific competition, predation, disease, water nutrient levels, length of growing season) have not been factored into the rainbow trout HSI model. The model contains the physical habitat variables which are deemed to be important in maintaining viable populations (Raleigh et al. 1984).

A high HSI score for habitat indicates that the habitat being evaluated is near optimal for the factors included in the model. Intermediate scores indicate average conditions are present and low HSI scores indicate poor habitat conditions. An HSI result of 0 does not infer absence of rainbow trout from an area, instead it indicates that the habitat is very poor and that rainbow trout are likely to be absent or scarce.

4.1.2 Arctic Grayling

Hubert et al. (1985) developed an HSI model for freshwater riverine habitats throughout the geographic range of Arctic grayling. The HSI model has two components (Figure 4.2; Appendix A.2.1):

- spawning and embryo development (A_1); and
- migratory and wintering habitat for juvenile and adult life stages (A_2).

As with the rainbow trout model, the Arctic grayling model was designed to produce an index value between 0 and 1. Hubert et al. (1985) indicated that the index is positively correlated with the suitability of habitat.

Figure 4.1 Habitat suitability index model for rainbow trout.

Figure 4.2 Habitat suitability index model for Arctic grayling.

The HSI model for Arctic grayling is not intended to predict standing crops of fish, as such factors limiting standing crop (i.e., interspecific competition, predation, angler exploitation) are not included in the model. The model includes physical habitat variables which are considered by Hubert et al. (1985) to be important in maintaining viable populations of Arctic grayling in North America. A high HSI score for habitat indicates that the habitat being evaluated is near optimal for the factors included in the model. Intermediate scores indicate average conditions are present, and low HSI scores indicate poor habitat conditions. An HSI result of 0 does not mean that Arctic grayling are not present, but that the habitat is very poor and that Arctic grayling are likely to be absent or scarce.

O'Neil et al. (1982b) applied a modified Arctic grayling model (Figure 4.3) to determine habitat suitability within various reaches of Hartley Creek, Alberta for Arctic grayling and to quantify the impacts of a diversion scheme on the fishery (Appendix A.2.2). This model was successful in predicting reaches which had a higher value for spawning and adult feeding.

4.1.3 Mountain Whitefish

A simple, descriptive HSI model for mountain whitefish was prepared by O'Neil et al. (1982c). The suitability indices developed by O'Neil et al. (1982c) were based on habitat utilization data for mountain whitefish collected in the Liard River system in northeastern British Columbia over several years.

The suitability indices developed by O'Neil et al. (1982c) were based on extensive data gathered from both published and unpublished sources of information. This model, which was developed for a large river system in northern British Columbia, is not applicable to streams in the Foothills Model Forest area, but it may have potential for application to the Athabasca River. It may be a useful starting point for developing similar models for smaller systems in the study area.

4.1.4 Bull Trout

A literature search failed to locate habitat suitability models for bull trout in the published scientific literature, however, a model is currently being developed by a graduate student at Montana State University (Bozeman), which utilizes basic habitat information that can be determined in the field or from topographic maps (e.g., stream gradient) to predict the presence or absence of bull trout. The thesis for which this model is being developed was to be published by May 1996 (Dr. Tom McMahon, Montana State University, Bozeman, Montana, pers. com. February 1996).

4.2 OTHER HABITAT SUITABILITY MODELS

Several other types of models have been develop for predicting the suitability of habitat to support trout populations. These models were developed independently from the U.S. Fish and Wildlife Service HSI/HEP program.

Figure 4.3 Modified habitat suitability index model for Arctic grayling.

Binns and Eiserman (1979) developed a Habitat Quality Index (HQI) to predict the standing crop of trout (brown, rainbow, brook and cutthroat) in Wyoming streams. Standing crop was then used to infer the quality of fish habitat in a stream: that is, the higher the standing crop the better the habitat. The HQI model assigns a rating value to a series of 22 habitat characteristics (physical, chemical and biological; Table 4.1). These values are then used to calculate the standing crop of trout in kilograms per hectare.

$$\begin{aligned}\mathrm{Log}_{10}(Y+1)=[(-0.903) \quad &+(0.807)\log_{10}(X_1+1)\\ &+(0.877)\log_{10}(X_2+1)+(1.233)\log_{10}(X_3+1)\\ &+(0.631)\log_{10}(F+1)+(0.182)\log_{10}(S+1)][1.12085]\end{aligned}$$

Where:

Y = Predicted trout standing crop;
X_1 = Late summer stream flow;
X_2 = Annual stream flow variation;
X_3 = Maximum summer stream temperature;
F = Food Index = $X_3(X_4)(X_9)(X_{10})$;
S = Shelter = $X_7(X_8)(X_{11})$;
X_7 = Cover;
X_8 = Eroding stream banks;
X_9 = Substrate;
X_{10} = Water velocity; and
X_{11} = Stream width

Table 4.1 Habitat attributes used in the Habitat Quality Index Model.

Physical	Chemical	Biological
late summer stream flow	nitrate nitrogen	stream bank vegetation
annual stream flow variation	total alkalinity	fish food abundance
maximum summer stream temperature	total phosphorus	fish food diversity
water velocity	total dissolved solids	fish food type
turbidity	hydrogen ion	
cover		
stream width		
stream depth		
stream morphology		
eroding banks		
substrate		
bed material		
silt deposition		

Source: Binns and Eiserman 1979.

Binns and Eiserman (1979) successfully used this model to predict the standing stock of trout in 36 Wyoming streams. Binns has been able to apply the HQI model over many years on various streams in Wyoming. Most recently, Binns and Remmick (1994) used HQI scores to evaluate the changes in habitat quality following habitat enhancement projects. Binns and Eiserman (1979) indicated that the HQI can be modified to give results in terms of habitat units, thus providing resource managers with an alternative evaluation method. They also indicated that, when a control stream is available, results can be expressed as a percentage in order to compare an impacted habitat against an unimpacted stream. Due to the amount of information required for this method, it is questionable if this model could be applied on a large scale to the Foothills Model Forest area.

Austin et al. (1994) used relative weight (W_r), young-to-adult ratio (YAR) and stock density index (SDI) to evaluate brook trout stocks. While their results were inconclusive, they felt that these variables were useful for describing fish populations in similar water bodies over short periods of time.

4.3 INFORMATION DEFICIENCIES

4.3.1 Habitat Suitability Models

The most notable deficiencies are the lack of models for two of the target species, bull trout and mountain whitefish. There appears to be sufficient information to develop Habitat Suitability Indices for bull trout and mountain whitefish. The existing models for Arctic grayling (Hubert et al. 1984 or O'Neil et al. 1982b) and rainbow trout (Raleigh et al. 1984) require further assessment to confirm their applicability to the Foothills Model Forest region and to determine if modifications are required.

4.3.2 Basic Habitat Information

If the HSI models are to be utilized in the Foothills Model Forest region to identify critical habitats and be useful tools for resource managers in planning land use activities, there needs to be a great deal of baseline habitat information gathered in the foothills region. To reduce costs of data collection, select representative sites on each stream, river or lake in the foothills region and have a team of biologists assess the habitat at that site.

Decisions are required as to the best methods and time of the year for collecting habitat data. As habitat requirements vary over the year and the instream habitat itself can also vary (e.g., water levels drop throughout the summer, therefore pool depths will change), it is necessary to decide when habitat information should be collected. Ideally, it would be best to collect the information for each season. Given the personnel and monetary commitments that such a level of effort would require, collecting habitat information every season is not likely feasible. Some information, (e.g., water temperature, water depth and water velocity) can be collected year round at relatively little cost through the use of automated data recorders. Other habitat information, such as substrate characteristics, instream cover and pool depths could be recorded once a year. If this option were selected, it is advisable that such information be assessed during the late summer or early fall under baseflow conditions.

Standards must be established for both data collection and reporting. One of the greatest difficulties encountered during this study was the inconsistencies in data presentation and reporting, which made data comparisons very difficult.

If the Athabasca rainbow trout are to be managed as distinct from other strains (stocked populations) in the Foothills region, a great deal of information will be required to determine the distribution of these fish in the study area. Furthermore, detailed habitat assessments will also be required to determine if the habitat requirements of the Athabasca rainbow trout differ from other rainbow trout populations (i.e., can information from other rainbow trout populations be extrapolated to Athabasca rainbow trout?).

4.4 AN ALTERNATIVE TO HABITAT MODELLING

Modelling of potential land use impacts on the aquatic environment using habitat assessment models is an attempt to determine limiting critical habitats and gauge impacts on a micro scale. This requires a large long-term commitment of personnel and money in order to gather the necessary information. An alternative to critical habitat modelling is risk assessment modelling applied on a watershed scale. This method involves the use of GIS to develop thematic maps in order to assess the potential risk to a watershed from a given disturbance (e.g., timber removal, road construction).

D.A. Westworth and Associates Ltd. (1992) prepared a document for the Department of Fisheries and Oceans, which summarized the potential impacts of forest harvesting on fish and fish habitat. In this document, the authors prepared two case studies using watershed risk assessment models to explore the feasibility of employing GIS to evaluate the environmental risks of timber harvesting. One of the case studies prepared was for the ALPAC Forest Management Area. In this case study a watershed map was prepared from existing drainage maps and incorporated known fisheries information from sources such as Alberta Fish and Wildlife, commercial fishing records, Sportfish Capability Maps, and the Alberta Oil Sands Research Project reports. Terrain sensitivity maps were developed, particularly for soil erosion, using existing databases of soil and forest inventory maps.

Environmental Risk Factors (ERF), in this case due to timber harvesting, were classified as high if more than 50% of the watershed consisted of productive forest, moderate when 30-50% of the watershed contained productive forest, low when 10-30% of the watershed contained productive forest, and negligible if the productive forest was less than 10% of the watershed. These criteria were based upon the need for, and value of, maintaining the hydrological regime in the watershed.

The ERF could then be adjusted according to the terrain sensitivity of the productive forest land. In order to accomplish this objective, thematic maps were produced in a GIS format showing areas of productive forest land and the associated terrain sensitivity. Whenever high soil erosion hazards overlapped with productive forests, the ERF was increased using one of the following equations:

for high erosion hazard: ERF = %PF + (2x%HH)
for medium erosion hazard: ERF = %PF + (1x%MH)

Where:

ERF = Environmental Risk Factor
%PF = Percent watershed area of productive forested land
%HH = Percent productive forest land and high erosion hazard
%MH = Percent productive forested land and moderate erosion hazard

The ERF was then modified based on the fisheries habitat value (i.e., if the fisheries value was high, the ERF was increased; conversely if the fisheries value was moderate or low, the ERF was decreased).

The ERF was then divided by a constant value of three to produce a number between 0 and 1.0. According to this scheme, watersheds with the highest sensitivity would receive a value of 1.0. The following criteria was used for rating the risk:

- ERF 0.50-1.00 = High environmental risk;
- ERF 0.30-0.50 = Medium environmental risk;
- ERF 0.10-0.30 = Low environmental risk;
- ERF 0.00-0.10 = Negligible environmental risk.

To factor in the increased hazards often associated with roads, a separate ERF was calculated; this constituted the environmental risk due to roads (ERFR). The ERFR was based on the potential frequency for roads to contact fish habitat. The greater the contact, the more pronounced the potential for impact due to erosion, nutrient loading, and over-exploitation of fish resources.

The first step in calculating the ERFR is to determine drainage density and road intensity. The drainage density (DD) is a ratio of stream channel length to watershed area. Road intensity (RI) is the length of road per unit area. The ERFR can then be calculated from the following equation:

EFRI = DDxRI

The ERFR is calculated on a scale of 0.0 to 3.0, where:

- ERFR 3.0 = High environmental risk;
- ERFR 1.0-2.9 = Medium environmental risk;
- ERFR 0.0-1.0 = Low environmental risk.

The end result of each step in this process is a new thematic map which can then be overlayed with other layers to determine the level of watershed risk. This approach may be a cost effective alternative to habitat suitability modelling to identify habitats at highest risk.

D.A. Westworth and Associates (1992) acknowledged that the ALPAC case study was preliminary and could be refined with a more thorough fisheries assessment of the ALPAC area. In addition, they indicated that for operational fisheries management, the information database in the GIS can be refined to include site specific biological and habitat information for each reach or waterbody. They indicated that biological information for the purposes of impact assessment (including cumulative impact assessments) and biomonitoring may include fish abundance, composition and life history data. Habitat variables can include bench mark values for substrate, hydraulic characteristics, fish cover values, aquatic vegetation, streambed elevation, bank height, slope, stability, chemical and physical water quality, and benthic invertebrate data. However, as is the case with habitat suitability models, the amount of detailed information that can be collected is a function of the amount of funding available.

5.0 PRELIMINARY ANALYSIS OF HABITAT SUITABILITY PARAMETERS FOR ATHABASCA DRAINAGE FOOTHILLS STREAMS

The habitat requirements of four key sportfish species (rainbow trout, Arctic grayling, bull trout, and mountain whitefish) were presented in the previous sections. Because there was a scarcity of information specific to the Foothills Model Forest study area, we relied heavily on information obtained during studies in other areas. In order to gain a better understanding of the habitat parameters that contribute to the suitability of Alberta foothills streams for the target species, we compared sportfish relative abundance (CPUE data) with selected habitat attributes obtained from a representative sample of studied streams in the Athabasca River drainage. This information could provide some insight into the key variables in Alberta foothills streams and would be the first step in developing models that could provide resource managers with a predictive tool. Also, the information would provide guidance in terms of the parameters that should be measured during future inventory and monitoring programs. The analyses focussed on native populations of rainbow trout (Athabasca strain); however, the process (and to some extent the results) should be relevant to the other target species (Arctic grayling, bull trout, mountain whitefish).

5.1 STUDY STREAMS AND DATA SOURCES

Although there are a large number of inventory reports available within Alberta Environmental Protection (Fisheries Management Division) and the private sector, we restricted our analyses to a cross section of reports that offered some geographic diversity and were readily available to us. In total, 16 streams (most of which included multiple sample sites along their lengths) were included in the sample; 34 individual sites with relative abundance (CPUE) data and corresponding habitat data were represented (Table 5.1). The samples were distributed as follows: McLeod River watershed (19 sites), Berland River watershed (3 sites), Sakwatamau River watershed (9 sites), and Freeman River watershed (3 sites). The information was drawn from the Tri Creeks Watershed study report series, Alberta Environmental Protection Phase II Reports, and R.L. & L. Environmental Services Ltd. reports (Table 5.1).

5.2 HABITAT VARIABLES ANALYZED

A large number of potential habitat variables have been identified by previous investigators as having an influence on fish distribution and abundance; many of these have been described in the preceding sections. The first step in developing a more focussed group of variables (with greater relevance to salmonid populations inhabiting Alberta foothills streams) was to compile a master list. This list of 40 variables (Appendix B) described geomorphic and drainage basin characteristics (n=7), channel morphology and flow (n=15), and fish habitat

Table 5.1 Study streams included in analysis of habitat parameters and sportfish abundance, Upper Athabasca River drainage.

Stream	Drainage	No. of Sites	Reference Source
Wampus Creek	Upper McLeod River	2	Sterling 1978, 1986 & 1990
Deerlick Creek	Upper McLeod River	2	Sterling 1978, 1986 & 1990
Eunice Creek	Upper McLeod River	2	Sterling 1978, 1986 & 1990
Little Mackenzie Creek	Mackenzie Creek/Upper McLeod River	2	Seidel 1983a
Mackenzie Creek	Upper McLeod River	2	Seidel 1983b
Sundance Creek	McLeod River	6	R.L. & L. Environmental Services Ltd., 1993
Little Sundance Creek	Sundance Creek/McLeod River	2	Hills, 1983b
South Drinnan Creek	Drinnan Creek/Gregg River	1	Hills, 1983a
Moberly Creek	Wildhay River/Berland River	3	Hominiuk 1985
Carson Creek	Sakwatamau River	3	R.L. & L. Environmental Services Ltd., 1995c
Bear Creek	Sakwatamau River	1	R.L. & L. Environmental Services Ltd., 1995c
Hope Creek	Sakwatamau River	2	R.L. & L. Environmental Services Ltd., 1995c
Unnamed Creek #3	Sakwatamau River	1	R.L. & L. Environmental Services Ltd., 1993c
Unnamed Creek #5	Sakwatamau River	1	R.L. & L. Environmental Services Ltd., 1995c
Unnamed Creek #2	Sakwatamau River	1	R.L. & L. Environmental Services Ltd., 1995c
Louise Creek	Freeman River	3	R.L. & L. Environmental Services Ltd., 1995c
Total = 16 Streams		**Total = 34 Sites**	

(*n*=18). From this group of habitat variables, we selected 30 for systematic analyses (Table 5.2). The attributes depicted by these variables were selected because descriptive data were more readily available in the inventory reports or could be obtained relatively easily. We focussed on attributes that defined habitat suitability on a macrohabitat or reach level bases (stream gradient, elevation, conductivity, flow characteristics, etc.). Although it is apparent that microhabitat characteristics (e.g., LWD accumulations) are very important in determining abundance and use patterns on a site-specific basis, they operate within a given set of conditions determined by geomorphic, hydraulic and fish life history parameters. Furthermore, the level and type of data in most inventory level reports does not allow for a systematic, statistically valid treatment of microhabitat parameters.

Of the 30 variables, five were rejected after completing the data collection phase due to insufficient or poor data, or because the information was in a form not suitable for statistical treatment. Rejected variables included: average flow during winter, average summer dissolved oxygen levels, influence of beavers on habitat type/quality, extent of eroding bank, and presence of other fish species. Each of the remaining 25 variables were subjected to statistical analyses to determine if they were correlated with abundance (CPUE) of rainbow trout; product-moment correlation coefficients were developed and their respective levels of significance were determined (Sokal and Rohlf 1981).

Table 5.2 Habitat variables (*n*=30) that were compared to Athabasca rainbow trout CPUE values, Alberta Foothills Streams.

A)		**Drainage Basin**
	1	Stream order at point of sample location[a]
	2	Average stream gradient in reach encompassing site (m/km)[b].
	3	Elevation of site (masl)
	4	Area of drainage basin upstream of site (km^2)

	5	Distance of site from confluence with parent stream (km)
	6	Distance of site from headwaters (km)
B)		**Channel Morphology/Flow**
	7	Average channel width (wetted) (m)
	8	Average channel depth (m)
	9	Channel sinuosity
	10	Substrate composition (% fines)
	11	Mean annual flow
	12	Recorded discharge (spot sampling)
	13	Average flow during summer as % of mean annual flow
	*14	Average flow during winter as % of mean annual flow
	15	Variation in flows during incubation/intragravel period (% of spawning flows)
	16	Stream power (watts) @ max. daily discharge
	17	Stream power (watts) @ max. recorded daily discharge
C)		**Fish Habitat (Biological, Physical Chemical)**
	18	Average suspended sediment levels
	19	Maximum summer water temperature
	20	Average summer water temperature
	*21	Average summer dissolved oxygen levels
	22	Water conductivity
	*23	Influence of beavers on habitat type/quality
	24	Contribution of erosional habitats (% riffle/run/pool habitat)
	25	Pool-riffle ratio
	26	Contribution of deep (>1.5 m) high quality holding Pools/Runs
	27	Availability of instream cover (% of channel area)
	28	Availability of bank vegetative cover (% of bank area)
	*29	Extent of eroding bank (% of bank length)
	*30	Presence of other fish species

*Variables excluded from analyses due to insufficient or poor quality data; or the format of the data did not lend itself to statistical analyses.

[a] Determined from 1:50 000 NTS maps.

[b] Average stream gradients for discrete habitat reaches (ranging in length from approximately 1 to 10 km). Available from reports or calculated from 1:50 000 NTS maps.

5.3 HABITAT-CPUE RELATIONSHIPS FOR RAINBOW TROUT

Seven habitat variables were significantly correlated (significance level of $P<0.05$) with rainbow trout abundance (CPUE) in foothills streams in the Athabasca River drainage (Table 5.3). Three of these (i.e., stream gradient, elevation, and percentage of high quality runs/pools) exhibited a positive correlation with rainbow trout CPUE. This suggests that rainbow trout occupying foothills streams in the Athabasca Drainage tend to increase in abundance with increasing stream gradient, elevation, and percentage of high quality holding areas. The remaining four variables were inversely correlated with rainbow trout CPUE, indicating that abundance decreased as these habitat attributes increased in magnitude. The negative correlates were as follows: channel depth, variability of flows following spawning, drainage basin area, and conductivity. An eighth variable (i.e., percent fines in the substrate), was very close to achieving statistical significance ($P=0.063$); this was an inverse relationship (abundance of rainbow trout decreased with increase in contribution of fines).

Table 5.3 Stream habitat variables correlated with relative abundance (CPUE) of rainbow trout in second and third order foothills streams, Athabasca River Drainage.

Habitat Variable	**Sample Size (N)**	**Correlation[a] Coefficient (r)**	**Level of Significance**
Stream gradient (m/km)	33	0.507	$P<0.01$
Elevation (masl)	33	0.593	$P<0.01$
Drainage basin area (km^2)	28	-0.436	$P<0.05$
Channel depth (m)	32	-0.605	$P<0.01$
% Fines in substrate	25	-0.378	approaching $P<0.05$ ($P=0.063$)
Flow variation following spawning (m)	6	-0.906	$P<0.05$
Conductivity (µS/cm)	32	-0.644	$P<0.01$
% High quality holding areas	9	0.754	$P<0.05$

[a] Correlations based on log-transformed data.

In total, 14 variables were not significantly correlated (positively or negatively) with rainbow trout CPUE. These included stream order, distance of site from parent stream confluence and headwater source, wetted channel width, channel sinuosity, several flow parameters, several water quality parameters (suspended sediment, average and maximum water temperatures) and a range of physical habitat characteristics (% riffle/run/pool, pool-riffle ratio, instream cover, bank cover). The lack of correlation obtained for these parameters may have been due to small sample sizes and differences in measurement techniques amongst various investigators. As such, there would be value in repeating these analyses using larger standardized data sets.

5.4 TYPICAL FOOTHILLS RAINBOW TROUT STREAM (ATHABASCA DRAINAGE)

Based on our array of sample sites, rainbow trout in the Athabasca drainage appear to prefer small streams or upper reaches of larger systems. Most rainbow trout were located in Order-2 and Order-3 stream settings. For all rainbow trout capture sites (i.e., regardless of selective abundance), 68% were located on third order streams and 28% were on second order streams. For sites with higher relative abundance (CPUE ≥0.5 fish/min.), 60% were located in third order streams and 40% were in second order streams (Table 5.4). The results also indicated that rainbow trout were generally located at higher elevations. The mean elevation for all rainbow trout capture sites was 1172 m (95% confidence limits of 1067 m and 1277 m); sites with higher capture rates (CPUE ≥0.5 fish/min.) were located at somewhat higher elevations (_ of 1354 m, 95% confidence limits of 1229 m and 1479 m). The average stream gradient at sites where rainbow trout were recorded was 14 m/km (95% confidence limits of 10.7 m/km and 17.3 m/km). For sites with CPUE >0.5 fish/minute, the average gradient was 18 m/km (95% confidence limits of 14.5 m/km and 21.5 m/km).

Table 5.4 Stream characteristics related to rainbow trout presence and abundance in Athabasca drainage foothills streams.

Stream Characteristics	All Sample Sites				Sample Sites with RT				Sample sites with CPUE ≥0.5 fish/km)			
Stream Characteristic	**N**	**Mean[a]**	**95% CI**	**Range**	**N**	**Mean[a]**	**95% CI**	**Range**	**N**	**Mean[a]**	**95% CI**	**Range**
Stream Order	34	-	-	1 (6%) 2 (26%) 3 (65%) 4(3%)	25	-	-	1 (4%) 2 (28%) 3 (68%) 4 (0%)	10	-	-	1 (0%) 2 (40%) 3 (60%) 4 (0%)
Average Gradient (m/km)	34	12	± 2.8	1-33	25	14	± 3.3	3-33	10	18	±3.54	13-27
Elevation (masl)	34	1094	± 90	780-1550	25	1172	± 105	785-1550	10	1354	± 125	930-1550
Basin Area (km^2)	29	77	± 28	7-274	25	63	± 23	7-217	10	34	± 18	11-82
Distance from Mouth (km)	34	49	± 3.8	<1-42	25	9	± 4.6	1-42	10	9	± 9.2	1-42
Distance from Source (km)	34	17	± 4.3	2-47	25	14	± 4.0	2-46	10	12	± 4.7	6-25
Wetted Channel Width (m)	33	5.7	± 0.96	1-14	25	5.4	± 0.92	2-10	10	4.5	± 1.4	2-9
Channel Depth (m)	33	0.33	± 0.06	0.1-0.9	25	0.28	± 0.05	0.1-0.05	10	0.18	± 0.05	0.1-0.3
Conductivity (µS/cm)	23	367	± 60	158-550	14	288	± 68	158-530	4	190	± 63	158-275

[a] Reader is cautioned that means were derived from a restricted data set (i.e., relatively small number of sample locations, concentrating on smaller foothills streams).

Stream channel widths at sites inhabited by rainbow trout were generally quite narrow. For all rainbow trout capture sites, the average mean wetted width was 5.4 m; the 95% confidence limits were 4.5 m and 6.3 m. For higher CPUE sites, average mean wetted width was 4.5 m and 95% confidence limits were 3.1 m and 5.9 m. Not unexpectedly, since most rainbow trout occupied sites in small, relatively high gradient stream settings, average channel depth was very shallow. For the highest CPUE sites (i.e., CPUE ≥ 0.5 fish/min.) the average depth was 18 cm (95% confidence limits of 13 cm and 23 cm).

The recorded conductivity values tended to reflect the headwater location of most rainbow trout capture sites. The average conductivity value for all sites (including those where rainbow trout were not captured) was 367 µS/cm (95% confidence limits of 307 and 427 µS/cm). In contrast, the average value at the higher CPUE sites was noticeably lower (mean of 190 µS/cm, 95% confidence limits of 127 and 253 µS/cm).

In total, 12 species of fish were recorded at the 25 sites where rainbow trout were captured (Table 5.5). Only nine other species were recorded at the higher CPUE capture locations, the most common being bull trout (90% of sites), burbot (40% of sites), mountain whitefish (30% of sites), and brook trout (20% of sites). Several other species were present at lower occurrence levels (i.e., at 10% of the sites); these included Arctic grayling, longnose sucker, white sucker, spoonhead sculpin and longnose dace. Four species (brown trout, northern pike, lake chub, brook stickleback) were not captured in association with rainbow trout at the 10 higher CPUE sites. The higher CPUE rainbow trout capture sites had relatively simple species assemblages (i.e., 8 of 10 sites had only one or two other species).

Table 5.5 Occurrence of other fish species at sample sites in foothills streams with and without rainbow trout, Athabasca River Drainage.

Other Species	% Occurrence at all capture sites (*n*=9)		% Occurrence at all RT capture sites (*n*=25)		% Occurrence at RT capture sites with CPUE ≥0.5 fish/min. (*n*=10)	
Arctic grayling	56	-5	32	-8	10	-1
Bull trout	11	-1	52	-13	90	-9
Brook trout	11	-1	32	-8	20	-2
Brown trout	22	-2	4	-1	0	0
Mountain whitefish	56	-5	36	-9	30	-3
Burbot	67	-6	44	-11	40	-4
Northern Pike	33	-3	4	-1	0	0
Longnose sucker	89	-8	24	-6	10	-1
White sucker	67	-6	24	-6	10	-1
Spoonhead sculpin	44	-4	28	-7	10	-1
Longnose dace	11	-1	12	-3	10	-1
Lake chub	56	-5	16	-4	0	0
Brook stickleback	11	-1	0	0	0	0

6.0 REVIEW OF LAND USE EFFECTS ON AQUATIC ENVIRONMENTS

An overview of the known effects of major land use activities on fish and fish habitat is presented below. Table 6.1 provides a summary of the effects discussed.

6.1 LINEAR DEVELOPMENTS

Linear developments (e.g., roads, railways, pipelines, etc.) have the potential to cause significant impacts on the aquatic environment. The main concerns are sediment introduction into streams and rivers as a result of work in the channel during construction (e.g., bridge, piers, excavation of trenches for pipeline stream crossings) and bank erosion during both the construction and operational phases. The impacts of construction tend to be transitory in nature. Furthermore, these impacts can be mitigated with the application of current technology, appropriate construction methods and careful siting (e.g., avoidance of critical habitat areas). However, the long term impacts of these developments have not been as extensively studied. Therefore, the following sections will examine impacts associated with the post construction or operational phases of linear developments.

6.1.1 Roads

Roads can impose significant long term impacts on fish and fish habitat. Culvert crossings can block or hinder fish passage if excessive water velocities, which are too high for fish to negotiate, are created. If culverts are insufficiently sized and do not accommodate peak flows, the road crossing can be washed out and subsequent erosion can introduce significant quantities of sediment to the stream. Roads and vehicle traffic on the roads are also significant sources of sediment. These potential impacts will be discussed below in greater detail.

6.1.1.1 Barriers to Fish Passage

Improperly designed and poorly maintained culverts at stream crossings can delay or prevent upstream fish movements, thus restricting access to critical habitats (e.g. spawning areas, overwintering areas, etc.). Common mistakes with culverts include placing culverts too high above the existing stream grade, thus creating vertical outfall barriers and undersizing culverts, which results in water velocities that are too high for fish to swim against (Furniss et al. 1991). Mountain whitefish, a relatively weak swimming fish, are particularly susceptible to high water velocities at culvert installations; this should be taken into consideration when designing culvert crossings in systems containing mountain whitefish (Northcote and Ennis 1994).

Table 6.1 Potential impacts of land-use activities on fish and fish habitat.

Potential Effect	Cause	Impact on Fish or Fish Habitat	Reference
Increased sediment loading on streams	- road/pipeline construction and construction of stream crossings - inadequate erosion control measures - mass soil failures as a result of logging or improper placement of roads	Effects will vary depending on the timing and quantity of sediment in the stream and the species of fish involved. Effects can include: - impaired growth - changes in metabolism and blood chemistry - abrasion of gill tissue - decreased ability to fight infection - avoidance of sediment plumes - loss of territorial behaviour - burial of eggs and death of eggs/embryos/alevins - burial and compaction of spawning substrate	Anderson et al. 1995; Dietz 1971; McLeay et al. 1983; Platts and Megahan 1975; Stuart and Chislett 1979
Changes in stream habitat	- construction of stream crossings - alteration of riparian vegetation at pipeline, electric transmission line and road stream crossings	- increased water velocity in right-of-way leads to increased erosion and downcutting of stream bed - changes in stream bed and bank vegetation in the right-of-way can, in some circumstances, create more suitable habitat	McCart and de Graaf 1973; Peterson 1993
Interruption of migratory movements (for spawning, feeding or overwintering)	- poorly constructed or improperly placed culverts - avoidance of sediment plumes	- improperly placed culverts can form impassable barriers to fish movements. - fish will avoid entering areas of sediment plumes	Anderson et al. 1995; Furniss et al. 1991
Loss of riparian vegetation	- logging, mass soil failures, operation of heavy equipment in riparian areas	- loss of allochthonous carbon and energy sources - changes in stream water chemistry which cause changes in stream productivity (both primary and secondary) - loss of overhanging vegetation (i.e., cover habitat) - loss of inputs of Large Woody Debris - increased water temperature which, in some areas, can result in avoidance of warm water (i.e., loss of habitat), premature emergence of fry, or mortalities	Green and Kauffman 1989; Hartman and Scrivener 1990; Hetherington 1987; Holtby 1988; Murphy et al. 1986; Nip 1991; Platts 1991
Changes in stream morphology	- mass soil failures - sediment loading - stream channel clearing	- loss of habitat - increased susceptibility to predation - decreased growth rates	Alexander and Hansen 1988; Chamberlain et al. 1991; Murphy and Meehan 1991; Ryan and Grant 1991
Changes in stream flow regime	- logging - road construction in the watershed	increased discharge and increased run-off from roads can lead to: - increased scouring of spawning areas - increased water velocities can displace fry downstream and out of rearing habitat - increased erosion	Dietz 1971; Hetherington 1987; Sterling 1986
Increased exploitation of resources	- improved access to remote areas due to the construction of new roads for logging and petroleum exploration/extraction	- overharvesting of fish. Bull trout and mountain whitefish are particularly vulnerable	Beak 1978; Berry 1994; D.A. Westworth 1992; Ford et al. 1995; Halstead 1984

6.1.1.2 Sediment Loading

The largest producers of sediment in forestry operations are roads and skidder trails. Maximum sediment loading usually occurs during road construction, with levels of over 8000 mg/L sediment reported in some small tributary streams (Hetherington 1987). During the construction of a major haul road in the Deerlick Creek watershed in 1975, sediment loads in Deerlick Creek were 26% (1975) and 52% (1976) higher than pre-harvest levels, respectively (Nip 1991). The sediment contribution per unit area of roads is generally greater than that associated with other forestry activities such as log skidding and yarding (Furniss et al. 1991). In a Washington state study, fine sediments in spawning grounds increased above natural levels when more than 2.5 percent of the basin was covered by roads (Furniss et al. 1991). Runoff, poorly placed roads, and improper culvert size can contribute to mass-soil failures. Studies in Oregon indicated that mass-soil movements associated with roads were 30 to 300 times greater than in undisturbed forests (Furniss et al. 1991).

In the Swan Hills region of Alberta, Lengellé (1975) examined sources of erosion. Many of the sites were associated with roads which were built for forestry and/or oil exploration and extraction. The magnitude of erosion varied from small, shallow gullies (4 to 20 cm deep) to large areas (“huge inverted hollow pyramids” with “sheet erosion and regressing back walls”). At one site, a bulldozer had been used to rework a hill side; the resulting ruts readily formed gullies that followed the available slope. By the end of the first summer, the gullies were 0.45 m deep and by the next year the gullies were 0.76 to 0.91 m deep.

Rothwell (1979) studied stream crossings in the Hinton and Edson regions of Alberta; the sediment loading at stream crossings and the effectiveness of sediment control measures were examined. At one untreated site, Rothwell (1979) observed that following the first rainstorm, the study area was covered with an extensive system of rills and gullies. The rills were 25 to 50 mm deep and the gullies were 0.50 to 1.0 m in depth. The areas of maximum erosion and sediment transport occurred in the road side ditches and culvert fill sections. Sediment resulting from the storm event was observed in the stream immediately below the crossing where it formed small deltas. Rothwell indicated that these small deltas were subsequently removed by the stream and retransported to downstream areas.

Rothwell (1979) experimented with two treatment methods to minimize erosion. They included direct seeding and application of a brush mulch on exposed soil areas in ditches and at stream crossings. Rothwell (1979) reported 1.5 to 5 times less sediment produced in areas treated with brush mulch than in untreated areas. This was due to slowing of overland flow during rain events and the trapping of sediment on the up slope side of logging debris. However, Rothwell cautioned that the sediment trapping capabilities of the small brush are exceeded after one or two rain events and that it is a temporary measure which can be used until a good vegetative community has become established. For further information or erosion control measures, there are a number of technical information sources such as Rothwell (1978) and the B.C. Forest Practices Code Guidebooks for Riparian Management Area, Forest Road Engineering, Forest Road Construction and Forest Road Maintenance.

Sediment associated with roads is not confined to run off; vehicular traffic also can contribute significant amounts of sediment to streams. For example, vehicular traffic on logging roads in the summer caused increased levels of Total Suspended Solids (TSS) and Total Phosphorus (TP) at downstream monitoring stations near Temagami, Ontario (Hall 1996).

Anderson et al. (1995) recently published a summary report which examined the effects of sediment release on fish and fish habitat. They concluded that increased sediment loading can have the following effects on fish:

- Increased physiological stress which manifests itself in many forms ranging from:
    - Impaired growth in steelhead trout and coho salmon exposed to 84 to 120 mg/L of clay; impaired growth of Arctic grayling exposed to 100 mg/L of clay. It is thought that impaired growth results from increased metabolic demands as a result of stress rather than from reduced feeding.
    - Changes in blood chemistry such as increased haematocrite and erythrocyte count, haemoglobin concentration, and elevated blood sugar, which can be observed within five days of exposure. Also observed are decreases in blood chloride content and depletion of liver glycogen. These changes often result in decreased swimming endurance in some species. Servizi and Martens (1992) reported that blood sugar levels in coho and sockeye salmon increased <6 mmol/L at 0 g/L suspended sediment to approximately 12 mmol/L at 1.6 g/L suspended sediment.
    - Abrasion of gill tissue. The severity of damage appears to be related to both the exposure dosage as well as size and angularity of particles. The amount of abrasion depends upon the dose and time of exposure. For example, gill damage in rainbow trout can be caused at exposures to suspended solids in concentrations as low as 270 mg/L for 13 days. Yet short duration (4 days) exposure to 1300 mg/L suspended sediment caused no gill damage in young-of-the-year Arctic grayling. A secondary effect of gill damage is increased parasitic infections.
    - Decreased ability to fend off infections.
- Behavioural changes such as:
    - Avoidance of sediment plumes and downstream movements of fish. For example coho salmon juveniles (*Onchorynchus kisutch*) avoided total suspended solid concentrations of 88 mg/L and Arctic grayling avoided concentrations in excess of 100 mg/L.
    - Reduced feeding ability, particularly among species which are "sight feeders."
    - Loss of territorial behaviour and the interruption of salmonid migratory movements.
- Direct effect on fish populations:
    - Sediment accumulation which results in the burial of eggs.
    - Short term exposure to very high concentrations of total suspended solids (11 000 to 55 000 mg/L) significantly increases salmonid mortalities.

Anderson et al. (1995) rated the tolerance of 106 fish species to Total Suspended Solids (TSS). They reported that the two salmonid species in their study, brook trout and rainbow trout, had a very low tolerance to TSS, (i.e., a high sensitivity).

Excessive quantities of fines in the substrate (i.e., sand and silt) kill embryos, alevins, and fry that occupy stream substrates. Deposition of fine sediment results in decreased water movements through the substrate. This, in turn, decreases permeability to dissolved oxygen which is required by embryos, causing build-up of metabolic waste products excreted from the developing embryo sometimes to lethal levels; also emergence can be delayed or, in the worst case, prevented altogether (Platts and Megahan 1975; Anderson et al. 1995). Deposition of fine sediment can occlude the interstitial spaces in the substrate, which are essential habitat for developing fry and juvenile overwintering. Loss of this habitat exposes the fry to increased predation (Anderson et al. 1995).

Fraley and Shepard (1989) identified concerns that increased sedimentation, in the North Fork of the Flathead River, Montana (arising from timber harvesting, road construction and coal mining) could negatively impact bull trout populations by degrading spawning and rearing habitat. Ford et al. (1995) also have identified increased sedimentation, arising from oil and gas extraction and coal mining, as having significant negative impacts on bull trout, mainly through the alteration of habitat.

Halstead (1984) observed that changes in habitat due to sedimentation continue to threaten bull trout stocks in Alberta. Halstead (1984) also reported that mountain whitefish can be indirectly affected by sedimentation through suppression of the benthic invertebrate community (i.e., reduced food supply).

Dietz (1971), who measured the abundance of rainbow trout in three similar streams in the Tri-Creeks Experimental Watershed, Alberta, suggested that the relative absence of trout in Eunice Creek, downstream from the access road to a drilling site, was at least in part due to siltation in that stream as compared with Wampus and Deerlick creeks. Old roads and seismic lines cut at right angles to the direction of flow of the creeks were the major sources of sediments. Besides reducing intragravel flows by filling the interstitial spaces, sediment loading also appears to compact the gravel and make it unsuitable to the small, headwater trout for spawning. Dietz (1971) and Sterling (1978) speculated that the bull trout, because of their larger size (compared to the rainbow trout) could still build redds in the lower portion of Eunice Creek and that this, coupled with the reduced competition with rainbow trout, has allowed the bull trout to become the predominate species in the lower portions of Eunice Creek.

Stuart and Chislett (1979) reported a year-class failure of Arctic grayling in the Sukunka River in 1976; it was thought to be a result of increased sediment load associated with extremely high flows. They reported that suspended sediments ranged from a maximum of 333.8 mg/L in the Sukunka River to a minimum of 66.5 mg/L in Martin Creek. They speculated that the elevated suspended sediment levels may have made it difficult for fry to locate food.

McLeay et al. (1983) examined the acute effects of suspensions of Yukon placer mining sediments on young-of-the-year Arctic grayling. They concluded that short-term exposure to sub-lethal concentrations of suspended sediment can cause a number of acute stress responses. However, the environmental relevance of these responses cannot be ascertained without further studies including the influence of sediment concentration and type on stress response, the impact of prolonged exposure, and the effects of sediment type and strength on other life stages.

Reynolds et al. (1989) compared two streams, an undisturbed system and one with mining activities upstream. The turbidity varied from 950 to 8200 NTU in the stream receiving mine discharge and was much lower (0.20 to 0.87 NTU) in the undisturbed stream. Reynolds et al. (1989) demonstrated that if Arctic grayling sac fry could not escape from the stream carrying mining sediments, they would not survive. Age-0 fingerlings and Age-2 juveniles suffered gill damage, starvation and slow maturation. They suggested that the indirect effects of sedimentation (loss of summer feeding opportunities and reduced reproductive success) may be more severe to the Arctic grayling populations than the direct effect of sedimentation on the health and survival of individual fish.

6.1.2 Pipelines and Seismic Lines

Perhaps the most visible impact of pipelines on the environment is direct mortality of aquatic organisms following the uncontrolled release of petroleum products into the aquatic environment as a result of a pipeline break. Pipeline breaks may occur for many reasons ranging from corrosion to bullet holes (Beak 1978). The impacts can range from long-term tainting of fish tissue (Beak 1978) to fish mortalities occurring immediately following a spill (Environment Canada and Department of Fisheries and Oceans 1988).

Short-term increases in sediment loads during the construction of a pipeline or seismic line can negatively impact fish populations. Ford et al. (1995) indicated that sediment arising from pipeline construction can cover Arctic grayling eggs (or other species depending on the timing of construction), often resulting in death due to suffocation. Northcote and Ennis (1994) cautioned that when instream work occurs during winter, sediment loading must be controlled due to possible impacts on mountain whitefish eggs and population recruitment.

The most significant long-term impact of pipeline and seismic lines on the aquatic environment is the introduction of sediment from approaches and banks at stream crossings. In a review of post-operational assessments of linear developments in northern Canada, Beak (1978) documented that erosion, gully formation, and mass wasting can continue for several years after construction. The effects of sediment loading from these types of land-use activities would be similar to those discussed above in Section 6.1.1.2. The magnitude of impacts would be dependent on the volume of material entering the stream and the length of time that such inputs are allowed to continue.

In some instances, increased erosion at the right-of-way (RoW) can alter the aquatic habitat to a more suitable form of habitat. For example, McCart and de Graaf (1973) observed that increased stream bed erosion at the Canol Pipeline crossing of Ray Creek, N.W.T. resulted in the development of a series of gravel bottomed riffles and pools in the RoW. McCart and de Graaf assessed the benthic community upstream and downstream of the RoW and in the RoW. Sites in the RoW had consistently higher numbers of benthic invertebrates than sites either upstream or downstream of the RoW.

When pipelines are constructed, the stream channel within the RoW is realigned to remove natural meanders; this usually results in higher velocities in the RoW. The stream banks are then stabilized with coarse gravel, cobble and boulders to prevent erosion. The combination of high velocity and coarse substrate can provide ideal fish habitat, particularly in low gradient streams. In an assessment of pipeline stream crossings near Rocky Mountain House, Alberta, some of the best quality fish habitat and highest numbers of fish in the surveyed streams, were found within the RoW (D. Hamilton, R.L. & L. Environmental Services Ltd., personal observation, October 1995).

6.1.3 Power Transmission Lines

As with other linear developments, erosion at transmission line stream crossings has the potential to cause significant negative impacts on the aquatic environment. The erosion potential at transmission line RoW's would be similar to that described for pipeline and seismic lines.

Peterson (1993) observed that removal of trees along a transmission line RoW increased incident solar radiation, which in turn promoted the growth of streambank vegetation. Peterson (1993) observed that in the RoW, the stream bank was covered with a forb and shrub layer, which often extended over the stream channel forming overhead cover. In contrast, the streambanks in adjacent areas, with intact forest canopy, had only a scattering of herbs and the occasional sampling. Peterson observed that, once established, the root mass of the forb and shrub community stabilized the stream banks and resisted erosion. Peterson also noted that the root mass prevented undercut soil from sloughing off. As a result, undercut trees which are normally found where the forest canopy is intact have been replaced with undercut banks in the RoW. Peterson (1993) hypothesized that the increased resistance to bank erosion resulted in increased bed erosion, which deepened the channel in the RoW and increased both the depth and area of pools. Peterson (1993) felt that the increased numbers of trout (rainbow and brook trout) observed in the RoW compared to naturally forested sections was the result of the change in habitat in the RoW. Peterson recorded an average of 30.8 trout per RoW reach compared to 18.9 trout per forested reach.

There is the potential, therefore, that once crossing sites have become revegetated, fish habitat may be improved relative to its previous state. When permitting stream crossing in the future, environmental regulators should require that as much natural stream side vegetation be left intact, as is practical, in order to facilitate regrowth and minimize erosion.

6.1.4 Resource Exploitation

As noted above, a by-product of resource extraction, be it for forestry, mining or oil and gas, is the construction of access roads. The network of roads required, particularly for forestry, can provide easier access into formerly inaccessible areas. Due to the ever increasing popularity of all-terrain vehicles, pipeline and transmission line RoW's can also facilitate access into formerly inaccessible area. D.A. Westworth and Associates (1992) observed that for every kilometre of road development there can be tens or hundreds of kilometres of new access provided by intersecting seismic lines, pipelines, transmission lines, and railways.

Easier access generally results in increased use of such areas for recreational activities. This in turn, results in increased angling pressure on streams and waterbodies in areas which formerly experienced little or no fishing pressure. Increased angling pressure after new access into an area has been developed is well-documented and a common concern among both the scientific community and government regulatory agencies (e.g., Beak 1978; Halstead 1984; D.A. Westworth and Associates 1992; Ford et al. 1995).

Increased angling is of particular concern for management of bull trout stocks, although there is concern for all native fish stocks (including Athabasca strain rainbow trout, Arctic grayling, and mountain whitefish). Due to the bull trout's diet and aggressive feeding habits, it is readily angled and susceptible to overfishing (Fraley and Shepard 1994; Berry 1994; Ford et al. 1995). As bull trout generally do not mature until Age 5+, overfishing can have long-term detrimental impacts on populations (Berry 1994; Ford et al. 1995). Northcote and Ennis (1994) attributed the decline of mountain whitefish in many regions of British Columbia to overharvesting. Therefore, prior to increasing access, strict fishing regulations must be established to prevent depletion of fish stocks.

6.2 TIMBER REMOVAL ACTIVITIES

Timber removal (for oil and gas exploration, agriculture, wildfires, and forestry) has been ongoing in Alberta for over 100 years and has taken place in over 50% of the province (Hebert 1994 in Alke 1995). Since the early 1900's, oil and gas exploration activities have accounted for 3-5% of the land clearing in northern Alberta. At the current rates of extraction, timber harvesting results in the clearing of approximately 0.25% of the land in northern Alberta each year (Hebert 1994 in Alke 1995). In terms of research and public awareness, the impacts of forestry activities have received more attention than the impacts of land clearing for agriculture or oil and gas extraction or mining (Alke 1995). Impacts of continued land clearing activities in northern Alberta (including the Foothill Model Forest region) are predicted to include (Alke 1995):

- altered water chemistry;
- soil erosion and flooding;
- increased water yield from cleared areas;
- decreased groundwater loading;
- changes in the water table (lower or higher depending on parent soil material);
- a dryer region due to increased snow sublimation and evaporation; and
- inhospitable habitats for indigenous wildlife and aquatic biota.

The following sections will examine the impacts of land clearing activity on the aquatic environment, in particular the potential for these activities to impact upon fish and fish habitat.

6.2.1 Riparian Zone

The most severe impacts of forestry activity are the removal of timber and disturbance of vegetation in the riparian zone. The term riparian is generally used to refer to the vegetative habitat adjacent to the bank of a stream or river. Many descriptions of riparian zones exist; they are based mainly on descriptions of physical characteristics and plant community associations (Gehardt et al. 1989). For example, riparian areas in British Columbia have been defined as being “areas of land adjacent to streams, rivers or lakes and containing vegetation that, due to the presence of water, is distinctly different from adjacent upland areas” (B.C. Forest Practices Code Regulation 174/95). For some stream rehabilitation projects in Manitoba, the riparian areas are defined simply as areas at least 10 m wide from the top of the stream bank (DLBAB 1994). Gehardt et al. (1989) have proposed a new classification based on stream hydrological and geomorphic parameters. The main defining feature of the riparian zone that they proposed is the area adjacent to rivers and streams that is affected by one-to-three-year floods.

The riparian areas along streams and rivers are important factors in determining water quality and stream morphology. Riparian zones provide significant sources of allochthonous carbon (fixed carbon) and energy to streams, particularly to small headwater streams through the input of leaf litter, groundwater seepage, and soil erosion (Murphy and Meehan 1991). It has been estimated that 90% of organic matter supporting headwater stream communities is of terrestrial origin and that 99% of energy input in small forested streams comes from allochthonous sources, while only 1% is produced by instream autochthonous sources (i.e., algal photosynthesis) (Green and Kauffman 1989).

The contribution of riparian vegetation to streambank stability is widely recognized (Heede 1986, Maddock 1972). Beeson and Doyle (1995) investigated the role of riparian vegetation in the protection of banks during major flood events in four streams in southern British Columbia. “Bends without riparian vegetation were found to be nearly five times as likely as vegetated bends to have undergone detectable erosion during flood events. Major bank erosion was 30 times more prevalent on non-vegetated bends as on vegetated bends.”

Riparian vegetation improves water quality by removing nutrients from subsurface water. The biogeochemical processes that occur in the riparian zone remove nutrients such as nitrate-nitrogen and phosphorus (Green and Kauffman 1989). Removal of nutrients in subsurface water minimizes nutrient inputs to streams, thus ensuring good water quality in receiving streams.

Vegetation in the riparian zone provides thermal cooling of water temperature. It also reduces water velocity and erosive energy during periods of peak flows and flooding (Green and Kauffman 1989). The trees growing in the riparian zone provide essential inputs of Large Woody Debris (LWD) to streams; instream LWD provides important cover habitat for fish. Fish habitat can be enhanced by instream or overhanging cover, forming quiet areas or pools for fish to rest in (Platts 1991). The presence of LWD in low order streams may help reduce the size of peak flows as well as provide stability to banks to prevent degradation (Gerhardt et al. 1989).

6.2.1.1 Effects on Water Temperature

Water temperature is one of the most critical variables used to determine suitability of a reach or stream for a particular fish species (Ford et al. 1995; Hubert et al. 1985; Raleigh et al. 1984).

Effects of vegetation removal in the riparian zone are manifested in altered temperature regimes during both summer and winter. Summer water temperatures increase, particularly in small, slow-moving streams, and freezing rates increase in winter (Hetherington 1987). Post-harvesting increases in maximum water temperature have been found to be generally less than 10°C, although increases of 15°C have been reported in extreme cases (Hetherington 1987, Beschta and Taylor 1988). Plamondon (1995) reported that in studies carried out in Quebec, there was a two degree increase in water temperature in 50% of the streams that had no buffer strip following harvesting activities. Nip (1991) reported that in the Tri-Creeks study, water temperature increases following logging were most evident during the summer. For example, the mean monthly August water temperature in Wampus and Deerlick creeks increased by 3.4 and 5.7°C, respectively. The highest daily maximum water temperature (21°C) was observed in Deerlick Creek. The length of time that post-logging increases in water temperatures can be observed is dependent on the post-logging riparian zone management. Beschta and Taylor (1988) observed increased water temperatures for 10 years after logging.

In addition to removal by timber harvesting, loss of cover along streams can also occur indirectly as a result of mass-soil failures or blow down. For example, in a 23-year post-harvest study, Ryan and Grant (1991) found that erosion of stream banks by debris torrents and severe floods, which were a result of clear cutting higher up in the watershed, caused the loss of riparian vegetation.

The effect of increasing water temperature on fish can result in avoidance of the area (i.e., loss of habitat), particularly among cold-water species such as salmonids, and in decreased success of fry emergence. In some cases, increased water temperatures have resulted in premature emergence of fry (Hartman and Scrivener 1990). This can adversely affect their long-term survival, particularly, if the food supply that the fry rely upon has not yet emerged (Holtby 1988).

The fact that Deerlick Creek reached 21°C following harvesting (Nip 1991) may be relevant. This temperature exceeds the upper maximum temperature tolerated by bull trout (18°C) and mountain whitefish (20.6°C; ref. Section 3). While 21°C is tolerated by rainbow trout and Arctic grayling (ref. Section 3), it is above the temperature considered optimal for growth for all four target species. Therefore, such temperatures could result in the reduced use or avoidance of areas of elevated water temperature by salmonids.

6.2.1.2 Effects on Stream Nutrients

The removal of trees along the banks of streams increases the amount of light reaching the stream, which may result in higher algal productivity. In a study of clearcut streams in Alaska, Murphy et al. (1986) found that primary and secondary productivity increased in reaches adjacent to clearcut areas. This resulted in an increase of salmon young in the summer as a result of an increase in food supply. Increased summer productivity however, was offset by reductions in winter carrying capacity, caused by removal of debris, collapse of undercut banks, and the de-stabilization or embedding of the channel substrate (Murphy et al. 1986).

Changes in water chemistry as a result of runoff from clearcuts is highly variable. There may be a decrease or an increase in nutrient and ion inputs from the uplands to the stream (Hetherington 1987). A study of clearcut watersheds by Hartman and Scrivener (1990) indicated that the total ion export to streams increased by 30 times following logging. The study also reported that nitrate levels were elevated for at least two years following logging, but that phosphate levels did not increase. In the post-logging years, the instream nitrate and phosphate concentrations decreased as nutrients were taken up by both algae and the regenerating forest.

In the Tri-Creeks Experimental Watershed, nitrate, nitrite and ammonia concentrations decreased significantly in the three study streams following logging. There were also significant increases in total phosphate loading in both Deerlick and Wampus creeks, but not in Eunice Creek (the control) following logging activity (Nip 1991).

Dissolved oxygen is a critical habitat component. A significant increase in organic material entering streams as a result of logging activities has the potential to increase the consumption of dissolved oxygen through the decomposition process, which could, in turn, reduce the amount of dissolved oxygen available for fish and other biota. Decreases in available oxygen could also reduce habitat availability by restricting use of portions of streams where the dissolved oxygen levels are sub-optimal or limiting to fish. Plamondon (1995) reported that reductions in dissolved oxygen of up to 6 mg/L have been observed following logging. These reductions were related to the amount of organic material left in the stream following harvesting.

6.2.2 Cutting Areas

The high water quality of streams in forested watersheds is usually a result of limited erosion, high soil infiltration rates, soil stability (provided by tree roots), and the ability of forest soils to retain nutrients. In addition, tree roots and fallen trees in streams provide suitable habitat for aquatic organisms by providing a network of pools, riffle zones, and rearing habitats (Hetherington 1987). There are several ways in which forestry operations can affect streams and lakes:

- direct inputs of sediment and debris;
- changing stream morphology;
- altering water temperature; and

- changing stream productivity.

6.2.2.1 Erosion and Sediment Loading

Extensive clear-cutting activities can result in increased sediment loading into streams via surface soil erosion, mass-soil failures (land slides), and stream bank erosion (Hetherington 1987; Hartman and Scrivener 1990). Erosion of mineral soils is accelerated if there is significant removal of the forest floor covering during road construction, by log skidding, and softwood silviculture practices of scarification and burning (Hetherington 1987). Plamondon (1995) indicated that logged ephemeral streams are significant sources of sediment. Plamondon further noted that in order to eliminate this source of sediment, the Quebec government has implemented a mandatory buffer strip of 5 m in width to protect ephemeral streams. A more detailed discussion of the effects of sediment loading on fish and fish habitat was provided in Section 6.1.1.2.

6.2.2.2 Stream Morphology

Stream channel morphology is an important factor that influences the type and abundance of fish habitat. Features important to fish are pools, riffles (shallow, fast-flowing water), spawning substrates, obstructions, side channels, etc. Alteration of these features will affect fish utilization of a stream. For example, if all the pieces of LWD are removed from a stream, this would likely result in the loss of quiet resting pools and cover, which provide important habitat for many species of fish. Pools can also fill up with sediment and spawning gravel can be covered with sediment as a result of increased sediment loading (Chamberlain et al. 1991). Changes in channel morphology, bed composition, and instream cover (e.g., filling in pools, smothering of gravel riffles) as a result of addition of fine material (i.e., sand) have been shown to increase the vulnerability of trout to predation and may decrease food availability (Alexander and Hansen 1988).

Mass-soil failures or debris flows often result from poorly designed clear-cuts and improper road placements. In addition to contributing to instream sediment loads, mass soil failures can alter stream channel morphology. A study of the Elk River Basin in Oregon showed that this debris caused an increase in the development of gravel bars (Ryan and Grant 1991). Debris flows in tributary channels can widen the stream channels with resulting deposition of material in the main channels of a river, causing constrictions of the river, along with reduced flows (Ryan and Grant 1991). Other mechanisms that can alter stream morphology include the removal of boulders and/or trees by the debris torrents (Chamberlain et al. 1991) or by the removal of fallen merchantable wood from the stream (Murphy and Meehan 1991).

6.2.2.3 Primary and Secondary Productivity

Continuous high sediment loads can decrease primary productivity by reducing light penetration and by smothering periphyton. Increased sediment loading as a result of forestry activities can also decrease the secondary productivity of a stream by smothering invertebrates (Burns 1970), and sealing interstitial spaces, thus preventing invertebrate access to preferred habitat (Brusven and Prather 1974). This, in turn, can result in decreased invertebrate abundance, hence less food for fish to eat. As a consequence, fish growth rates may decrease (Alexander and Hansen 1988). In addition, reduced light availability can also reduce fish feeding efficiency because many fish are sight-feeders (Langer 1974).

6.2.2.4 Flow Regime

Depending on the slope of the land, soil type, and infiltration rates, cutting activities can temporarily increase the annual water yield in streams. Additionally, runoff from clear-cuts can accentuate flooding caused by peak rain events. Anderson et al. (1976) observed that peak flows and storm flows increased following logging activities.

At Carnation Creek, Vancouver Island, B.C. road construction was responsible for many of the hydrological changes observed during active logging operations. Road construction interrupted sub-surface water flow, increased surface erosion and re-routed water in the drainage basin through culverts and ditches. This inturn resulted in a decrease in time between peak rainfall and peak discharge and an increase in peak seasonal flows and erosive energy (Tschaplinsky 1992).

Logging may cause flooding because of greater snowmelt run-off from clear-cuts. Chapman (1995) indicated that a watershed that had 30% of its timber removed accumulated 20% more snow, which resulted in a 7% increase in the amount of water reaching the ground. In some cases, spring peak floods in streams can be ameliorated by desynchronization of snowmelt runoff (i.e., snowmelt runoff from clear-cuts occurs earlier than in other areas) (Hetherington 1987).

Swanson and Hillman (1977) reported that the average increase in storm flow in cut watersheds near Hinton, Alberta, was 1.5 to 2.0 times (range was 0 to 3.3) greater than in similar basins which had no forestry activity. The annual water yield increased by 27% in the logged watersheds. Swanson and Hillman (1977) concluded that the effects of logging on run off and water yield can be expected to last for approximately 30 years.

In addition to increased flows from direct snowmelt, harvesting activities can affect the forests' hydrological cycle. Normally 65 to 80% of the annual precipitation is lost to the atmosphere via evapotranspiration. The remaining 20% will infiltrate the soil, and eventually move through subsurface pathways (i.e., groundwater) into streams and wetlands. When the forest is cut, the transpiration pathway is lost, resulting in wetter soils and higher water tables. Higher water tables, in turn, result in more overland flow during rain events. Roads constructed in a watershed can dramatically increase the rate of surface runoff, sometimes by as much as 100 to 200% (Chapman 1995).

Increases in peak flows as a result of logging could impact fish populations by increasing scouring of the spawning riffles. For example, rainbow trout eggs, because of the shallow deposition depth, may be more susceptible to scouring than bull trout eggs which are deposited deeper and hatch earlier (Dietz 1971). Sterling (1986) indicated that in the Tri-Creeks study the discharge needed to scour gravel from spawning areas was 0.566 m^3/s. Sterling (1990) indicated that the rainbow trout year class failures observed in 1969 and 1980 were a result of flood events which had discharges above the critical limit. The occurrences of these flooding events during the Tri-Creeks study was unfortunate, because the impacts masked any changes to population abundance as a result of habitat alterations resulting from logging (Sterling 1990).

7.0 SUMMARY AND DISCUSSION

The goal of the present study was to assemble existing information on habitat requirements for the four key sportfish species (by lifestage). The intent was to integrate this information into a comprehensive, computerized database which would assist resource managers to 1) evaluate land management alternatives 2) assess cumulative effects and environmental impacts, and 3) identify management and inventory priorities. A database, and a descriptive report, were completed to fulfill this objective. The report provides additional detail on the information entered into the database, including sources of habitat data, HSI models, non-HSI predictive tools and impacts of land-use activities. Each of these report sections is briefly summarized and discussed below.

7.1 REVIEW OF SPECIES HABITAT REQUIREMENTS

The information collection phase included a computer literature search which allowed a review of university and government libraries and key scientific journals. In total, the literature search yielded 398 citations either related to the four target species or the impacts of forest harvesting on aquatic environments. Selected references from this group were reviewed in detail and the relevant information was entered into the computer database. The detailed habitat information was stored in a relational database that was developed in Microsoft Access (Ver. 2.0). Other components of the study included original analyses of rainbow trout habitat relationships (using a representative sample of inventory reports for the Foothills Model Forest region) and a literature review of impacts associated with forest harvesting activities.

In the report, the habitat requirements of the four species are described according to several life requisite functions (spawning and egg development, rearing, adult feeding/holding and overwintering). Within the various sections, habitat requirement data is presented from both the study area (if available) and from areas within the geographical range of that particular species. The information is also summarized in a tabular format in order to facilitate reader access of key habitat information.

The literature review phase specific to the Foothills Model Forest Region revealed limited information on the habitat requirements of the four target species. Most studies conducted to date have been inventory-type investigations (i.e., small number of sampling sites, visited on only one occasion, etc.) designed more to determine fish distribution and relative abundance in a watershed and record basic habitat conditions rather than identify and describe preferred or critical habitat factors. An exception in this regard is the Tri-Creeks Experimental Watershed research study which provided long-term data primarily on rainbow trout in three small foothills streams.

To develop the type of information base required, there is a need to collect more detailed habitat information data and to ensure that it is linked to the abundance of a given species and species life-stage (e.g., abundance of rainbow trout juveniles in pools with and without large woody debris cover). It also may be advantageous to carry out a repetitive sampling program at representative sites (or reaches) in the various watersheds to determine seasonal and annual variations in habitat selection and population densities. Of particular importance is the need to investigate the winter ecology of foothills streams since most are under ice cover for approximately six months of the year.

While more detailed information is being generated, resource managers and planners will have to rely on fish-habitat relationship data from regions outside the study area. Much of this information can be adopted since species have relatively similar habits and preferences throughout their geographic range. To take advantage of the extensive inventory data which has been collected by resource agencies, consultants and industry over the past (approximately) 40 years, this information should be organized into a computerized database. This could provide useful information on habitat preferences on a macrohabitat scale (e.g., reach basis) which would be useful for management and planning purposes until more detailed (i.e., microhabitat scale) information is developed and tested.

7.2 REVIEW OF EXISTING HABITAT SUITABILITY MODELS

As part of the present study, an assessment of the availability and applicability of existing habitat suitability models for the four target species was carried out. Published models were located for rainbow trout (Raleigh et al. 1984) and Arctic grayling (Hubert et al., 1985). These Habitat Suitability Index (HSI) models were developed by the U.S. Fish and Wildlife Service; they were intended for application throughout the geographic range of the target species. The models provide an index of habitat suitability on a numerical scale (0.0 to 1.0; 1.0 indicating optimum conditions) and can be simple or sophisticated depending upon the amount and specificity of the habitat and fish use data available. The development of the HSI models was to have included a structured follow-up phase whereby key habitat parameters were identified leading to a simplified, yet responsive model. Unfortunately, the second phase was not carried out and a directive was issued to the study group to begin field testing the existing HSI models. As of this writing, field tested models for rainbow trout or Arctic grayling have not been published (although there could be individuals or groups pursuing this on a regional basis). The applicability of these two models to the Foothills Model Forest area is unknown at this time. To determine this would require the application of systematic field verification and testing procedures (e.g., comparison of test model outputs to actual fish population densities).

A modified version of the Arctic grayling HSI model was developed and applied for Hartley Creek, a tributary to the Muskeg River (Athabasca River drainage) near Fort McMurray, Alberta. Although the model used a small number of easily measured parameters, it allowed the investigators to characterize the various stream reaches, according to their suitability for Arctic grayling. While this model has not been tested on other watersheds in Alberta, and may or may not be applicable, it appears to indicate that HSI models may have considerable potential.

We were not able to locate habitat suitability models specifically developed for bull trout or mountain whitefish; although, a model for bull trout is currently being developed at Montana State University. This model which utilizes basic habitat information to predict the presence or absence of bull trout may be transferrable to the Foothills Model Forest region.

The applicability and transferability of the Habitat Quality Index Model (HQI) developed for Wyoming streams by Binns and Eiserman (1979) was also assessed. The model uses 22 habitat parameters to predict standing crops of trout (brown, rainbow, brook, cutthroat); standing crop was then used to rate the habitat quality of the stream in question. While this technique was successfully used to predict the standing crop of trout in 36 Wyoming steams, it is unlikely that this model could be applied on a large scale in the Foothills Model Forest because of the comprehensive field data requirements.

The study also discussed risk assessment modelling applied on a watershed scale, as an alternative to critical habitat modelling. This method, described by D.A. Westworth and Associates Ltd. (1992) involves the use of GIS to develop thematic maps in order to assess the potential risk to a watershed from a given disturbance (e.g., timber removal, road construction). A range of criteria were included in the assessment including percentage of watershed containing productive forest, soil erosion hazard, and fisheries habitat values. The objective was to produce a rating (0.0-1.0; 1.0 being high environmental risk) which defines the level of watershed risk. This approach may be a cost-effective alternative to habitat suitability modelling in the Foothills Model Forest region; however, the data requirements remain considerable.

7.3 PRELIMINARY ANALYSIS OF HABITAT SUITABILITY PARAMETERS FOR ATHABASCA DRAINAGE FOOTHILLS STREAMS

Because of the scarcity of information specific to the Foothills Model Forest study area, we relied heavily on information obtained from other areas. To gain a better understanding of the habitat parameters that contribute to the suitability of Alberta foothills streams for one of the four target species, rainbow trout, we compared relative abundance (catch-per-unit-effort (CPUE) data) to selected habitat attributes in a representative sample of the studied streams in the Athabasca River drainage. In total, 16 streams and 34 individual sample sites were included in the analyses. From a list of 40 habitat variables, 25 were selected for statistical treatment (i.e., comparison to fish CPUE data). Seven habitat variables were significantly correlated with rainbow trout abundance. Three of these, stream gradient, elevation and percentage of high quality runs/pools, were positively correlated (i.e., higher numbers of fish with increasing stream gradient, elevation and percentage of high quality holding areas). The remaining four variables were negatively correlated with rainbow trout CPUE; these included channel depth, variability of flows following spawning, drainage basin area and conductivity. Based on the array of sample sites selected, we were able to provide a description of a typical rainbow trout stream or stream reach. In general, rainbow trout appeared to prefer small streams or the upper reaches of larger systems (i.e., Order-2 and Order-3 stream settings). The results also indicated that rainbow trout were generally located at higher elevations (i.e., mean elevation of 1354 m for sites with higher CPUE values), and in reaches characterized by stream gradients averaging 18 m/km (for sites with higher CPUE values). Other features that were described included channel width, conductivity and the presence of other fish species. This statistical method of predicting which habitat variables are significant appeared to be a useful tool; however, additional application is required to draw firm conclusions.

7.4 REVIEW OF LAND USE EFFECTS ON AQUATIC ENVIRONMENTS

The study included a review of the effects of major land use activities on fish and fish habitat. Impacts associated with linear developments (roads, pipelines, seismic lines, power transmission lines) and timber removal activities were discussed.

7.4.1 Linear Developments

Linear developments have the potential to adversely affect aquatic habitat. Of particular concern are roads built to service resource developments such as timber harvesting, mining and oil and gas extraction. Roads can affect fish populations by 1) delaying or blocking fish migrations at culvert installations, 2) adding sediment during the construction and operational phases, and 3) increasing angler access to streams or stream reaches that previously were protected due to their inaccessibility.

Locating a culvert crossing in the lower reach of a tributary can be particularly harmful if the installation impedes fish migrations (i.e., due to a poor design, construction or maintenance). This reduces the value of the fishery in the entire stream, and possibly nearby streams, if the system in question provides a centralized spawning and rearing function. While the technology and expertise exist to design and construct culverts that are passable to fish, in the past they have not always been implemented. Regular monitoring of culvert installations is essential in order to ensure that suitable fish passage conditions are maintained over the operational life of the road. If fish passage cannot be guaranteed at a downstream situated culvert crossing (through a proper culvert installation or a bridge) it may be necessary to locate the stream crossing near the headwaters (i.e., reduced risk of encountering fish migrations).

Fish passage concerns in the Foothills Model Forest region are considerable because many of the smaller streams, which are suitable for culvert placement from an engineering perspective, are inhabited by sportfish species with demonstrated migratory tendencies (i.e., bull trout, Arctic grayling, mountain whitefish, rainbow trout).

Sediment produced at road crossings and approaches, during the construction and operational phases, should be a major concern to resource managers. Short-term or chronic input of sediment can adversely affect fish and fish habitat in a variety of ways including: direct mortality due to gill damage (particularly in immature age-classes), reduced feeding success and growth rates, reduced density and diversity of periphyton and benthic macroinvertebrates, sedimentation of spawning substrates, and infilling of prime holding areas (e.g., deep pools used during summer low flow events and winter). Because of the potential for sediment production and subsequent downstream travel, the location of stream crossings needs to be carefully assessed. It is essential that critical habitat (e.g., spawning areas) be identified prior to road siting. There is an increased risk to the stream fishery when road crossings are located in the upper reaches, due to the potential for the entire stream to be affected (i.e., downstream travel of sediment). This is particularly true when considering crossings on smaller, lower gradient streams which have a reduced capability to cleanse themselves. It is important that managers weigh the relative merit and risk of placing road crossings in upper versus lower reaches (i.e., fish passage concerns at culvert installations versus potential sediment problems).

Expanding a network of roads into a formerly inaccessible watershed will result in increased angling pressure. Overfishing has been implicated as a major factor in the depletion of sportfish stocks (particularly bull trout, Arctic grayling, and mountain whitefish). This may be of particular concern in the Foothills Model Forest region due to the presence of relatively unproductive stream habitats (i.e., fish populations inhabiting these systems characterized by slow growth, low population densities, etc.). More stringent regulations (e.g., reduced bag limits, catch and release, bait ban) should be implemented prior to the opening of new road systems to provide immediate protection.

7.4.2 Timber Harvesting

Perhaps the greatest concern associated with timber harvesting is the potential removal or disturbance of vegetation in the riparian zone. The riparian zone contributes to the fisheries suitability of streams in a variety of ways, including: 1) contributes to stream bank stability, thus reducing erosion and sediment input, 2) is a significant source of organic matter which can provide the majority of energy input in headwater stream communities, 3) improves water quality by removing nutrients that may have increased due to land use, 4) provides channel shading which moderates water temperature, 5) reduces water velocity and erosive energy during peak flow periods, and 6) is a continual source of Large Woody Debris (LWD) which is essential instream cover for fish. The role of the riparian zone in maintaining the suitability and diversity of aquatic habitat cannot be underestimated.

Extensive timber harvesting in the watershed can also affect aquatic habitats by 1) increasing sediment loading into streams via surface soil erosion, mass-soil failures and stream bank erosion, 2) increasing annual water yield and peak flows, with decreasing base flows during low flow periods, 3) increasing water temperatures, 4) increasing nutrient inputs, and 5) changing physical instream habitat characteristics (channel morphometry, availability and quality of holding areas, etc.) due to altered flow regime, and increased sediment. Many of the negative impacts to fish and fish habitat associated with forestry activity are often attributable to road construction. The ability to predict the impact of altering the watershed through largescale forest removal on fish populations in the Foothills Model Forest is limited at present due to the lack of information on detailed habitat requirements and the distribution of critical habitats.

8.0 CONCLUSIONS

Following a review of information on the habitat requirements of four key sportfish species (rainbow trout, Arctic grayling, bull trout, mountain whitefish) in the Foothills Model Forest study area, models currently available to describe and predict habitat suitability, and impacts due to land use we conclude the following:

1. The present inventory of streams and rivers (which includes over 3300 km of fluvial habitat), and their fish populations in the FMF area is incomplete. Because of variable program requirements and sampling regimes, historical data sets are not readily comparable. Furthermore, a sizeable portion of the work is dated (e.g., 15 or more years old). Most of the information has not been entered into a centralized computer database, particularly the work done by industry and consultants. Also, very few studies have been carried out in a sufficient level of detail to be useful for monitoring changes in habitat or fish populations in response to an alteration in land use or fisheries management strategy. The ongoing inventory program being conducted by the FMF should provide information to address some of the deficiencies of the historical data.

2. It is generally accepted that aquatic habitats in the FMF study area are relatively unproductive due to a number of factors (i.e., short annual growing season, 6 - 7 months of ice cover, variable annual flow regimes, low nutrient levels).

3. Very little is known about the winter ecology of fish populations in the FMF study area. This is surprising given the fact that many streams are under ice cover for at least one half of the year. The lack of specific information regarding the location of overwintering sites and prevailing habitat conditions reflects the difficulty in obtaining this type of data. Of particular concern are the numerous small streams which presumably experience the harshest winter conditions. Many of these streams may be used only during the open-water season, at least by the mature age-classes (i.e., adults spawning in the small stream and overwintering in the parent stream); although, young-of-the-year and juveniles of some species (e.g., Arctic grayling) may reside in these streams year round.

4. Seasonal movement patterns of the four target species remain largely unknown on a stream-specific basis. However, there is sufficient information available to indicate that spawning, feeding, and overwintering movements are extensive. This has additional implications when considering the development of roads due to possible fish blockages caused by improperly designed or maintained culverts. These impacts can have widespread implications. Seasonal use of small streams in the FMF study area has been documented. It is our opinion that the importance of these systems to the entire ecology of the area has been underestimated.

5. Habitat suitability/assessment models specific to the FMF region are currently not available. Habitat Suitability Index (HSI) Models have been developed by the U.S. Fish and Wildlife Service, however, they need to be tested, “fine tuned”, and verified prior to their application in the FMF area. This will involve determining which of the wide array of habitat parameters are most critical in determining fish utilization patterns. The HSI Model approach appears to be applicable to the FMF area. It has value in that it can be applied at varying levels of detail (i.e., from simple narrative model to complex arithmetic model), thus allowing for improvements in model accuracy as additional data becomes available.

6. Numerous studies in other areas have established that increased sediment loading from timber harvesting, oil/gas development, and other land use activities seriously impact aquatic habitats. Increased sediment production is often associated with development of roads (temporary and permanent). It is evident that sediment is also a serious concern in the FMF area due to the specific terrain and climate characteristics and demonstrated sensitivity of key fish species (e.g., bull trout, Arctic grayling). The literature reviewed suggest that other potential impacts may also occur as a result of land use activities (e.g., alteration in flow regime associated with forest removal and alteration in availabilities of large-woody-debris).

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APPENDIX A

DESCRIPTIONS OF HABITAT SUITABILITY MODELS

APPENDIX A1

RAINBOW TROUT HABITAT SUITABILITY MODEL

A.1.1 Riverine Model

The HSI scores can be derived for a single life stage, a combination of two or more life stages or all life stages combined. With the exception of C_E, an HSI is obtained by combining one or more life stages with the C_O component. The following will briefly discuss the methods provided by Raleigh et al. (1984) for determining the HSI value.

Equal Component Value

The equal component value method assumes that each component exerts an equal influence when determining the HSI. This method should be used unless the available information indicates that individual components should be weighted differently.

$$HSI = (C_A \, x \, C_J \, x \, C_F \, x \, C_E \, x \, C_0)^{\frac{1}{N}}$$

Where N = the number of components in the equation

There will be a minimum of two components (i.e., the life stage component[s] and C_O) unless only C_E is being evaluated, then HSI = C_E.

Unequal Component Value

This method is utilized when data suggests that components should have different weighting when calculating the HSI. The method utilizes a life stage approach with the five components (i.e., C_A, C_J, C_F, C_E, and C_O). In this method the C_O component is subdivided into two portions; food (C_{OF}) and water quality (C_{OQ}). The model assumes that the C_{OF} subcomponent can either increase or decrease the suitability of the habitat by affecting the growth of all life stages except the embryo. The C_{OQ} subcomponent is assumed to equally influence all other model components. This method assumes that the water quality is excellent (i.e., $C_{OQ} = 1$) and that when C_{OQ} is <1 the HSI is decreased.

$$C_{OF} = \frac{(V_9 \, x \, V_{16})^{\frac{1}{2}} + V_{11}}{2}$$

Calculating the subcomponents of C_O:

Note: Should any variable be ≤ 0.4, then C_{OQ} = the lowest value.

$$C_{OQ} = (V_1 \, x \, V_3 \, x \, V_{13} \, x \, V_{14})^{\frac{1}{4}}$$

Calculation of the HSI

Once the subcomponents of C_O are determined, the HSI can be calculated in two ways: (1) the noncompensatory or (2) compensatory method.

Noncompensatory Method:

This option assumes that degraded water quality conditions can not be compensated by good physical habitat conditions. Raleigh et al. (1984) suggest that this method be utilized for small streams (i.e., <5 m wide) or for systems with persistently degraded water quality.

$$HSI = (C_A \; x \; C_J \; x \; C_F \; x \; C_E \; x \; C_{OF} \;)^{\frac{1}{N}} \; x \; C_{OQ}$$

Where N = the number of components and subcomponents inside the parentheses or, if the model or subcomponents are weighted, N=the summation of weights selected.

Note: Should any component be ≤ 0.4, then HSI = the lowest component value.

If only C_E is being evaluated, then HSI = $C_E \times C_{OQ}$.

Compensatory Option:

The compensatory method assumes that degraded water quality can be partially compensated by good physical habitat. Raleigh et al. (1984) suggest that this method is most applicable to large rivers (i.e., >50 m wide) in which poor water quality conditions are transient.

$$H'SI = (C_A \; x \; C_J \; x \; C_F \; x \; C_E \; x \; C_{OF} \;)^{\frac{1}{N}}$$

Where N = the number of components and subcomponents inside the parentheses or, if the model or subcomponents are weighted, N=the summation of weights selected.

Note: Should any component be ≤ 0.4, then HSI_ = the lowest component value.

If only C_E is being evaluated, substitute C_E for HSI_.

Then: If C_{OQ} is <HSI_, then HSI = HSI_×[1-(HSI_-C_{OQ})]

If C_{OQ} is >HSI_, then HSI = HSI_

A.1.2 Lacustrine Models

The following should be utilized when evaluating the suitability of lacustrine habitat to support rainbow trout. This model contains only two components: water quality (C_{WQ}) and reproduction (C_E).

$$C_{WQ} = (V_1 \, x \, V_3 \, x \, V_{13})^{\frac{1}{3}}$$

Water Quality:

Note: Should any of the suitability indices values for V_1 or V_3 be <0.4 then C_{WQ} is the lowest value of V_1 or V_3.

$$C_E = (V_2 \, x \, V_3 \, x \, V_5 \, x \, V_7 \, x \, V_{16})^{\frac{1}{5}}$$

Embryo:

Lacustrine HSI:

$$HSI = (C_{WQ} \, x \, C_E)^{\frac{1}{2}}$$

Note: If only the lacustrine habitat is to be evaluated then HSI = C_{WQ}.

The lacustrine model requires that a tributary stream suitable for spawning and embryo development be available.

APPENDIX A2

ARCTIC GRAYLING HABITAT SUITABILITY MODEL

A.2.1 Standard Habitat Suitability Model

The standard Arctic grayling HSI model was development by Hubert et al. (1985). As with the rainbow trout model, the variables which comprise the two components have to be measured in the field. The result of each variable is then assigned a suitability index (SI) value which is then used in determining the HSI value for each component of the model. The HSI value can be determined for a single component (i.e., A_1 or A_2), or for both components combined.

Calculating Components

Spawning, Embryo and Fry (A_1):

A_1 = The lowest value of V_1, V_2, V_3, V_4, V_5 or V_6 **(9)**

Adult and Juvenile (A_2):

A_2 = The lowest value of V_7, V_8, V_9, or V_{10} **(10)**

When determining the HSI value for a single component then:

HSI = The value of the component in question (i.e., A_1 or A_2) **(11)**

If both components are used in the determination of the HSI value then:

HSI = A_1 or A_2, whichever is the lower value **(12)**

A.2.2 Modified Habitat Suitability Model

As part of the aquatic investigations for the SandAlta Project in the early 1980's R.L. & L. Environmental Services Ltd. developed a modified Arctic grayling HSI model to determine the suitability of habitat in Hartley Creek, Alberta. After extensive evaluation of scientific literature, as well as experience gained by the investigators in northern Alberta and other regions of northern Canada which are inhabited by Arctic grayling, a model was developed which examined the life stage habitat requirements in more detail than the model proposed by Hubert et al. (1985). Figure 4.3 illustrates the components in the modified HSI model. The methodology for the development of this model is described in O'Neil et al. (1982b). This model has been successfully applied to Hartley Creek. As well R.L. & L. Environmental Services Ltd. has performed internal reviews on data sets collected from several other watersheds in northern Alberta which are inhabited by Arctic grayling (e.g., Sundance Creek) and the modified model has proven to be applicable to the other test streams (Jim O'Neil, R.L. & L. Environmental Services Ltd., Edmonton, Alberta, unpublished data).

Within the variables listed in the model, several were identified as being the most important to determining the habitat suitability for Arctic grayling. The factors used in establishing which variables to use in determining HSI were:

- scientific literature sources which suggested that certain variables are more important or should be given more weight than others in the HSI calculations;
- applicability to the situation, for example Hartley Creek was a small unimpacted watershed and the water quality (V_6) was considered to be equal to 1 in all cases, and therefore water quality was not a determining factor and not included in HSI calculations; and
- availability of habitat information at the required level of detail.

On the other end of the spectrum of HSI models, a simple descriptive model was developed and successfully utilized to characterize and assess habitat suitability for Arctic grayling in the Liard River of northern British Columbia and the Northwest Territories (O'Neil et al. 1982c). In this model the HSI rating system was modified to provide a wider range of suitability characterizations (Table A.1).

Table A.1 Habitat suitability scheme developed for the application to Arctic grayling habitat in the Liard River[1]

HSI Suitability Rating	Subjective Suitability Rating
<0.30	very low
0.31-0.45	low
0.46-0.55	low to moderate
0.56-0.65	moderate to high
0.66-0.75	high
>0.75	very high

[1] O'Neil et al. (1982c).

APPENDIX B

HABITAT AND GEOMORPHIC CHARACTERISTICS

Table B1 Possible variables contributing to, or acting as indicators of, habitat suitability for the four target species (rainbow trout, bull trout, Arctic grayling, mountain whitefish).

A.		**Drainage Basin**
	1	Stream order (Strahler Classification)
	2	Average gradient (map generated)
	3	Stream type (Rosgen Classification)
	4	Basin/reach elevation
	5	Watershed erosion potential (e.g., slope x soil type/stability)
	6	Drainage basin area
	7	Location of site or reach (e.g., distance from mouth, proximity to headwaters)
B.		**Channel Morphology/Flow**
	8	Channel width (rooted)
	9	Channel width (wetted)
	10	Channel width ratio (rooted/wetted) (at low flow)
	11	Channel depth (thalweg or transect) (at low flow)
	12	Width to depth ratio
	13	Entrenchment ratio (flood width/bankfull width)
	14	Slope (field generated/site specific)
	15	Sinuosity (stream length/valley length)
	16	Channel materials (e.g., D90, D50, sediment content)
	17	Mean annual discharge
	18	Average flow during summer (as % of average annual flow)
	19	Average flow during winter (as % of average annual flow)
	20	Ratio of spawning flows to incubation/pre-emergence flows
	21	Stream power (slope x discharge x constant) (e.g., at 1:2 and 1:10 year floods)
	22	Suspended sediment (turbidity) levels during summer (average, max.)
C.		**Fish Habitat Variables (Biological, Physical, Chemical)**
	23	Ice conditions (frazil, anchor, ice cover)
	24	Maximum summer water temperature
	25	Average summer water temperature
	26	Water temperature during egg incubation/pre-emergence period
	27	Minimum dissolved oxygen during summer
	28	Average dissolved oxygen during summer
	29	Minimum dissolved oxygen during winter
	30	Average dissolved oxygen during winter
	31	Nutrient levels (e.g., N, P, TDS)
	32	Presence/location of migration barriers
	33	Influence of beaver (number of dams/km, % of reach impounded)
	34	Contribution of erosional habitats (% riffle/run/pool vs. flat)
	35	Contribution of high quality (e.g., >1.5 m) holding pools/runs
	36	Availability of habitat in preferred range (depth/velocity/substrate) for spawning
	37	Availability of instream cover (LWD, depth turbulence, boulders, undercuts) (% of total channel area)
	38	Availability of bank cover (% total channel length)
	39	Extent of eroding bank (% of total bank length)

	40	Availability of invertebrate production areas (% of channel in riffle with preferred depth, velocity and sediment content)