Keeyask Generation Project

Environmental Impact Statement

Supporting Volume

Aquatic Environment

ᑮᔭᐢᒃ
KEEYASK
Hydropower Limited Partnership

June 2012

SECTION 4
LOWER TROPHIC LEVELS

KEEYASK
Hydropower Limited Partnership

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF PHOTOS

LIST OF APPENDICES

4.0 LOWER TROPHIC LEVELS

4.1 GENERAL INTRODUCTION

Lower trophic levels, as discussed in this document, include all aquatic organisms apart from fish that occupy the aquatic environment, including algae, aquatic plants, zooplankton, and macroinvertebrates. The overall Aquatic Environment Study Area (Section 1) encompasses a diverse range of habitats, from relatively large rivers to streams, a variety of sizes of lakes, and flooded terrestrial areas, and as such harbours many lower trophic groups. Changes in the abundance and distribution of these groups as a result of chemical and physical changes in habitat are an important linkage to effects to fish.

The importance of lower trophic levels to fish communities is recognized in the *Fisheries Act*, which includes in the definition of fish habitat, the food sources on which fish depend to carry out their life processes (*e.g.*, growth). An understanding of the existing lower trophic level community structure will allow for a more accurate prediction of potential impacts of the proposed Generation Project on fish populations.

The lower trophic levels program focussed on four groups, as follows:

- Phytoplankton (also referred to as algae; Section 4.2);
- Aquatic macrophytes (also referred to as aquatic plants) and attached algae (Section 4.3);
- Zooplankton (Section 4.4); and
- Aquatic macroinvertebrates (Section 4.5).

For each group, the approach to the assessment (study area, data and information sources, and assessment approach), information pertaining to the environmental setting, and assessment of the Project effects (during construction and operating periods, mitigation, residual effects, environmental monitoring and follow-up) are presented.

From a biodiversity and conservation perspective, the Aquatic Environment Study Area is not unique. The area is similar to the aquatic environment in much of the northern boreal forest of Manitoba, Ontario, and western Quebec. Within the lower trophic communities investigated between 1997 and 2006 no 'species of conservation concern' were identified. This term includes species that are rare, disjunct (discontinuous or separated distribution), or at risk throughout their range, or the portion of their range within Manitoba, and in need of further research. Also included are species listed under *The Manitoba Endangered Species Act* and the *Species At Risk Act* (SARA), and those that have special designation by the Committee On the Status of Endangered Wildlife In Canada (COSEWIC). In the York Landing Arm of Split Lake in 2000, a previously unreported species of caddisfly in Manitoba, *Molannodes tinctus* (Zetterstedt), was found near the mouth of the Aiken River. This species has been documented in Alaska, the Yukon, and sporadically in central Saskatchewan and northern Ontario.

4.2 PHYTOPLANKTON

4.2.1 Introduction

The phytoplankton consists of small, aquatic, plant-like organisms (*i.e.*, algae) that are most often found suspended or entrained in the water column. Several groups of freshwater algae comprise the phytoplankton: chrysophytes (Chrysophyceae [yellow-green or yellow-brown algae]), diatoms (Bacillariophyceae), chlorophytes (green algae), cyanophytes (blue-green algae or cyanobacteria), dinoflagellates (Peridineae), cryptophytes (cryptomonads) and euglenophytes (Photo 4-1). Many other aquatic organisms rely on phytoplankton, directly or indirectly, as a food source. Consequently, changes in phytoplankton abundance or composition can result in changes to invertebrate and fish populations. For these reasons, phytoplankton biomass and species composition were determined for lakes sampled in the Aquatic Environment Study Area.

Source: North/South Consultants Inc. [P. Badiou], 2004

Photo 4-1: A type of diatom (*Gyrosigma* sp.) found in the Aquatic Environment Study Area

Measurement of chlorophyll *a* (a green pigment found in aquatic plants and algae) in water is commonly used as an indicator of the amount (biomass) of algae growing in the water, and in turn, as an indicator of the productivity of an aquatic ecosystem. However, this method is not very sensitive and does not provide any information on the type of phytoplankton present. Furthermore, because the chlorophyll *a* content varies between species of phytoplankton (0.3–3.0% of dry weight among algal species), the concentration of chlorophyll *a* may not accurately represent the absolute quantity of phytoplankton present (Lee 1980).

The growth of photosynthetic organisms is limited to the euphotic zone, which extends from the lake surface to the lower limit at which there is sufficient light for photosynthesis (depends on water clarity). Rates of production by photosynthetic organisms are also strongly affected by the availability of nutrients, temperature, and water movement. Of the nutrients, nitrogen and phosphorus tend to be required in the largest amounts and their supply frequently determines the quantity and type of producers observed. The rates of physiological processes (*e.g.*, photosynthesis, respiration, and reproduction) tend to increase with temperature to an upper limit, after which warmer temperatures are harmful. Water movement plays a key role in determining the productivity of photosynthetic organisms. A certain degree of wave action or mixing within the euphotic zone is essential to maintain supplies of nutrients and carbon dioxide; however, excessive mixing can decrease the extent of the euphotic zone by increasing turbidity. Because hydroelectric development may affect various factors that influence phytoplankton growth and survival (*e.g.*, thermal regimes, nutrient concentrations, water clarity, and hydrological cycles) (see Section 2 and Section 3 for assessments of water quality and aquatic habitat, respectively), the phytoplankton community may be altered in regulated systems.

4.2.2 Approach and Methods

4.2.2.1 Overview to Approach

The approach taken for the phytoplankton effects assessment was similar to the general approach used for other aquatic environment components and was comprised of two major steps:

- A description of the existing aquatic habitat conditions to provide the basis for assessing the potential effects of the Project on these components; and
- An effects assessment in which the predicted post-Project environment was described and changes from existing environment quantified.

An ecosystem-based approach was employed to assess the potential impacts of the Project on the phytoplankton community. Information presented incorporates findings from other aquatic environment components (*e.g.*, surface water quality and aquatic habitat). This approach is consistent with the views held by the KCNs, and widely held ecological views, that all components of the aquatic environment are important to maintaining the whole, and that all organisms are interdependent and, therefore, of importance and value.

The environmental setting is described using several sources of information, including existing published information and studies conducted specifically as part of the EIS of the Project between 1999 and 2005. Potential Project-related effects on the phytoplankton community were assessed using basic models (*i.e.*, simple conceptual models, quantitative models based on changes in habitat area, and qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments). These sources of information and effects assessment approaches are described in the following sections.

4.2.2.2 Study Area

The study area for phytoplankton investigations extends along the Nelson River from Split Lake and adjoining waterbodies downstream to Stephens Lake in the east (Map 1-2). The magnitude of physical change (*e.g.*, changes in water levels and flows) differs substantially among areas (Section 3.2.2) and, consequently, the phytoplankton study area was divided into three areas on the Nelson River as follows:

- Split Lake area (Split Lake and adjoining waterbodies, including Assean Lake and Clark Lake). This area is upstream of any direct Project influence. The phytoplankton community in this area was described to provide supporting information for studies of surface water quality and other aquatic biota (Section 2 and Section 5).
- Keeyask area (Nelson River extending from the outlet of Clark Lake to approximately 3 km downstream of Gull Rapids, *i.e.*, hydraulic zone of influence, and tributary streams). Project-related changes to the water regime and direct losses of habitat due to the presence of the generating station (GS) will occur within this reach (Section 3.2.2). This area was subdivided at Gull Rapids, as the rapids mark a boundary between the reservoir and downstream environment in the post-Project environment.
- Stephens Lake area (Stephens Lake and adjoining waterbodies). This area is immediately downstream of the Keeyask area and the Project will not affect the water regime. Stephens Lake, as the reservoir of the Kettle GS formed in the early 1970s, provides a useful proxy to assist in predicting effects of the Project (Section 1). Changes in the upstream environment as a result of the Project may also affect the phytoplankton community in Stephens Lake.

The majority of lower trophic levels investigations were conducted in the Keeyask area, as this area will be directly affected by the Project. Aquatic biota was also described as part of the assessment of the north and south access roads stream crossings.

4.2.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for phytoplankton are detailed below.

A number of phytoplankton community studies have been previously conducted in the Aquatic Environment Study Area. These data collection programs were primarily focussed on the effects of hydroelectric generating stations (*e.g.*, construction and operation of the Kettle GS) or on the effects of the Churchill River Diversion (CRD)/Lake Winnipeg Regulation (LWR) project and were largely limited to GS reservoirs along the lower Nelson River, and Split and Stephens lakes.

Phytoplankton samples were collected from Split Lake and below the Kettle GS in the early 1970s as part of the Lake Winnipeg Churchill and Nelson River Study Board (LWCNRSB) program (Hecky and Harper 1974). In the late 1980s, phytoplankton data (including chlorophyll *a* concentrations) were collected in Split and Stephens lakes as part of the Manitoba and Federal Ecological Monitoring Programs (MEMP and FEMP) (Livingston 1987, 1988, 1989; Ramsey *et al.* 1989; Green 1990; Janusz

1990a, c; Strange 1990). During the 1990s and early 2000s, limited phytoplankton data were collected for Manitoba Hydro by KCNs Members together with North/South Consultants Inc. (NSC) as part of the Lower Nelson River Forebay Monitoring Program (included Kettle, Long Spruce, and Limestone reservoirs and the lower Nelson River) (Schneider and Baker 1993; NSC 2012). The effects of previous hydroelectric development in northern Manitoba on the Split Lake Resource Management Area (RMA) were assessed as part of the Split Lake Cree Post-Project Environmental Review (PPER, Split Lake Cree - Manitoba Hydro Joint Study Group 1996a, b, c).

Phytoplankton sampling to determine abundance and community composition was conducted within Aquatic Environment Study Area lakes once during the open-water season in 1999, four times during the open-water season in 2001 and 2002, and once during the ice-covered seasons of 2001 and 2002. Chlorophyll *a* concentration, a relative measure of phytoplankton biomass, was also measured at the majority of water quality sampling locations (2001 to 2004), including several access road stream crossings along the north side and south side of the Nelson River (2003 to 2005). The detailed approach and methods for phytoplankton community studies conducted between 1999 and 2002 are presented in Appendix 4A and those for chlorophyll *a* sampling conducted as part of the water quality program between 2001 and 2005 are presented in Appendix 2C.

4.2.2.4 Assessment Approach

Given the complexity of the aquatic ecosystem, models were used for predicting effects of the Project. Within the aquatic assessment, the complexity of models employed depended on: the importance of the issue; availability of information or suitable models; and utility of modelling approaches.

Basic model types used to assess potential Project effects on the phytoplankton community were:

- Simple conceptual models (*e.g.*, alteration in off-current areas with respect to nutrient and total suspended solids [TSS] concentrations leads to an effect on phytoplankton biomass). The scientific literature was used to describe and support linkages to the Project.
- Quantitative models based on changes in aquatic habitat area (*e.g.*, calculation of total phytoplankton biomass [*i.e.*, 'standing stock'] increase post-impoundment based on the predicted increase in reservoir volume) over the short-term and long-term post-Project.
- Qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments. For example, Stephens Lake was used as a surrogate for long-term post-Project conditions in the Keeyask reservoir.

The evaluation of certainty for predicted effects was based in part on the agreement of predicted effects among the various approaches.

4.2.3 Environmental Setting

4.2.3.1 Overview and Regional Context

The environmental setting has been described based on available background data and information collected during the course of the Keeyask environmental studies. The phytoplankton community in the study area has been influenced by past hydroelectric development in northern Manitoba (*e.g.*, Kelsey GS, CRD, and LWR). Members of the Tataskweyak Cree Nation (TCN) indicated the effects of hydroelectric development included more common occurrences of algae in the Burntwood River, and Split, Clark and Gull lakes (E.E. Hobbs and Associates Ltd. 1993 in Split Lake Cree - Manitoba Hydro Joint Study Group 1996c; Socio-economic Environment, Resource Use, and Heritage Resources Supporting Volume). Members of Fox Lake Cree Nation have observed increased amounts of algae in their fishing nets (Fox Lake Cree Nation [FLCN] 2008 Draft).

Eighty-seven taxa of phytoplankton have been recorded in the study area between 1999 and 2002 (Appendix 4B); however, community composition is discussed below in terms of major groups. None of the identified species are listed as invasive on the Invasive Species Council of Manitoba website (Invasive Species Council of Manitoba 2012).

Phytoplankton abundance and composition varied between study area waterbodies and between years. In the open-water season, phytoplankton biomass was lower in 2001 than in 2002 and was considerably lower during the ice-cover season than during the open water season. In both years, peak phytoplankton biomass occurred in June at most sites. Lakes, with the exception of Assean Lake, were dominated by diatoms throughout most of the open water season. Phytoplankton biomass in the study area is at the lower (oligotrophic-mesotrophic) end of the general range reported for temperate zone waterbodies (Kalff and Knoechel 1978; Heinonen 1980 and 1982). The seasonal progression of the phytoplankton communities followed the general trend of early summer diatom peaks proceeded by an increase in cyanophytes/chlorophytes or cryptophytes in late summer and then a secondary peak in diatoms in fall. In the Assean River system, phytoplankton biomass in the open water season was generally much lower and the relative abundance of algal groups such as chrysophytes, chlorophytes, euglenophytes, and dinoflagellates were usually greater.

Chlorophyll *a* concentrations varied during the open water season. Seasonal mean chlorophyll *a* concentrations were generally similar among years for all sites. Typically, chlorophyll *a* concentrations were lowest in spring. Mean chlorophyll *a* concentrations at sites located off the mainstem of the Burntwood/Nelson River system, such as Assean Lake, and the Gull Lake tributary sites were low relative to those recorded on the mainstem. The absence of consistent differences in chlorophyll *a* concentrations among sites over a considerable area of study suggests that the presence of lakes does not result in an overall increase in phytoplankton as water moves through the study area. Primary production is typically limited under ice-cover due to low temperatures and reduced light levels and, as expected, chlorophyll *a* was consistently lower and often undetectable in samples collected under the ice. The range of chlorophyll *a* concentrations observed in study area waterbodies was indicative of oligo- to mesotrophic conditions. Seasonal variations in the phytoplankton community and chlorophyll *a* concentrations are typical of north temperate ecosystems where light and temperature vary considerably over the year.

Suitable growing conditions for phytoplankton are strongly influenced by the stability of the water column. Studies in several northern Manitoba lakes and reservoirs have indicated that the available phosphorus does not limit phytoplankton growth (*e.g.*, Southern Indian Lake, Hecky and Kilham 1988). Rather, phytoplankton growth is limited by wind-induced turbulence in combination with turbid water. It is unlikely that phytoplankton are a major source of production in most regions in the study area given that the water is turbid, wind-induced wave action causes considerable mixing, and retention time of water is relatively short.

The range of chlorophyll *a* concentrations observed in Keeyask waterbodies was indicative of low to moderate levels of primary productivity (oligo- to mesotrophic conditions). Overall, there is poor correlation between phosphorus and chlorophyll *a* in the study area, which indicates that factors other than nutrients limit algal growth (Section 2.4.2.1.5). This is further supported by concentrations of phosphorus and the low phytoplankton biomass observed in the study area. A higher trophic status would be assigned to the study area on the basis of phosphorus concentrations than on the basis of chlorophyll *a* concentrations. Phosphorus concentrations in the study area reflect meso-eutrophic to eutrophic conditions based on the Canadian Council of Ministers of the Environment categorization schemes (CCME 2004).

Phytoplankton composition and biomass within the study area are comparable to other waterbodies in northern Manitoba (Manitoba Hydro and Nisichawayashik Cree Nation [NCN] 2003). Lakes of the Burntwood River system, including Cranberry, Sesep, Wuskwatim, Opegano, Birch Tree and Kinosaskaw lakes, Wuskwatim Brook, and portions of the Burntwood River had similar phytoplankton community seasonal trends and variability as observed for lakes in the study area.

Similar to the Keeyask area lakes, phytoplankton biomass in the Burntwood River system generally peaked in June when diatoms dominated the community. As the season progressed, diatoms decreased in biomass while cyanophytes, chlorophytes, or cryptophytes increased. By fall, diatoms generally dominated the phytoplankton community again. Similar to the study area, cyanophyte blooms occurred in late summer in the Burntwood River system.

Phytoplankton biomass was also found to be greater in the Burntwood River system during the open water season than during the ice-cover season.

4.2.3.2 Split Lake Area

4.2.3.2.1 Split and Clark Lakes and the Nelson River System

Dominant phytoplankton genera in Split Lake in the early 1970s and late 1980s were seasonally variable, changing from a diatom dominated community in early summer/spring to one dominated by cryptophytes and chlorophytes in late summer. In some years, diatoms remained dominant throughout the open-water period and in other years cyanophytes dominated the community in July and August (Hecky and Harper 1974; Livingston 1987, 1988, and 1989; Janusz 1990a). Generally, maximum algal biomass observed in the Nelson River system (Split Lake included) in 1972 was less than that observed in the Churchill and Rat-Burntwood River system (Hecky and Harper 1974). The lower net production was probably due to the lower transparency and lower residence time of water in the lakes of the Nelson River system.

Mean phytoplankton biomass in Split Lake in 1987–1988 was 25–50% higher than reported in 1972-73. Major differences in community composition between the pre- and post-CRD/LWR studies included the generally low importance of cyanophytes in 1987–1988 and the greater importance (at least in 1987) of diatoms, compared to the findings from 1972 (Ramsey *et al.* 1989).

Similar to other sites in the study area, phytoplankton biomass in Split and Clark lakes was lower in 2001 than in 2002 (Table 4-1). Peak phytoplankton biomass generally occurred in spring (Figure 4-1 and Figure 4-2) and was approximately four to 148 times greater in the open water season relative to the ice-cover season.

In spring, diatoms were the dominant algal group at all sites in Split and Clark lakes (Figure 4-1 and Figure 4-2). In early summer, diatoms remained dominant at the majority of sites. In late summer 2001, a cyanophyte bloom occurred at most sites, such that cyanophytes were either the dominant algal type or co-dominant with diatoms. A cyanophyte bloom did not occur in Split or Clark lakes in 2002; however, the relative abundance of diatoms was typically lower in late summer than early summer throughout the area. In late summer 2002, chlorophytes were the dominant algal group at the downstream end of Split Lake while cryptophytes were the dominant algal group at the upstream end of the lake. At this time, cryptophytes were also a significant portion of the phytoplankton community at three other sites (SPL-4, -6, and -8). In fall, diatoms were again the dominant algal species in the lakes.

Large chrysophytes generally comprised a small component of the phytoplankton community throughout the open-water season in Split and Clark lakes. Other algal species identified were euglenophytes and dinoflagellates. The contribution of these groups to the phytoplankton biomass of the two lakes was generally small and their presence was variable between sites and times.

Under the ice, diatoms generally dominated the phytoplankton community in Split Lake in both years (Figure 4-3). Unlike other samples collected in Split Lake, the phytoplankton composition at the site located near the mouth of the Aiken River (SPL-5) in 2002 was co-dominated by dinoflagellates and cryptophytes (Figure 4-3).

Chlorophyll *a* concentrations in the area ranged from less than 1 to 15 micrograms per litre (μg/L) during the open-water season, and from less than 1 to 2 μg/L under the ice (Table 4-2 and Table 4-3). Chlorophyll *a* concentrations in the area were similar to other mainstem sites throughout the study period (Figure 4-4). Based on the open-water values, trophic categorization schemes for lakes would classify Split and Clark lakes as mesotrophic, while a trophic categorization scheme developed for streams would classify the Burntwood and Nelson mainstem and tributary sites as oligotrophic (Table 4-2).

4.2.3.2.2 Assean River System

Phytoplankton biomass, composition, and seasonal succession in Assean Lake were different from other sites within the study area. In both years, phytoplankton biomass was lowest in Assean Lake (Table 4-1). The phytoplankton composition in Assean Lake was more diverse relative to sites on the

Burntwood/Nelson River system and, generally, the algal species in Assean Lake were different from those at other sites.

The relative abundance of diatoms was much lower in Assean Lake than in the rest of the study area throughout the open water season, and the diatom species that were found in Assean Lake differed from the species found at other sites.

Phytoplankton composition during the ice-cover season on the Assean River system was very different from the composition observed on the Burntwood/Nelson River system. Furthermore, phytoplankton composition within Assean Lake was quite variable between sites and years. During the 2001 ice-cover season, the phytoplankton community in the west basin was dominated by chrysophytes, while chrysophytes and euglenophytes co-dominated the phytoplankton biomass in the east basin (Figure 4-3). In winter 2002, diatoms dominated the phytoplankton community in the west basin, while euglenophytes dominated the phytoplankton biomass in the east basin (Figure 4-3).

Chlorophyll *a* concentrations in Assean Lake were lower and less variable than other sites within the Split Lake area (Figure 4-4). Chlorophyll *a* concentrations ranged from 1–6 μg/L during the open water season and from less than 1 to 2 μg/L during the ice-covered season (Table 4-2 and Table 4-3). Based on the open water values, Assean Lake would be classified as oligotrophic-mesotrophic using trophic categorization schemes for lakes, which is less productive than the other lakes in the study area (Table 4-2).

4.2.3.3 Keeyask Area

No data or assessment of the effects of hydroelectric development on the phytoplankton community prior to 1997 in the reach of the Nelson River between Clark Lake and Stephens Lake were located in the published literature.

The phytoplankton community sampled in the Keeyask area as part of the environmental studies was similar to sites within the Burntwood/Nelson River during both the open water and ice-covered seasons (Figure 4-1 and Figure 4-2). Phytoplankton composition was similar at all locations within the area and was dominated by diatoms (Figure 4-1 and Figure 4-2). Cyanophytes, chlorophytes, and cryptophytes were also present.

In early summer, chlorophytes increased in relative abundance in Gull Lake; this did not occur at Gull Rapids or at the other sites in the Burntwood/Nelson River system (Figure 4-1 and Figure 4-2). In late summer 2002, cryptophytes were an important component of the phytoplankton community at the downstream site in Gull Lake, as was observed at several sites in Split Lake.

Chlorophyll *a* concentrations recorded in 1999 were lower than those typically observed within the Keeyask area in the fall of other sampling years (Table 4-2). Between 2001 and 2004, chlorophyll *a* concentrations within the area were similar to other mainstem sites in the study area. However, in 2003, one site downstream of Birthday Rapids recorded chlorophyll *a* concentrations well above the usual winter range: at Gull Rapids where the chlorophyll *a* concentration was approximately four times higher than the open water mean at that site, and was the single highest concentration recorded in the study area.

Chlorophyll *a* concentrations measured in small tributary streams of Gull Lake were found to be lower than concentrations in the remainder of the study area (Figure 4-4).

Based on the open water chlorophyll *a* values, trophic categorization schemes for lakes would classify Gull Lake as mesotrophic, while a trophic categorization scheme developed for streams would classify the Nelson River mainstem and tributary sites as oligotrophic (Table 4-2).

4.2.3.4 Stephens Lake Area

Considerable year-to-year variation in mean phytoplankton biomass observed pre-CRD/LWR (1972–1973) in the newly formed Kettle Reservoir/Stephens Lake (Hecky and Harper 1974) persisted into the late 1980s (Ramsey *et al.* 1989). As a result, there was no clear difference in mean phytoplankton biomass between pre-CRD/LWR and MEMP studies.

Prior to development, the phytoplankton community was distinctive from that found on the mainstem Nelson River; cryptophytes dominated the community in both 1972 and 1973 (Hecky and Harper 1974). In contrast, cryptophytes were not important contributors to community biomass on average post-CRD/LWR (Ramsey *et al.* 1989). Despite this change in community composition, the pattern of seasonal biomass fluctuation remained the same (highest in June and lowest in late summer).

As was observed in the early 1970s, phytoplankton biomass and composition varied considerably between MEMP study years (1987 and 1988) and among areas of the lake (Ramsey *et al.* 1989). There were no consistent differences in biomass or composition noted between backwater and mainstem sampling locations even though flushing rates on the mainstem locations were greater. Despite the considerable year-to-year variation in both studies, standing biomass in 1987–1988 was greater than in 1972–1973 (Ramsey *et al.* 1989).

Stephens Lake was similar to other sites in the study area in terms of overall phytoplankton biomass, composition, and seasonal succession. Relative to most other sites in the system, however, chlorophytes were high at the upstream site in Stephens Lake during the spring of 2002 (Figure 4-2). Additionally, in the early summer of 2002, cryptophytes contributed substantially to the total algal biomass at this same site. Under the ice in March of 2002, cyanophytes constituted a relatively high percentage of the biomass at the upstream site and, similar to Assean Lake, euglenophytes were present at the other site in Stephens Lake.

Chlorophyll *a* concentrations in this area ranged from 1–16 μg/L during the open water season and from less than 1 to 6 μg/L during the ice-covered season. This is similar to other mainstem sites. The highest chlorophyll *a* concentration measured during the open water season was in 2002 in Stephens Lake near the town of Gillam water intake (Figure 4-4). Based on the open water values, the trophic status of Stephens Lake was similar to other lakes in the study area and would be considered mesotrophic (Table 4-2).

4.2.3.5 Access Road Area

To obtain some measure of algal biomass, chlorophyll *a* samples were obtained during the open-water seasons of 2003, 2004, and 2005 (May only) at eight proposed Keeyask GS access road stream crossings sites, three on the north side of the Nelson River and five on the south side. Chlorophyll *a* concentrations ranged from less than 1 to 16 μg/L on the north side and from less than 1 to 5 μg/L on the south side (Table 4-4). Overall, chlorophyll *a* concentrations in 2003 were higher than those in 2004 (Table 4-4). Based on chlorophyll *a* concentrations, these sites were classified as oligotrophic and therefore representative of relatively low productivity (Dodds *et al.* 1998).

4.2.3.6 Current Trends

Historic phytoplankton community data were located for Split and Stephens lakes. These data were collected from Split Lake in the early 1970s as part of the LWCNRSB program and from Split and Stephens lakes in the late 1980s as part of the MEMP and FEMP. Comparison of these data sets with phytoplankton data collected as part of the Keeyask environmental studies for the purpose of assessing current trends is limited. There were differences in sampling and analytical methods employed among the studies, and phytoplankton abundance and composition varied considerably within waterbodies and among study years. However, qualitative comparisons of phytoplankton data over time are presented.

Generally, mean phytoplankton biomass appears to have increased in Split Lake, but not in Stephens Lake, since the early 1970s. However, phytoplankton biomass in the current study area remained at the lower (oligotrophic-mesotrophic) end of the general range reported for temperate zone waterbodies. The current seasonal progression of the phytoplankton communities in Split and Stephens lakes continued to follow the general trend observed historically of early summer diatom peaks proceeded by an increase in cyanophytes/ chlorophytes or cryptophytes in late summer and then a secondary peak of diatoms in fall. However, as was observed in the 1970s and 80s, phytoplankton abundance and composition varied between study years and among areas of the lake. In Split Lake, major differences in community composition between the pre- and post-CRD/LWR studies included the generally low importance of cyanophytes in the late 1980s in comparison to the early 1970s; however, in late summer 2001, a cyanophyte bloom occurred at most sites in Split Lake, such that cyanophytes were either the dominant algal type or co-dominant with diatoms. Cryptophytes dominated the Stephens Lake community in the early 1970s, but were not important contributors to community biomass in the late 1980s, however, in the early summer of 2002, cryptophytes contributed substantially to the total algal biomass in some areas of Stephens Lake.

Throughout the environmental studies, chlorophyll *a* concentrations for Split and Stephens lakes were all within the ranges observed at similar locations sampled between 1986 and 1989 (Green 1990; Ramsey 1991). From 2002 to 2004, the range of chlorophyll *a* concentrations measured in Split Lake was similar to samples collected near the community of Split Lake between 1980 and 2001 (Manitoba Water Stewardship 2002). However, in 2001, the range of chlorophyll *a* concentrations measured in Split Lake in this study exceeded the range reported by Manitoba Conservation.

Parameters that appear to have changed notably (temporal trend) in the north arm of Stephens Lake since the 1970s include chlorophyll *a* (Appendix 2E); mean chlorophyll *a* concentration measured in 2004 was lower than in the 1970s and 80s in the north arm of Stephens Lake and concentrations were also lower in 2004 in the north arm relative to the southern mainstem portion of the lake. Evaluation of potential temporal changes in the southern area of Stephens Lake from the 1970s to 2004 (open water seasons) indicated that mean chlorophyll *a* concentrations have ranged from approximately 3 to 8 μg/L (Appendix 2E, Figure 2E-7) in the southern area of the lake. Generally, the data indicate a fair amount of variability within a given sampling year and there are no temporal trends immediately evident from this information.

4.2.4 Project Effects, Mitigation and Monitoring

4.2.4.1 Construction Period

The following section considers potential effects related to the construction of the GS and south access road, and operation of the construction camp and north and south access roads during the construction period. The construction of the north access road was assessed in the Keeyask Infrastructure Project Environmental Assessment Report (KIP EA) (Keeyask Hydropower Partnership Ltd. 2009). Stream crossing locations are provided in Map 1-4.

An assessment of potential Project effects on the phytoplankton community during the construction period is based on the assessment of construction-related effects to surface water quality (Section 2.5.1, Table 2-12). The primary potential effect(s) on phytoplankton is related to inputs affecting water quality, such as increases in TSS concentrations and related variables (*i.e.*, turbidity) due to in-stream activities (*e.g.*, cofferdam placement and removal, river impoundment and diversion) and nutrient inputs (*e.g.*, with treated sewage effluent discharge to the mainstem of the Nelson River, with particulate materials [*i.e.*, TSS]). It is expected that construction effects (*i.e.*, inputs affecting water quality) will be managed through appropriate mitigation measures (Section 2.5.1), thereby reducing the duration and magnitude of any construction-related effects on the phytoplankton community.

Currently, phytoplankton biomass is relatively low at flowing water sites; phytoplankton tend to be relatively unimportant in lotic environments, as these planktonic organisms cannot maintain positive net growth rates (Hynes 1970).

4.2.4.1.1 Upstream of the Outlet of Clark Lake

No construction-related effects on the phytoplankton community are expected upstream of the outlet of Clark Lake as there are no linkages between Project construction and surface water quality in Split, Assean, or Clark lakes (Section 2.5.1).

4.2.4.1.2 Downstream of the Outlet of Clark Lake

The following sub-sections present the assessment of potential effects of construction activities on the phytoplankton community in the Keeyask area and downstream.

Changes to Water Quality

Total Suspended Solids, Turbidity, and Water Clarity

Overall, the activities with the greatest potential to increase TSS concentrations in the lower Nelson River during construction of the GS are related to cofferdam placement and removal, and river impoundment and diversion (Section 2.5.1.1). Effects of suspended fine sediments on phytoplankton are likely primarily related to its effect on light penetration; light attenuation by inorganic turbidity decreases the fraction of light absorbed by photosynthesizing algae. Generally, the installation and removal of cofferdams will generate an increase in TSS of less than 5 milligrams per litre (mg/L) above background (Section 2.5.1). Larger TSS increases are expected to be of relatively small magnitude and short duration. Peak levels are predicted to be up to 15 mg/L for one day or up to 7 mg/L for one month (Section 2.5.1). Increased TSS concentrations may be detectable in the river immediately below the construction site, but would diminish by approximately 30% through Stephens Lake. Increases downstream of the Kettle GS would be approximately 2 and 5 mg/L for one month periods during Stage II construction, and attenuate further downstream. Drainage of surface runoff to the Nelson River will be controlled through a Drainage Management Plan (as described in the Project Description Supporting Volume [PD SV]) to minimize the amount of sediment produced and the potential for sediment to enter watercourses. If the TSS concentration in water pumped out of cofferdam and excavation areas and in concrete wash water is greater than 25 mg/L the water will remain in a settling pond until it meets this TSS criterion before being discharged to the Nelson River. As the magnitude and duration of any increases in TSS are typically within the 30-day Manitoba Water Quality Standards, Objectives, and Guidelines (MWQSOG) for the protection of aquatic life (PAL) (an increase of 5 mg/L above background where background is less than or equal to 25 mg/L), the phytoplankton community may be somewhat negatively affected in this downstream environment as photosynthetic efficiency may be reduced, thereby somewhat limiting primary production (*i.e.*, small, undetectable reductions in phytoplankton biomass may occur in areas affected by elevated TSS concentrations during the construction period).

Nutrients

Nutrient (nitrogen and phosphorus) inputs with treated sewage effluent discharge to the mainstem of the Nelson River are not expected to be detectable in the fully mixed river condition, but concentrations may be elevated in the vicinity of the effluent outfall (Section 2.5.1.3.3). Additionally, any increases in nutrients associated with expected increases in particulate materials (*i.e.*, TSS) are expected to be small. As the expected level of increase in nutrient inputs to the Nelson River during the construction period is small, nutrient inputs will not have a measurable effect on phytoplankton beyond the immediate receiving environment. During the latter stage of the Stage II Diversion, when water levels are increased to near full supply level (FSL), flooding of organic materials is expected to lead to nutrient release to surface waters, thereby increasing concentrations of nutrients, notably over flooded habitat. These effects (*i.e.*, due to reservoir impoundment) are discussed in detail in the assessment of operation-related effects in Section 2.5.2.2 for surface water quality and Section 4.2.4.2 for phytoplankton.

Metals and Contaminants

Small amounts of metals will be introduced into the aquatic environment in association with construction activities that release sediments, as discussed in Section 2.5.1.6. However, given the proposed mitigation measures to manage sediment levels, these inputs are not expected to cause marked increases in metal levels and, consequently, will have no detectable effect on the phytoplankton community.

The presence and levels of hydrocarbons in the aquatic environment could potentially be affected by accidental spills or release of substances containing hydrocarbons (*e.g.*, diesel fuel, gasoline, lubricating oil, *etc.*). Other hazardous substances will also be used during the construction period. As described in Section 2.5.1.6, the release of significant quantities of hazardous substances to the aquatic environment as a result of accidental spills and releases is considered unlikely due to the development and implementation of good management practices.

4.2.4.1.3 South Access Road Stream Crossings

No response is expected as effects to surface water quality are predicted to be small due to the application of various mitigation measures (Section 2.5.1.7). Additionally, phytoplankton tend to be relatively unimportant in small streams and other lotic environments as these planktonic organisms cannot maintain positive net growth rates due to a variety of reasons, including downstream losses (Hynes 1970). The algae in the water column are typically benthic species sloughed from the stream bed.

4.2.4.1.4 Net Effects of Construction with Mitigation

Collectively, the above assessment points to the potential for small decreases in phytoplankton biomass during the construction period, most likely due to a reduction in light penetration from increases in TSS concentrations. Changes in biomass would occur over the short-term downstream of the outlet of Clark Lake

4.2.4.2 Operation Period

Phytoplankton (and therefore chlorophyll *a* concentrations) in large rivers are generally influenced by: concentrations of nutrients required for growth (*i.e.*, nitrogen and phosphorus); water temperature; light availability; and physical conditions in the river such as turbulence and velocity. Because hydroelectric development may affect various factors that influence phytoplankton growth and survival (*i.e.*, thermal regimes, nutrient concentrations, water clarity, and hydrological cycles), phytoplankton (biomass, community composition) may be altered in regulated systems.

4.2.4.2.1 Upstream of the Outlet of Clark Lake

No response is expected. Selection of a 159 m above sea level (ASL) reservoir elevation instead of a higher elevation will avoid Project-related effects as the Split Lake area is beyond the upstream extent of the expected hydraulic zone of influence.

4.2.4.2.2 Outlet of Clark Lake to the Keeyask Generating Station

Potential Project Effects and Proposed Mitigation

Operation-related pathways (*i.e.*, linkages to the Project) that were assessed for potential effects to the phytoplankton community included: changes in surface water quality (decrease in TSS along mainstem; increase in TSS, nutrients, organic carbon, colour in off-current areas for the initial period post-impoundment) (Section 2.5.2) and changes in reservoir water residence time (increase in water level and volume, reduction in water velocity) (Section 3.4.2.2). Summaries of predicted responses of phytoplankton to changes resulting from the operation of the Project are presented in Figure 4-5. Where feasible, the effects of these pathways were considered using modelling exercises (*e.g.*, quantification of potential effects), empirical information from Stephens Lake and other reservoirs in northern Manitoba, reservoirs in other northern temperate areas, and the scientific literature.

Assessment of Operation-Related Effects

Modeling Approach

Impoundment of rivers is generally associated with a large increase in phytoplankton biomass due to nutrient enrichment and increased water retention time (Henriques 1987). However, detectable changes in mean phytoplankton biomass along the mainstem are not expected as increased water residence time will remain too short to permit a measurable increase in phytoplankton biomass; although total biomass ('standing stock') would increase with the predicted increase in reservoir volume (approximate doubling in comparison to the existing environment) (Section 3.4.2.2). The lack of detectable effects may be attributed to high water flushing rates through the mainstem portion of the reservoir (*i.e.*, post-Project water residence time will be in the order of 15–30 hours, depending on flow; Section 3.4.2.2). Short retention times are often associated with high turbulence and a lack of thermal stratification; phytoplankton require a minimum retention time to allow development (McCartney *et al.* 2000). If rates of water movement through a reservoir exceed a few millimetres per second, little plankton will develop (Hynes 1970).

Off-current areas could experience periodic phytoplankton blooms (*i.e.*, small to moderate increases in biomass), depending on the balance between the positive effect of increased nutrients and the negative effect of light depletion, as water residence time in bays is estimated to be substantially longer than in the mainstem and could be up to one month long (Section 3.4.2.2). Reduced light transmission may moderate the effect of nutrient loading. High dissolved organic carbon (DOC) concentrations can affect primary productivity by influencing light penetration and adding carbon for processing. For example, benthic diatoms and total diatom concentrations increased significantly during conditions of high DOC concentrations and low water transparency, whereas planktonic forms decreased , in subarctic lakes (Pienitz and Vincent 2000). In Southern Indian Lake, northern Manitoba, high DOC concentrations decreased light penetration sufficiently to cause a switch from nutrient to light limitation of primary production (Hecky and Guildford 1984). Initial post impoundment conditions may favour bacteria over phytoplankton (Paterson *et al.* 1997). The addition of large amounts of newly flooded terrestrial organic matter may stimulate bacterial activity (increase the flow of carbon to higher trophic levels through the detrital pathway) and increase bacterial biomass (post-flooding food resource for zooplankton) in the medium term (5–10 years post-impoundment) instead of phytoplankton. Large increases in methane and CO_2 production following flooding would provide an indication of increased bacterial production.

Information from Other Reservoirs

The growth of phytoplankton tends to be limited in the riverine portions of the lower Nelson River system as the water is turbid (reduced light availability), wind-induced wave action causes considerable mixing, and retention time of water is relatively short. Because phytoplankton have relatively high growth rates they are less susceptible to downstream loss in short water residence systems in comparison to larger organisms, such as zooplankton. Therefore, phytoplankton in the Nelson River may be more limited by the other factors noted above, such as reduced light availability (*i.e.*, water is turbid and well mixed), in addition to the relatively short water residence times experienced.

Presently, mean chlorophyll *a* concentration and phytoplankton biomass observed at mainstem sites in the Keeyask area and Stephens Lake during the open water period (2001 and 2002) were comparable. Phytoplankton biomass from each area was within the range observed at Long Spruce and Limestone reservoirs, and downstream Nelson River mainstem sites in 1992. Diatoms dominated the community at all sites, despite differences in surface water quality (TSS and turbidity decrease along the flow of the Nelson River in Stephens Lake and downstream, increasing again at the lower end of the Nelson River) and water **residence times** (NSC 2012). Results of chlorophyll *a* analyses (1990–2004) indicate no consistent temporal or spatial differences among the Long Spruce and Limestone reservoirs and the Nelson River mainstem, suggesting that impoundment had little, if any, effect on phytoplankton biomass. Chlorophyll *a* data suggest that the area can be classified as oligotrophic based on trophic classification information presented in Dodds *et al.* (1998). The absence of a marked increase in phytoplankton biomass is likely due to the short water residence time within the Long Spruce and Limestone reservoirs, which, although longer than the unimpounded river, is still too short to allow substantial growth of phytoplankton (NSC 2012).

As was observed in the early 1970s in Stephens Lake, phytoplankton biomass and composition varied considerably between provincial EMP study years (1987 and 1988) and among areas of the lake (Ramsey *et al.* 1989), but no consistent differences were noted between backwater and mainstem sampling locations even though flushing rates on the mainstem locations were greater. Mean chlorophyll *a* concentration measured in 2004 was lower than in the 1970s and 80s in the north arm of Stephens Lake and concentrations were also lower in 2004 in the north arm relative to the mainstem portion of the lake (Appendix 2E). As reported in the Split Lake Cree PPER, the Cree Nation indicated that turbidity, sediment, and algae (*i.e.*, a general increase in phytoplankton biomass) were observed to increase following CRD and flooding associated with the Kettle GS in Stephens Lake (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). Based on predicted nutrient concentrations in the proposed lower Churchill River reservoir reaches (Province of Newfoundland and Labrador), primary productivity in the Churchill River was expected to be lower than the initial post-impoundment potential (0–10 years post-impoundment) (Stockner *et al.* 2001). Productivity below potential in the reservoir system was attributed to the reduced light transmission from the increased suspended sediment load in the reservoir. Given that peak nutrient (particularly phosphorus) and TSS concentrations were projected to occur over a similar time period, the antagonistic interaction of these two parameters on primary productivity and the flushing effect of river flow were expected to moderate water quality issues related to eutrophication in the early periods post-impoundment (0–10 years).

4.2.4.2.3 Downstream of the Keeyask Generating Station

Downstream effects on water quality are not expected to be substantive as the conditions of the reservoir outflow will not be considerably different from current conditions (Section 2.5.2.3). The major exception is a predicted decrease in TSS at the outflow of the GS. Furthermore, TSS is expected to decrease further as water moves through Stephens Lake and this area of reduced TSS would likely extend approximately 10–12 km downstream of the GS. This improvement in water clarity is expected to result in a long-term, small increase in phytoplankton biomass in the affected portion of Stephens Lake (Figure 4-6). The absence of a marked increase in phytoplankton biomass is likely due to the relatively short water residence time within the portion of Stephens Lake along the main flow of the Nelson River, which, although longer than the unimpounded river, is still too short to allow substantial growth of phytoplankton.

4.2.4.2.4 Access Road Stream Crossings

No response is expected. Phytoplankton tend to be relatively unimportant in small streams and other lotic environments as these planktonic organisms cannot maintain positive net growth rates due to a variety of reasons, including downstream losses. The algae in the water column are typically benthic species sloughed from the streambed.

4.2.4.2.5 Net Effects of Operation with Mitigation

Collectively, the above information points to the potential for small to moderate increases in phytoplankton biomass over the long-term in reservoir bays with longer water residence times (*i.e.*, site-specific), depending on the balance between the positive effect of increased nutrients and the negative effect of light depletion. An improvement in water clarity downstream of the GS is expected to result in a long-term, small increase in phytoplankton biomass in the affected portion of Stephens Lake (*i.e.*, local extent).

4.2.4.3 Residual Effects

4.2.4.3.1 Construction Period

No residual effects on phytoplankton are expected.

4.2.4.3.2 Operation Period

There is the potential for small increases of phytoplankton in several areas: the off-current, sheltered bays of the reservoir due to the increase in water residence time and nutrients; the mainstem of the reservoir due to increased water clarity; and the southwestern area of Stephens Lake due to increased water clarity.

4.2.4.3.3 Summary of Residual Effects

The effects of the construction and operation of the Project on phytoplankton are expected to be large over a small geographic extent (at the GS site), small over a medium geographic extent (local area with the reservoir and immediately downstream in Stephens Lake), and long-term. Expected residual effects to the phytoplankton community in terms of biomass were assessed and are presented in Table 4-5A and Table 4-5B for the construction and operation periods, respectively.

The technical phytoplankton assessment is based on models, scientific literature, and information collected from a proxy reservoir (*i.e.*, Stephens Lake) and the overall certainty associated with the predictions is moderate to high. Overall, there is high certainty regarding the nature and direction of effects and the magnitude of effects predicted for the mainstem of the reservoir, and moderate certainty regarding the magnitude of effects in nearshore areas of the reservoir.

4.2.4.4 Environmental Monitoring and Follow-up

As described in Chapter 8 of the Response to EIS Guidelines, Environmental Monitoring Plans have been developed as part of the Environmental Protection Program for the Project. A comprehensive Aquatic Environment Monitoring Program (AEMP) will be developed that specifically outlines monitoring to measure the effects of the Project on the aquatic environment, and discusses how results will be used as a basis for adaptive management. The AEMP will include monitoring of the phytoplankton community to verify the results of the phytoplankton assessment (*e.g.*, to confirm predicted response of biota to sediment inputs during construction).

Phytoplankton community variables are not considered VECs from an environmental assessment perspective; however, as supporting variables for other AEMP components, phytoplankton community variables do provide important measurement endpoints indicating the suitability of waterbodies to support aquatic life, and indicating potential change within or outside the range of natural variability that may be attributed to the operation of the Project.

Monitoring activities for the phytoplankton community may be divided into two major categories: (1) core monitoring (CM); and (2) specific effects monitoring (SEM). The former is aimed at evaluating effects of the operation of the Project throughout the Aquatic Environment Study Area (*i.e.*, over a broad geographical scale) while the latter encompasses a more focussed monitoring component that will be geared towards evaluating effects of the Project in relation to predicted site-specific and/or local effects (*e.g.*, local effects predicted in reservoir bays with longer water residence times). Phytoplankton community monitoring would be conducted annually during instream construction and for the first three years of operation; monitoring would then be conducted every three to five years for the first 20–30 years of operation, depending on results obtained.

Reports detailing the outcomes of monitoring programs will be prepared and submitted to Manitoba Conservation and Water Stewardship (MCWS) and Fisheries and Oceans Canada (DFO; formerly known as the Department of Fisheries and Oceans), in compliance with the *Environment Act* and the *Fisheries Act*, respectively.

4.3 AQUATIC MACROPHYTES AND ATTACHED ALGAE

4.3.1 Introduction

Aquatic macrophytes (plants) grow within the littoral zone. The littoral zone is defined as the area of a lake from the region of the highest seasonal water level to the deepest point at which aquatic plants occur, *i.e.*, where there is sufficient light for photosynthesis to occur. The extent to which the littoral zone can support vascular aquatic plants depends on the availability of bottom sediments sufficiently fine-textured and stable to permit roots to take hold, the degree of wave exposure, and water levels stable enough to minimize disruption of the roots by ice scour during the winter months or desiccation due to exposure (periodic dewatering) during the open water season (Photo 4-2). The clarity of the water column is an important factor in controlling the vertical distribution of submersed plants, while substrate type and exposure to waves and other water currents primarily determines spatial distribution. Emergent plants (*e.g.*, bulrush [*Scirpus* spp.] and cattail [*Typha* spp.]) are found in the uppermost part of the littoral zone, while submersed plants (*e.g.*, coon's tail [*Ceratophyllum demersum*] and common bladderwort [*Utricularia vulgaris*]) may grow at considerable water depth. Lake regulation may affect plant density and distribution, as altering lake levels may influence the availability of light in predominantly wetted areas and substrate availability or stability.

Source: North/South Consultants Inc. [L. Dolce-Blanchard], 2003

Photo 4-2: Aquatic macrophytes growing in the littoral zone of the Aquatic Environment Study Area

Attached algae (non-vascular plants, including macroalgae) generally colonize the surfaces of plants, rocky substrates such as boulder and bedrock shorelines, and open areas of fine sediment (*i.e.*, mud flats). The extent of growth of attached algae depends largely on the stability of the substratum (*e.g.*, shifting sand limits the growth of algae) and on water level fluctuation. Algae will not grow if exposed to air for prolonged periods at low water levels, or if increased water depth reduces light availability below required levels for extended periods. Production by attached algae can be comparable to production by vascular plants due to the rapid growth and turnover time of these microscopic organisms. Algal cells that grow on the surface of plants often provide the basis for a rich community (*i.e.*, biofilm) consisting of the algal cells, detrital particles trapped by the matrix of algal cells, bacteria and fungi digesting the detritus and organic material released by the plant, and microfauna such as protozoa consuming the detritus, decomposers, and algae. This mix of producers, consumers, and decomposers provides nutrition for many kinds of herbivorous and deposit-feeding animals such as snails, certain minnows, and aquatic insect larvae.

4.3.2 Approach and Methods

4.3.2.1 Overview to Approach

The approach taken for the aquatic macrophyte effects assessment was similar to the general approach taken for other aquatic environment components and was comprised of two major steps:

- A description of the existing aquatic habitat conditions to provide the basis for assessing the potential effects of the Project on these components; and
- An effects assessment in which the predicted post-Project environment was described and changes from existing environment quantified.

An ecosystem-based approach was employed to assess the potential impacts of the Project on the aquatic macrophyte community. Information presented incorporates findings from other aquatic environment components (*e.g.*, surface water quality and aquatic habitat). This approach is consistent with the views held by the KCNs, and widely held ecological views, that all components of the aquatic environment are important to maintaining the whole, and that all organisms are interdependent and, therefore, of importance and value.

The environmental setting is described using several sources of information, including: existing published information; and studies conducted specifically as part of the Project between 2001 and 2006. Potential Project-related effects on the aquatic macrophyte community were assessed using basic models (*i.e.*, simple conceptual models, quantitative models based on changes in habitat area, and qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments). These sources of information and effects assessment approaches are described in the following sections.

4.3.2.2 Study Area

The study area for aquatic macrophyte investigations extends along the Nelson River from Split Lake downstream to Stephens Lake in the east (Map 1-2). The magnitude of physical change (*e.g.*, changes in water levels and flows) differs substantially among areas (Section 3.2.2) and, consequently, the aquatic macrophyte study area was divided into three areas on the Nelson River as follows:

- Split Lake area (Split Lake and adjoining waterbodies, including Assean Lake and Clark Lake). This area is upstream of any direct Project influence. The aquatic macrophytes in this area were described to provide supporting information for studies of surface water quality and other aquatic biota (sections 2.0 and 5.0).
- Keeyask area (Nelson River extending from the outlet of Clark Lake to approximately 3 km downstream of Gull Rapids, *i.e.*, hydraulic zone of influence, and tributary streams). Project-related changes to the water regime and direct losses of habitat due to the presence of the GS will occur

within this reach (Section 3.2.2). This area was subdivided at Gull Rapids, as the rapids mark a boundary between the reservoir and downstream environment in the post-Project environment.

- Stephens Lake area (Stephens Lake and adjoining waterbodies). This area is immediately downstream of the Keeyask area and the Project will not affect the water regime. Stephens Lake, as the reservoir of the Kettle GS formed in the early 1970s, provides a useful proxy to assist in predicting effects of the Project (Section 1).

The majority of lower trophic levels investigations were conducted in the Keeyask area, as this area will be directly affected by the Project. Aquatic biota was also described as part of the assessment of the north and south access roads stream crossings.

4.3.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for aquatic macrophytes and attached algae are detailed below.

Few aquatic macrophyte community studies have been previously conducted in the Aquatic Environment Study Area. These data collection programs were primarily focussed on the effects of hydroelectric generating stations (*e.g.*, construction and operation of the Kelsey GS) and were limited to GS reservoirs along the lower Nelson River and Split Lake.

During the early 1990s and early 2000s, limited aquatic macrophyte data were collected for Manitoba Hydro by KCNs Members together with NSC as part of the Lower Nelson River Forebay Monitoring Program (included Long Spruce and Limestone reservoirs, and the lower Nelson River) (Schneider and Baker 1993; NSC 2012).

Occurrences of emergent and submersed aquatic macrophytes in Split Lake were noted and mapped during the bathymetric and aquatic habitat characterization studies conducted in September 1997, by the Tataskweyak Environmental Monitoring Agency (TEMA) for TCN and Manitoba Hydro (Kroeker 1999). The presence and relative abundance of aquatic macrophytes were also noted in conjunction with benthic invertebrate and fish community TEMA studies conducted in 1997 and 1998 (Fazakas 1999; Fazakas and Lawrence 1998; Fazakas and Zrum 1999; Lawrence and Fazakas 1997). The effects of previous hydroelectric development in northern Manitoba were assessed on the Split Lake RMA as part of the Split Lake Cree PPER (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

Information on aquatic macrophyte abundance, species composition, and distribution (*i.e.*, location of areas supporting rooted plants visible from the surface) was recorded during the aquatic habitat surveys undertaken in 2001, 2003, and 2006 in the study area (Section 3.2.3.2; Appendix 3A). Detailed sampling to describe aquatic plant abundance and composition at selected sites was conducted in 2001 and 2002 between Birthday and Gull rapids, in 2003 and 2004 for Clark Lake to Gull Rapids and in 2005 and 2006 (years with higher than average water levels; Section 3.3.2.3) in two areas (Ross Wright and O'Neil bays) of Stephens Lake that were historically inundated; these two areas in Stephens Lake were chosen to provide a proxy for the post-impoundment Keeyask reservoir.

Drifting aquatic plants and algae were sampled using drift traps at various locations along the Nelson River mainstem during the 2003 and 2004 open water seasons to provide the basis for assessing potential changes in the production of these groups from specific areas (*i.e.*, Birthday Rapids, Gull Rapids, Stephens Lake) associated with the proposed Keeyask hydroelectric development. Attached algae (periphyton) likely colonize extensive areas of faster water habitat in the study area; however, the distribution of periphyton in relation to aquatic habitat was not quantitatively assessed, as large sections of the study area cannot be safely accessed under most flow conditions, precluding adequate baseline sampling. As periphyton is highly variable, both spatially and temporally, an intensive, repeatable program would be required to obtain sufficient data to support an analysis of effects on the distribution of attached algae in relation to aquatic habitat.

The detailed approach and methods for aquatic macrophyte community studies conducted between 2001 and 2006 are presented in Appendix 4A.

4.3.2.4 Assessment Approach

Given the complexity of the aquatic ecosystem, models were used for predicting effects of the Project. Within the aquatic assessment, the complexity of models employed depended on: the importance of the issue; availability of information or suitable models; and utility of modelling approaches.

Basic model types used to assess potential Project effects on the aquatic macrophyte community were:

- Simple conceptual models (*e.g.*, plant species may proliferate in the slower flow rates and wider littoral zones associated with typical reservoir habitat). The scientific literature was used to describe and support linkages to the Project.
- Quantitative models based on changes in aquatic habitat area (*e.g.*, calculation of occupied aquatic macrophyte habitats based on specific areas of habitat types that had been described in the existing environment) over the short-term and long-term post-Project.
- Qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments. For example, Stephens Lake was used as a surrogate for post-Project conditions in the Keeyask reservoir (Section 3.2.4).

A quantitative habitat-based model was used to estimate the area occupied by aquatic macrophytes in the newly created Keeyask reservoir at four time steps (Years 1, 5, 15, and 30 post-impoundment). The area of each habitat type with aquatic plant beds was estimated for the Nelson River between the outflow of Clark Lake and the Keeyask GS location in the existing environment using Manitoba Hydro's shoreline data (the spatial extent of habitat types was modeled at 95th percentile flow conditions) and in Year 30 post-Project using the predicted shoreline at a water level elevation at the face of the dam of 158 m ASL for minimum operating level (MOL) or at 159 m ASL under 95th percentile flow conditions for FSL (Section 3.2.4; Appendix 3D). The Year 30 habitat areas were modified for the intermediate time steps to account for shoreline erosion, peat disintegration and transport, and loss and subsequent establishment of aquatic plant beds (Appendix 3D).

The evaluation of certainty for predicted effects was based in part on the agreement of predicted effects among the various approaches.

4.3.3 Environmental Setting

4.3.3.1 Overview and Regional Context

The environmental setting has been described based on available background data and the information collected during the course of the Keeyask environmental studies. The aquatic macrophyte community in the study area has been influenced by past hydroelectric development in northern Manitoba (*e.g.*, Kelsey GS, CRD, and LWR).

Thirty-four vascular and five non-vascular taxa of aquatic macrophytes have been recorded in the study area (Table 4-6). No species are listed on Schedule 1 of SARA and none have been assessed as "at risk" by COSEWIC. The Manitoba Conservation Data Centre lists *Nymphaea tetragona* as an S2 species (rare in province, maybe vulnerable to extirpation) in the Churchill River Upland ecoregion, but its distribution does not extend into the area directly affected by the Project (Manitoba Conservation Data Centre 2012a; Manitoba Conservation Data Centre 2012b). None of the identified species are listed as invasive on the Invasive Species Council of Manitoba website (Invasive Species Council of Manitoba 2012).

In lacustrine environments (*e.g.*, Split, Gull, and Stephens lakes), the occurrence of aquatic plants was generally restricted to areas shallower than 2 m water depth, although some plants were observed in water depths up to 3 m. Distribution tended to be patchy with localized macrophyte beds ranging from very sparse to dense. Aquatic plants were most common in nearshore (*i.e.*, shallow water depths) sheltered bays and channels between islands characterized as having standing water and soft, mineral-based bottom sediments. Pondweeds (*Potamogeton* spp., *Stukenia* spp.) were typically the most common plants observed. Vertical zonation was typical in some areas with *Stukenia* spp. occurring in deeper water and *P. richardsonii* generally restricted to shallower water depths. The shallow zone was shared by other macrophytes including, common spikerush (*Eleocharis palustris*) and vernal water-starwort (*Callitriche palustris*), two aquatic plant species that are more resistant/tolerant to periodic dewatering/desiccation (*i.e.*, amphibious) and ice scour stress, and the less tolerant northern watermilfoil (*Myriophyllum sibiricum*). The diversity of the plant community was comparable among lacustrine habitats sampled with as many as 10 species observed in a waterbody.

In the Nelson River mainstem, aquatic plants were restricted to the shallow margins of tributary mouths, or adjacent areas within the mainstem, and to a few small, sheltered bays within the mainstem. The distribution of macrophytes varied among years depending on annual conditions (predominantly water level); however, plant distribution was always limited and density was typically sparse. The composition of the plant community reflected what was observed in other areas, with the exception of white water-crowfoot (*Ranunculus aquatilis*), which was only observed downstream of Gull Rapids. The greatest proportions of aquatic macrophytes (*i.e.*, greatest percentages of macrophytes observed in the study area) were typically within the more extensive backwater inlets with reduced water velocities and relatively shallow water depths. As for the mainstem portion of the river, the distribution of macrophytes varied among years depending on annual conditions (predominantly water level). Aquatic vegetation was observed exclusively within shallow water areas characterized as having standing water or low water velocity and soft, mineral-based bottom sediments. Plant growth ranged from low (10–30 grams [g] dry weight of plants/square metre [m^2] of bottom area) to high (greater than 60 g/m^2) relative density with Pahwaybanic, Kahpowinic, and John Garson bays typically supporting the most relatively dense communities. Star duckweed (*Lemna trisulca*), pondweeds, common spikerush, and northern watermilfoil were the most common plants observed, with northern watermilfoil, *Potamogeton* spp. and *Stukenia* spp. being the most abundant. Aquatic plants were most abundant in the intermittently exposed zone of the backwater inlets sampled. The intermittently exposed zone was defined using historic water level percentiles. This area is the shore zone bounded by the 5th and 95th water level percentiles and represents a band along the edge of the waterbody that has experienced exposure, *i.e.*, dewatering, 5 to 95% of the time since 1977 (Section 3.2.4.1). The community in this type of habitat was dominated by northern watermilfoil and was shared with other species including the more amphibious common spikerush. In shallow water habitat that was predominantly wetted, the community was dominated by pondweeds, particularly *Potamogeton* spp. Filamentous green algae, Cyanophycota (blue-green algae), and aquatic moss (*Sphagnum* spp.) were also commonly observed in the backwater inlets.

Several aquatic macrophyte species more tolerant of exposure or periodic episodes of dewatering and ice scour stress, such as vernal water-starwort and common mare's-tail (*Hippuris vulgaris*), were found in the study area. Notably, common spikerush, common mare's-tail, and lady's thumb (*Polygonum persicaria*) were all observed in the Keeyask area. Common spikerush develop a thick root mass that is resistant to compaction and erosion. These species were sometimes found in association with relatively less resilient aquatic plants, such as bur-reed (*Sparganium* spp.) and common bladderwort (*Utricularia vulgaris*), which can grow in shallow water but are intolerant of prolonged periods of exposure.

In riverine environments of the study area, the flowing water transports relatively large amounts of aquatic plant and algal biomass downstream. Drifting aquatic plants and algae were sampled to gain an overall understanding of the spatial and temporal differences in abundance and distribution of biomass within the study area, and provide the basis for assessing potential changes in production from Birthday and Gull rapids, and Stephens Lake associated with the proposed Keeyask hydroelectric development. A minimum of 22 plant species, including unidentified algae (all macrophytes were grouped and identified to genus and species, when possible), were collected in drift traps set in the Nelson River mainstem between the upstream extent of Birthday Rapids and downstream of the Kettle GS in 2003 and 2004. Non-vascular plants (specifically algae, mainly filamentous) were the most abundant vegetation collected in drift traps. Grasses (Poaceae), sedges (Cyperaceae), rushes (Juncaceae), and pondweeds were also relatively abundant. Overall, the drift traps downstream of Birthday Rapids were the most productive in terms of drifting plant (particularly non-vascular) biomass within the study area, followed by traps upstream of Gull Rapids (at the downstream end of Gull Lake), downstream of Gull Rapids, upstream of Birthday Rapids, and downstream of the Kettle GS. From this, it may be inferred that the majority of drifting plant biomass in the study area was produced by the Nelson River aquatic habitats within Birthday Rapids, between Birthday and Gull rapids, including Gull Lake, and within Gull Rapids. The majority of drifting plant biomass was collected later in the growing season (August and September).

The aquatic macrophyte community has been described as part of the Wuskwatim GS EIS for portions along the Burntwood River system, using methods comparable to those employed in the study area

(Manitoba Hydro and NCN 2003). Direct comparisons of aquatic plant community abundance, composition, and distribution between different watercourses is challenging as the distribution and density of aquatic macrophytes differs over time (particularly among years) in response to fluctuations in water levels and other growing conditions.

There were 24 vascular and three non-vascular aquatic plants collected and identified in the lacustrine and riverine areas of the Wuskwatim area. The aquatic macrophyte community observed in the Nelson River study area was more diverse with 34 vascular and five non-vascular aquatic macrophytes observed. Sampling of the aquatic plant community in the Nelson River was more intensive over a longer study period and this could partially account for the increased diversity observed. Sixteen vascular macrophytes were common to both areas and the pondweeds (*Potamogeton* spp., *Stukenia* spp.) were the most diverse group with seven species observed in the Wuskwatim areas and eight in the study area. Pondweeds tended to dominate the aquatic vascular plant community in both areas.

The occurrence of aquatic plants was similar between the Wuskwatim and Keeyask study areas. In lacustrine environments, the occurrence of aquatic plants was typically restricted to shallower water depths, and plants were most common in sheltered bays and channels between islands characterized as having minimal water movement and soft, mineral-based bottom sediments. In the mainstem portion of rivers, aquatic plants tended to be restricted to the shallow margins of tributary mouths or adjacent areas within the mainstem, and to the few small, sheltered bays within the mainstem; more extensive areas of aquatic plants at relatively high densities typically occurred in the few extensive backwater inlets along the Burntwood and Nelson rivers. Several aquatic macrophyte species more tolerant of periodic episodes of dewatering and ice scour stress, such as vernal water-starwort and common mare's-tail, were found in both the Keeyask and Wuskwatim study areas.

4.3.3.2 Split Lake Area

As part of the PPER process, TCN Members reported that the abundance of rooted aquatic plants has diminished due to ice build-up on shorelines because of past hydroelectric development in the Split Lake RMA (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

Emergent sedges and grasses were widely distributed but never abundant, being generally restricted to narrow shoreline bands within sheltered bays. Submersed aquatic macrophytes were also widely distributed throughout Split Lake. Aquatic plants were generally restricted to areas shallower than 2 m, although some plants were observed in depths up to 2.5–3.0 m. Within this depth range, distribution was notably patchy with localized macrophyte beds ranging from very sparse to dense. Macrophyte beds were most common in sheltered bays and channels between islands, especially in off-current areas (Map 3-4). Macrophyte growth was probably limited primarily by turbidity (restricted euphotic depth) and secondarily by exposure to waves and currents.

Pondweeds were the most common plants observed, with big-sheath pondweed (*Stukenia vaginata*), whitestem pondweed (*Potamogeton praelongus*), variableleaf pondweed (*P. gramineus*), and Richardson's pondweed (*P. richardsonii*) being the most abundant. Vertical zonation was typical for these species, with bigsheath pondweed occurring in deeper water (up to 3 m), whitestem pondweed and variableleaf pondweed dominating intermediate depths, and Richardson's pondweed generally restricted to shallower water. The shallow zone was also shared with other macrophytes including northern watermilfoil, water smartweed (*Polygonum amphibium*), and fineleaf pondweed (*S. filiformis*).

Quantitative surveys of aquatic macrophytes were undertaken in Clark Lake in 2001, 2003 and 2004 as part of the environmental studies (Photo 4-3). In 2001, aquatic vegetation occupied an area of only 5.8 hectares (ha; 0.5 % of the total area of Clark Lake) (Table 4-7) exclusively within shallow water areas (off-current bays) characterized by standing water and soft, mineral-based bottom sediments. Plant growth was typically of sparse to low density and the aquatic plant community consisted of five species in 2003 (Table 4-8) and 10 species in 2004 (Table 4-9). Macrophyte growth was likely limited primarily by turbidity (limited euphotic depth) and secondarily by exposure to waves and currents.

Source: North/South Consultants Inc. [L. Dolce-Blanchard], 2003

Photo 4-3: Aquatic macrophyte sampling location in Clark Lake, 2003

As in Split Lake, pondweeds were the most common plants observed in Clark Lake, with Richardson's pondweed being the most abundant. Vertical zonation was typical with big-sheath pondweed occurring in deeper water (up to 2 m) and Richardson's pondweed generally restricted to shallower water depths. The shallow zone was also shared by other macrophytes including, common spikerush and vernal water-starwort, two species that are more tolerant of periodic dewatering and ice scour stress, and the less resilient northern watermilfoil.

4.3.3.3 Keeyask Area

No data or assessment of the effects of hydroelectric development on the aquatic plant community prior to 1997 in the reach of the Nelson River between Clark Lake and Stephens Lake were located in the published literature.

Quantitative surveys of aquatic macrophytes in the mainstem of the Nelson River were undertaken in 2001, 2003, and 2006 as part of the environmental studies. The distribution of aquatic plants varied among years depending on annual conditions (predominantly water level); however, plant distribution was limited and density was sparse. Upstream of Gull Lake, the maximum area occupied by plants was 8.2 ha (1.1 % of the total area immediately upstream of Gull Lake) in 2006 (Table 4-10), whereas the maximum observed in the area immediately downstream of Gull Rapids was only 0.9 ha in 2001 (Table 4-7). Aquatic vegetation was restricted to the shallow margins of tributary mouths, or adjacent areas within the mainstem, and to a few small, sheltered bays within the mainstem (Map 3-16). Macrophyte growth and distribution were likely limited primarily by turbidity (limited euphotic depth) and secondarily by exposure to waves and currents.

Aquatic plant abundance was not assessed in the mainstem of the Nelson River. However, the presence of aquatic macrophyte species was noted and the community upstream of Gull Rapids consisted of sedges (*Carex* spp.), northern watermilfoil, pondweeds, pond lilies (*Nuphar* spp.), water smartweed, and the more amphibious common spikerush and lady's thumb, whereas the community downstream of Gull Rapids was limited to northern watermilfoil, pondweeds, water smartweed, white water-crowfoot, and bur-reed.

There are a few extensive areas within the backwater inlets along the Nelson River and in Gull Lake with relatively shallow water depths and reduced water velocity (Map 3-7 and Map 3-8). The streams enter bays off the mainstem and are inundated with water from the Nelson River. Typically, the bottom sediments in these areas were soft, mineral-based with variable amounts of decaying plant material present where aquatic plants were observed (Map 3-14).

The aquatic macrophyte community in select backwater inlets was quantitatively assessed between 2001 and 2004, and again in 2006. More intensive sampling occurred in Pahwaybanic Bay, Kahpowinic Bay, the small bay east of Rabbit Creek, John Garson Bay, Tub Bay, and John Kitch Bay (between Morris Point and John Kitchekeesik Point) (Map 3-16). Although the areas in the backwater inlets occupied by aquatic vegetation varied among years, the greatest proportions of aquatic macrophytes (*i.e.*, greatest percentage of macrophytes observed) were typically within these aquatic habitats. Among backwater inlets in 2001, the greatest proportion of aquatic macrophytes was observed in John Garson Bay (15.8% of total macrophytes observed) occupying 59.3 ha (Table 4-7). The area occupied by plants in John Garson Bay was comparable in 2003 to 2001, however, the area with macrophytes in John Kitch Bay was notably greater and totalled 100.6 ha (35.6 % of total) (Table 4-11). In 2006, plant distribution in these two backwater inlets was noticeably reduced, however, distribution in Kahpowinic Bay had increased and macrophytes occupied 42.3 ha (26.3 % of total macrophytes observed). Aquatic vegetation was observed exclusively within shallow water areas characterized as having either standing water or low water velocity and soft, mineral-based bottom sediments (Map 3-16). The distribution (or shape) of plant beds is strongly influenced by water level over time. In general, the location of beds tended to move 'land-ward' (*i.e.*, shift to a higher elevation) under higher water conditions (2006; Section 3.3.2.3) and 'water-ward' (*i.e.*, shift to a lower elevation) when water levels had been lower than average (2003 conditions; Section 3.3.2.3), *i.e.*, the observed variability in plant bed distribution was primarily due to variation in water level (Map 3-16). Plant growth ranged from low to high relative density with Pahwaybanic, Kahpowinic, and John Garson bays typically supporting the densest communities (Table 4-8, Table 4-9,

Table 4-12, and Table 4-13). The least diverse plant community with only three species occurred in John Garson Bay in 2002 (Table 4-13), whereas Pahwaybanic and Kahpowinic bays supported the greatest variety of aquatic plant species observed, with nine species each in 2004 (Table 4-9).

Star duckweed, pondweeds, common spikerush, and northern watermilfoil were the most common plants observed, with northern watermilfoil, *Potamogeton* spp. and *Stukenia* spp. being the most abundant. Aquatic plants were most abundant in the intermittently exposed zone of the backwater inlets sampled. The community in this type of habitat was dominated by northern watermilfoil (Photo 4-4) and was shared with other species including the more amphibious common spikerush. In shallow water habitat that was predominantly wetted, the community was dominated by pondweeds, particularly *Potamogeton* spp. (Photo 4-5). Common spike rush and star duckweed were much more common in shallow water habitat with standing water than in those classified as having low water velocity. Filamentous green algae, blue-green algae, and aquatic moss species were also observed in the backwater inlets and comprised up to 12%, 11%, and 15%, respectively, of the community sampled.

Source: North/South Consultants Inc. [L. Dolce-Blanchard], 2005

Photo 4-4: ***Myriophyllum sibiricum* in the Keeyask area**

Source: North/South Consultants Inc. [L. Dolce-Blanchard], 2005

Photo 4-5: *Potamogeton* spp. in the Keeyask area

Quantitative surveys of the aquatic macrophyte community in Gull Lake were performed between 2001 and 2004, and again in 2006 as part of the environmental studies. The area in Gull Lake occupied by aquatic vegetation varied among years and ranged between 238.5 ha (7% of the total area of Gull Lake) in 2001, a year with median water elevation (Table 4-7) (Section 3.3.2.3) and 58.8 ha (2%) in 2006, a high-water year (Table 4-10) (Section 3.3.2.3). As in the backwater inlets, aquatic vegetation was observed exclusively within shallow water areas characterized as having either standing water or low water velocity and soft, mineral-based bottom sediments (Map 3-16). Similar to the backwater inlets, the distribution or shape of plant beds was strongly influenced by water level over time. In general, the location of beds tend to move 'land-ward' (*i.e.*, shift to a higher elevation) under higher water conditions (2006; Section 3.3.2.3) and 'water-ward' (*i.e.*, shift to a lower elevation) when water levels had been lower than average (2003 conditions; Section 3.3.2.3) (Map 3-16). Plant growth north of Caribou Island ranged from low to high relative density and the aquatic plant community consisted of between two species observed in 2002 (Table 4-13) and six in 2003 (Table 4-8).

Pondweeds were the most common plants observed, with big-sheath pondweed and Richardson's pondweed being the most abundant. Vertical zonation did not appear to be present with species occurring together in water up to 1.5 m deep. The shallow nearshore was also occupied by less common macrophytes including northern watermilfoil, and the more amphibious common spikerush, common mare's-tail, and lady's thumb. Filamentous green algae and aquatic moss species were also commonly observed in this area and comprised as much as 27% and 30%, respectively, of the community sampled.

Aquatic moss tended to be observed in the very shallow zone (water depths less than 0.4 m) with filamentous green algae more common in water depths of up to 1.5 m.

In riverine environments of the study area, the flowing water transports relatively large amounts of aquatic plant and algal biomass downstream. Drifting aquatic plants and algae were sampled to gain an overall understanding of the spatial and temporal differences in abundance and distribution of biomass within the study area, and provide the basis for assessing potential changes in production from Birthday and Gull rapids associated with the proposed Keeyask hydroelectric development.

A minimum of 22 macrophyte species, including unidentified algae, were collected in drift traps set in the Nelson River mainstem between the upstream extent of Birthday Rapids and the downstream extent of Gull Rapids in 2003 and 2004. Non-vascular plants (specifically algae, mainly filamentous) were the most abundant vegetation collected in drift traps located upstream of Birthday Rapids, downstream of Birthday Rapids, and downstream of Gull Rapids in terms of dried weight (milligrams [mg] dried weight/100 m^3) (Table 4-14). Vascular plants (specifically, grasses, sedges, rushes, and pondweeds) tended to be relatively more abundant in drift traps located upstream of Gull Rapids (Table 4-14).

Overall, the drift traps downstream of Birthday Rapids were the most productive in terms of drifting plant biomass (56 mg dried weight/100 m^3) within the study area, followed by traps upstream of Gull Rapids (at the downstream end of Gull Lake) (35 mg dried weight/100 m^3), downstream of Gull Rapids (26 mg dried weight/100 m^3), and upstream of Birthday Rapids (10 mg dried weight/100 m^3) (Table 4-14). From this, it may be inferred that the majority of drifting plant biomass in the study area was produced by the Nelson River aquatic habitats within Birthday Rapids, between Birthday and Gull rapids, including Gull Lake, and within Gull Rapids. The majority of drifting plant biomass was collected later in the growing season (August and September).

4.3.3.4 Stephens Lake Area

Elders and resource harvesters from TCN have commented that there has been a large decrease in plant life (including areas of very large reeds) in Stephens Lake that they feel is due in part to deeper, more turbid water, which does not allow the sun to reach the plants. They noted that the loss of plants appears more prevalent in the last ten years (changes started in the mid-1980s), which they related to an observed decrease in water quality. The resource harvesters also remarked on the colour of the algae attached to rocks along the shoreline, which they say has changed from green to a 'brownish colour'. These changes were linked to hydroelectric development in general and not to one specific project (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

Quantitative surveys of aquatic macrophyte abundance and composition were undertaken in 2005 and 2006 (years with higher than average water levels; Section 3.3.2.4) in two areas (Ross Wright and O'Neil bays) of Stephens Lake that were historically inundated. These two areas were chosen to provide a proxy for the post-impoundment Keeyask reservoir and form the basis for the following discussion (Cooley and Dolce 2008).

Aquatic vegetation was observed exclusively within shallow water areas characterized as having standing water and soft, mineral- or organic-based bottom sediments. Plant growth in these areas ranged from medium to high relative density and the aquatic plant community was relatively diverse consisting of nine taxa in 2005 and eight in 2006 (Table 4-15). Macrophyte growth was generally limited by turbidity (limited euphotic depth) and exposure to waves and currents.

Pondweeds, northern watermilfoil, and water smartweed were the most common plants observed, with Richardson's pondweed and northern watermilfoil being the most abundant and relatively dense. Aquatic plants were most abundant in the shallow water areas sampled with soft, mineral-based sediments. The intermittently exposed portion of this habitat type was dominated by Richardson's pondweed; however, northern watermilfoil was also relatively common and *Stukenia* spp. and aquatic moss species were also found. The community sampled in the predominantly wetted portion was almost exclusively Richardson's pondweed. In shallow water habitat with soft, organic-based sediments, most typically observed at the terminal ends of the inundated bays (*e.g.*, Ross Wright Bay), the community composition shifted and northern watermilfoil was predominant, with water smartweed also being abundant in localized areas. Aquatic plant species that are more tolerant of periodic dewatering and ice scour stress were not observed in the areas sampled.

A minimum of eight aquatic macrophyte species, including unidentified algae, were collected in drift traps set downstream of the Kettle GS (*i.e.*, downstream of Stephens Lake) in 2003 and 2004. As at the majority of drift trap locations in the Keeyask area, non-vascular plants (specifically algae, mainly filamentous) were the most abundant vegetation collected in terms of dried weight (mg dried weight/100 m^3) (Table 4-14). Overall, the least amount of drifting plant biomass in the study area was collected in drift traps located downstream of the Kettle GS (5 mg dried weight/100 m^3) (Table 4-14). The relatively low drifting plant biomass downstream of the Kettle GS may be the result of: drift traps being located approximately 1 km downstream of the GS structure, thereby potentially sampling the drifting plants predominantly originating from this relatively short section of the river only, rather than also from Stephens Lake; and/or the majority of drift traps being located in areas with relatively slower water velocities in comparison to those located in the Keeyask area. However, the lack of drifting plant biomass information from immediately upstream of the Kettle GS in Stephens Lake makes it difficult to determine whether this low downstream biomass is contributed to by sampling location and/or a paucity of plant biomass originating from Stephens Lake and drifting through the Kettle GS. Similar to other sites along the Nelson River mainstem, the majority of plant biomass was collected later in the growing season (August and September).

4.3.3.5 Current Trends

Historic information for all areas is lacking to compare to data collected during the Keeyask environmental studies to describe current trends for aquatic macrophytes and attached algae.

4.3.4 Project Effects, Mitigation and Monitoring

4.3.4.1 Construction Period

The following section considers potential effects related to the construction of the GS and south access road, and operation of the construction camp and north and south access roads during the construction period. The construction of the north access road was assessed in the KIP EA (Keeyask Hydropower Partnership Ltd. 2009). Stream crossing locations are provided in Map 1-4.

An assessment of potential Project effects on the aquatic macrophyte community during the construction period is based on the assessment of construction-related effects to surface water quality (Section 2.5.1, Table 2-12) and physical attributes of aquatic habitat (Section 3.4.1). The primary potential effect(s) on aquatic macrophytes is related to inputs affecting water quality, such as increases in TSS concentrations and related variables (*i.e.*, turbidity) due to instream activities (*e.g.*, cofferdam placement and removal, river impoundment and diversion) and nutrient inputs (*e.g.*, with treated sewage effluent discharge to the mainstem of the Nelson River, with particulate materials [*i.e.*, TSS]). Predicted increases in TSS will alter downstream substrate due to sedimentation, and this could influence any aquatic macrophyte beds in affected areas. Cofferdam placement and dewatering of the area within cofferdams would affect any attached algae in the immediate vicinity of any works (the majority of aquatic habitat affected during construction will also affected by the permanent works; some construction works will remain in place and be submerged during impoundment). Additionally, some aquatic habitat disruption will occur during construction of stream crossings to accommodate the south access road and may affect aquatic plant cover at the crossings. It is expected that construction effects (*e.g.*, inputs affecting water quality) will be managed through appropriate mitigation measures (Section 2.5.1; Section 3.4.1), thereby reducing the duration and magnitude of any construction-related effects on the aquatic macrophyte community.

4.3.4.1.1 Upstream of the Outlet of Clark Lake

No construction-related effects on the aquatic macrophyte community are expected upstream of the outlet of Clark Lake as there are no linkages between Project construction and surface water quality (Section 2.5.1) or aquatic habitat (Section 3.4.1) in Split, Assean, or Clark lakes.

4.3.4.1.2 Downstream of the Outlet of Clark Lake

The following sub-sections present the assessment of potential effects of construction activities on the aquatic macrophyte community in the Keeyask area and downstream.

Changes to Water Quality

Total Suspended Solids, Turbidity, and Water Clarity

Overall, the activities with the greatest potential to increase TSS concentrations in the lower Nelson River during construction of the GS are related to cofferdam placement and removal, and river impoundment and diversion (Section 2.5.1.1). Effects of suspended fine sediments on aquatic macrophytes are likely primarily related to their effect on light penetration; light attenuation by inorganic turbidity decreases the fraction of light absorbed by photosynthesizing plants. Generally, the construction and removal of cofferdams will generate an increase of less than 5 mg/L of TSS above background downstream of Gull Rapids (Section 2.5.1). Larger TSS increases are expected to be of relatively small magnitude and short duration. Peak levels are predicted to be up to 15 mg/L for one day or up to 7 mg/L for one month (Section 2.5.1). Drainage of surface runoff to the Nelson River will be controlled through a Drainage Management Plan (as described in the PD) to minimize the amount of sediment produced and the potential for sediment to enter watercourses. If the TSS concentration in water pumped out of cofferdam and excavation areas and in concrete wash water is greater than 25 mg/L the water will remain in a settling pond until it meets this TSS criterion before being discharged to the Nelson River. As the magnitude and duration of any increases in TSS are typically within the 30-day MWQSOG for PAL (an increase of 5 mg/L above background where background is less than or equal to 25 mg/L), aquatic plants and attached algae may be somewhat negatively affected in this downstream environment as photosynthetic efficiency may be reduced, thereby somewhat limiting primary production (*i.e.*, small, undetectable reductions in biomass may occur in affected areas during the construction period).

Nutrients

Nutrient (nitrogen and phosphorus) inputs with treated sewage effluent discharge to the mainstem of the Nelson River are not expected to be detectable in the fully mixed river condition, but concentrations may be elevated near the effluent outfall (Section 2.5.1.3.3). Additionally, any increases in nutrients associated with expected increases in particulate materials (*i.e.*, TSS) are expected to be small. As the expected level of increase in nutrient inputs to the Nelson River during the construction period is small, nutrient inputs will not have a measurable effect on aquatic macrophytes beyond the immediate receiving environment. During the latter stage of the Stage II Diversion, when water levels are increased to near FSL, flooding of organic materials is expected to lead to nutrient release to surface waters, thereby increasing concentrations of nutrients, notably over flooded habitat. These effects (*i.e.*, due to reservoir impoundment) are discussed in detail in the assessment of operation-related effects in Section 2.5.2.2 for surface water quality and Section 4.3.4.2 for aquatic macrophytes.

Metals and Contaminants (*e.g.*, Hydrocarbons)

Small amounts of metals will be introduced into the aquatic environment in association with construction activities that release sediments, as discussed in Section 2.5.1.6. However, given the proposed mitigation measures to manage sediment levels, these inputs are not expected to cause marked increases in metal levels and, consequently, will have no detectable effect on the aquatic macrophyte community.

The presence and levels of hydrocarbons in the aquatic environment could potentially be affected by accidental spills or release of substances containing hydrocarbons (*e.g.*, diesel fuel, gasoline, lubricating oil, *etc.*). Other hazardous substances will also be used during the construction period. As described in Section 2.5.1.6, the release of significant quantities of hazardous substances to the aquatic environment as a result of accidental spills and releases is considered unlikely due to the development and implementation of good management practices.

Alteration and Destruction of Aquatic Habitat

Downstream Sedimentation

Under current conditions, natural sedimentation occurs in areas of reduced water velocity (*e.g.*, sheltered areas behind individual cobbles and boulders, the water within aquatic macrophyte beds). Aquatic macrophyte beds typically enhance the deposition and accumulation of fine sediments and act as sieves (*i.e.*, beds trap suspended sediment particles). It is predicted that approximately 30 % of the additional sediment resulting from shore erosion during Stage I and II Diversions will be deposited in Stephens

Lake before it reaches the Kettle GS (Section 2.5.1.1.3); most of the deposition is expected to occur near the entrance of Stephens Lake, downstream of Gull Rapids (Section 3.4.1.5). This additional sedimentation could negatively influence any aquatic macrophytes (vascular and non-vascular) in the affected area depending on the size of sediment particles, the spatial extent (*e.g.*, greater negative potential if an entire plant bed is affected) and depth (*e.g.*, greater negative potential if depth of sediments exceeds 5 cm) of deposited sediments, the rate of deposition, and if deposited sediments are stable or transient (*e.g.*, washed away with the next higher flow event). Cumulative sediment input from all construction sources, over a four-year period for instream work, is expected to result in a depth of deposited sediments less than 0.6 cm (very low rate of deposition) through the south arm of Stephens Lake. Deposited material will likely be a combination of silt, sand, and coarser material, and is unlikely to be remobilized during the GS operating period. The sensitivity of aquatic plants to sedimentation is species-specific and some are more tolerant as they are able to respond by adjusting their rooting levels if sedimentation is not sufficiently rapid or of sufficient depth to bury plant stands. However, based on the low rate of deposition and resultant minimal depth of deposited sediments over the four years of instream work, downstream sedimentation is not expected to have a measurable effect on aquatic macrophyte beds during the construction period.

Loss of Aquatic Habitat in Footprint of Supporting Infrastructure

The construction of cofferdams will result in the temporary loss of aquatic habitat in Gull Rapids (Section 3.4.1.1). Areas of filamentous algae may be negatively affected by the loss of aquatic habitat due to either cofferdam footprint or dewatered area; additionally, starting during construction, there would be a site-specific decrease in the production of drifting filamentous algae from Gull Rapids.

4.3.4.1.3 South Access Road Stream Crossings

No response is expected due to the input of sediments into natural watercourses as effects to surface water quality are predicted to be small due to the application of various mitigation measures (Section 2.5.1.7).

At each of the three stream crossings, the footprint of the road, combined with the installation of the culvert(s), will result in several changes in aquatic habitat (Section 3.4.1.6). Aquatic plants that occur at proposed south access road stream crossings will be lost due to infilling of a relatively small amount of aquatic habitat at crossings. Potential effects to aquatic plants at the stream crossings will be addressed by following procedures described in the "Manitoba Stream Crossing Guidelines for Protection of Fish and Fish Habitat" and other pertinent regulatory guidelines.

4.3.4.1.4 Net Effects of Construction with Mitigation

The construction of cofferdams will result in a moderate reduction in the production of drifting, non-vascular plant (filamentous algae) biomass originating from Gull Rapids. The decrease in filamentous algae is expected to be long-term due to effects continuing through the operation period.

Access road stream crossings will result in the permanent loss of aquatic plants in the immediate footprint of the access road and culvert(s).

4.3.4.2 Operation Period

Water flow-related aspects (*e.g.*, flow regime and extremes, substrate composition and stability) typically determine the distribution and abundance of aquatic macrophyte communities. The extent to which reservoir development may affect aquatic macrophytes depends largely on existing water flow patterns, reservoir design, and the post-impoundment flow regime. Regulation of water flows affects the ability of macrophytes to attach to the substrate (Bunn and Arthington 2002), with successful colonization influenced by water level extremes (flooding, desiccation), localized variations in water velocity (turbulence, shear stress), timing, hydraulics, and substrate composition and stability (Nalcor Energy 2009).

4.3.4.2.1 Upstream of the Outlet of Clark Lake

No response is expected. Selection of a 159 m ASL reservoir elevation instead of a higher elevation will avoid Project-related effects as Split Lake area is beyond the upstream extent of the expected hydraulic zone of influence.

4.3.4.2.2 Outlet of Clark Lake to the Keeyask Generating Station

Potential Project Effects and Proposed Mitigation

Operation-related pathways that were assessed for potential effects to aquatic macrophyte distribution and abundance included: flooding (loss of existing habitats, creation of new habitats); conversion of existing hard substrates to silt/clay due to sedimentation in Gull Lake; increase in the frequency of water level fluctuations; and a reduction in the extent and severity of ice scour. Summaries of predicted responses of aquatic macrophytes to changes resulting from the operation of the Project are presented in Figure 4-5. Where feasible, the effects of these pathways were considered using modelling exercises (quantification of potential effects), empirical information from Stephens Lake and other reservoirs in northern Manitoba, reservoirs in other northern temperate areas, and the scientific literature.

Assessment of Operation-Related Effects

Modeling Approach

Post-impoundment, the vast majority of existing aquatic vascular plant beds will be lost due to an increase in water levels (*i.e.*, flooding) (Section 3.4.2.2; Table 4-16). A very small area with plants (approximately 0.1 to 2 ha, primarily in shallow, standing water habitat with soft, mineral-based substrates, depending on the GS mode of operation) will remain in the more riverine, upstream portion of the reservoir (reaches 2A-4). Aquatic plant beds are expected to begin to develop in the downstream portion of the Keeyask reservoir (reaches 5-9A) between 5 and 15 years after impoundment (Section 3.4.2.2; Table 4-16). New vascular plant beds will likely develop in shallow flooded areas (*i.e.*, areas with standing water and soft, organic-based substrates) and other shallow areas that are characterized as having standing water and soft, mineral-based substrates, but the spatial extent and type of vegetation will depend on the interaction of effects related to bottom type, wave action, bottom slope, water quality (turbidity/light penetration), and ice scour (Map 3-35). Aquatic plants may experience somewhat improved growing conditions due to a reduction in ice scour stress (*i.e.*, potential increase in the amount of habitat suitable for attached plants) (Section 3.4.2.2). However, some evidence indicates that aquatic plants respond favourably to ice break-up disturbance, particularly after years of less severe break-up (Prowse and Culp 2003), therefore improved growing conditions may not be experienced by all plants in these areas of reduced ice scour stress.

Model results indicate an expected area-wide (*i.e.*, local extent) decrease in occupied aquatic macrophyte habitat in the long-term post-impoundment (Section 3.4.2.2; Table 4-16); although estimates vary from an approximate 20 ha reduction (*i.e.*, a 9% loss relative to existing conditions) in occupied aquatic plant habitat (when the reservoir is at FSL; Table 4-16) to a 158 ha decrease (*i.e.*, a 76% loss relative to existing condition) (when the reservoir is at MOL; Table 4-16). A relatively large amount of anticipated occupied plant habitat will be found within the portion of the littoral zone that is expected to be more or less dewatered on a frequent or infrequent basis depending on the mode of GS operation (peaking or base loaded). Water level cycling during peaking mode of operation in the future reservoir (*i.e.*, increased frequency of water level fluctuations in comparison to the existing environment) will degrade the quality of a portion of the upper littoral zone and aquatic macrophyte abundance, distribution, and community composition will likely respond to this cycling. When the GS is operating in peaking mode, water levels in the 19 km section of the reservoir upstream of the powerhouse could fluctuate by as much as 1.0 m per day (excluding wind effects); the magnitude of water level variation would diminish further upstream to the upstream boundary of the hydraulic zone of influence (Section 3.4.2.2). The negative effects of fluctuating water level may be less severe if the water is transparent, due to the wider productive zone, than if it is turbid (Rorslett 1988). The water quality assessment predicts reduced water clarity in flooded bay areas (particularly in shallow flooded bays off the mainstem of present-day Gull Lake), with greatest effects expected in the first year of impoundment and declining thereafter as peatland disintegration and erosion declines (Section 2.5.2.2). New vegetation in the upper littoral may consist of more disturbance-tolerant species due to any ongoing shoreline erosion and reductions in water clarity, and increased frequency of fluctuating water levels. Examples of such aquatic plant species in the Keeyask area include common spikerush, common mare's tail, lady's thumb, and white water-crowfoot.

The expected long-term reduction in aquatic habitat that produces filamentous algae (*i.e.*, fast water, hard substrate at rapids being lost due to flooding) will possibly result in up to an approximate 60% decrease in the production of drifting, non-vascular plant biomass from predominantly within Birthday Rapids (Appendix 3D, Table 3D-1). Although water velocity is being reduced and water depth is increasing in areas of the reservoir, water flow is expected to be adequate (short water residence time; low-medium-high water velocity aquatic habitat present) through shallow portions (shallow portion corresponds to euphotic zone) of the mainstem to produce drifting, non-vascular plant biomass. There will also be a substantial decrease in the production of drifting vascular plant biomass from predominantly within Gull Lake for at least 15 years post-impoundment as new, vegetated littoral habitat is established. A corresponding decline in the contribution of detritus (*i.e.*, organic carbon in the form of aquatic plant material) to downstream food webs will also likely occur.

Information from Other Reservoirs

Studies of water flow modifications in many temperate rivers show increases in aquatic macrophyte abundance following inundation (Bunn and Arthington 2002). This is due to the ability of plant species to proliferate in the slower flow rates and wider littoral zones associated with typical reservoir habitat. The limited areas of aquatic vegetation within the Keeyask area are in relatively protected bays (*i.e.*, off-current areas). Most species present are adapted to the existing conditions of the already regulated Nelson River: shallow, soft-bottom, minimal water velocity areas with annual water level fluctuations as much as 2.2 m (post-impoundment water level fluctuations will be reduced to 1.0 m, but with increased frequency [*i.e.*, daily vs. annual]).

In Stephens Lake, the occurrence of aquatic plants was generally restricted to areas shallower than 2 m water depth, although some plants were observed in water depths up to 3 m. Distribution tended to be patchy with localized macrophyte beds ranging from relatively very sparse to dense. Pondweeds, northern watermilfoil, and water smartweed were the most common plants observed, with Richardson's pondweed and northern watermilfoil being the most abundant and relatively dense. Aquatic plants were most abundant in shallow water areas with soft, mineral-based sediments. The intermittently exposed portion of this habitat type was dominated by Richardson's pondweed; however, northern watermilfoil was also relatively common and *Stukenia* spp. and aquatic moss species were also found. The community sampled in the predominantly wetted portion was almost exclusively Richardson's pondweed. In shallow water habitat with soft, organic-based sediments, most typically observed at the terminal ends of the inundated bays (*e.g.*, Ross Wright Bay), the community composition shifted and northern watermilfoil was predominant with water smartweed also being abundant in localized areas. Species that are more tolerant of periodic dewatering and ice scour stress were not observed in the areas sampled.

Elders and resource harvesters from TCN have stated that there has been a large decrease in plant life (including areas of very large reeds) in Stephens Lake that they feel is due in part to deeper, more turbid water, which does not allow the sun to reach the plants. They stated that the loss of plants appears more prevalent in the last ten years (changes started in 1984 or 1985), which they related to an observed decrease in water quality. The resource harvesters also commented on the colour of the algae attached to rocks along the shoreline, which they say has changed from green to a 'brownish colour'. These changes were linked to hydroelectric development in general and not to one specific project (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

Few aquatic macrophytes were found in the lower Nelson River mainstem during surveys conducted in 1992 (NSC 2012). The majority identified were located along a shallow 3 km reach of the north shore of the Long Spruce reservoir, an area that was flooded in 1979 (impounded for approximately 13 years at the time of sampling). Within this area, a macrophyte community, including two sedges (water sedge [*Carex aquatilis*] and common spike-rush) and water parsnip (*Sium sauve*), covered approximately 90% of the shoreline. In the same area, approximately 10–20% of the littoral zone supported beds of submerged plants dominated by Richardson's pondweed. Over the remainder of the Long Spruce reservoir, aquatic plant growth was patchy and limited to bays or flooded islands. Only 5–10% of the shoreline supported emergent vegetation and less than 1% of the littoral zone contained submergent macrophytes. Limited growth of submerged rooted vegetation is typical of reservoirs with pronounced water level fluctuations, where little permanently wetted habitat receives sufficient light to support photosynthesis during the growing season. In addition, the limited area of fine-textured substratum along the shoreline and periodic ice-scour on the lower Nelson River limit potential habitat for macrophytes.

In contrast to the Long Spruce reservoir, no aquatic macrophytes were observed in 1992 along the entire shoreline of the Limestone reservoir, including the creek mouths (sampled approximately three years after impoundment) (NSC 2012). Studies conducted in 2004, subsequent to the completion of the Limestone Monitoring Program, have found that aquatic plants have colonized limited areas in sheltered aquatic habitats such as flooded creek mouths and bays (Burt 2007).

Typically, changes in the littoral zone of regulated lakes result from alterations in water level fluctuation (extent, frequency). As reported in numerous Scandinavian lakes, exposure and erosional processes directly affect the littoral zone, disrupting the rooted aquatic macrophytes and affecting the succession of vegetation species (Hellsten 2000). The new vegetation on eroded shores typically consists of disturbance-tolerant species (*e.g.*, white-water crowfoot, common spikerush) adapted to the altered ecological environment (*e.g.*, Murphy *et al.*, 1990), which is under succession for several decades (Nilsson and Keddy 1988). The effects of water level regulation are also related to water quality. When the water is transparent, the negative effects of a fluctuating water level are less severe due to the wider productive zone, than in the case of turbid water (Rorslett 1988). In the upper littoral, white-water crowfoot and common spikerush are relatively tolerant of bottom freezing, intermittent exposure, and erosion; pondweeds (specifically, *Potamogeton* spp.) grow in deeper water to avoid the ice pressure zone (sensitive to ice) (Hellsten 2000).

4.3.4.2.3 Downstream of the Keeyask Generating Station

Operation-related pathways that were assessed for potential effects to aquatic macrophyte distribution and abundance included: a reduction in the extent and severity of ice scour and deposition of fine sediments in certain areas near the inflow of the river to Stephens Lake, and direct loss of aquatic habitat due to dewatering of Gull Rapids and the footprint of the GS structure. Summaries of predicted responses of aquatic macrophytes to changes resulting from the operation of the Project are presented in Figure 4-6. Where feasible, the effects of these pathways were considered using modelling exercises (quantification of potential effects), empirical information from Stephens Lake and other reservoirs in northern Manitoba, reservoirs in other northern temperate areas, and the scientific literature.

The direct loss of aquatic habitat that seems to produce filamentous algae (*i.e.*, fast water, hard substrate at rapids being lost due to dewatering and GS footprint) may result in a decrease in the production of drifting, non-vascular plant biomass (predominantly filamentous algae) from within Gull Rapids. Additionally, the GS itself will act as a physical barrier, thereby impeding or restricting the drift of plant biomass downstream to some extent.

4.3.4.2.4 Access Road Stream Crossings

Loss of aquatic plants due to the placement of the culvert and alteration due to the placement of riprap in the smaller streams will continue through the operating period. No incremental effects related to sediment inputs from erosion are expected due to the application of erosion control measures. No effects to aquatic plants in Looking Back Creek are expected.

4.3.4.2.5 Net Effects of Operation with Mitigation

The impoundment of the Nelson River at Gull Rapids will produce large changes in the aquatic macrophyte community, predominantly within the more lacustrine, downstream portion of the reservoir and to a lesser extent in the Nelson River immediately downstream of the GS. Post-impoundment, the vast majority of existing aquatic vascular plant beds will be lost due to flooding. This will also result in a substantial decrease in the production of drifting vascular plant biomass from predominantly within Gull Lake for up to 15 years post-impoundment as new, vegetated littoral habitat is established, and a corresponding decline in the contribution of detritus to downstream food webs. Overall, a reduction in occupied aquatic macrophyte habitat is expected in the reservoir in the long-term. New vegetation in the upper littoral may consist of more disturbance-tolerant species due to any ongoing shoreline erosion and reductions in water clarity, and increased frequency of fluctuating water levels. Examples of such aquatic plant species in the Keeyask area include common spikerush, common mare's tail, lady's thumb, and white water-crowfoot. Species that are more tolerant of periodic dewatering and ice scour stress were not observed in the intermittently exposed, shallow areas sampled in Stephens Lake during the environmental studies; depending on the substrate type, either Richardson's pondweed (mineral-based sediments) or northern watermilfoil (organic-based sediments) dominated the community. The reduction in fast water (high velocity) and hard substrate at rapids due to flooding, dewatering, and/or footprint of GS will result in a reduction in the production of drifting, non-vascular plant (filamentous algae) biomass and a corresponding decline in the contribution of detritus to downstream food webs.

Access road stream crossings will result in the loss of aquatic plants in the immediate footprint of the access road and culvert(s).

4.3.4.3 Residual Effects

4.3.4.3.1 Construction Period

The residual effects of construction will include a moderate decrease in the production of filamentous algae in Gull Rapids as the area is progressively dewatered during the construction of cofferdams.

No residual effects on aquatic macrophytes are expected.

4.3.4.3.2 Operation Period

The residual effects of operation on aquatic macrophytes and attached algae are:

- Loss of existing plant beds in Gull Lake;
- Establishment of new plant beds 10–15 years post-impoundment, though the total area occupied is expected on average to be less than in the existing environment given that many shallow flooded areas are organic and that plants in the reservoir are expected to colonize and inhabit less of the available habitat (based on a study of Stephens Lake); and
- A reduction in the production of drifting, filamentous algae from fast-flowing hard substrate areas of the river.

4.3.4.3.3 Summary of Residual Effects

The effects of the construction and operation of the Project on aquatic macrophytes are expected to occur over a medium extent in the reservoir, and will be large in the short-term, decreasing to small in the long-term. The effects of the construction and operation of the Project on attached algae are expected to be large over a small geographic extent (at the GS site), small over a medium geographic extent (local area with the reservoir and immediately downstream in Stephens Lake), and long-term. Expected residual effects to the aquatic plant community in terms of abundance were assessed and are presented in Table 4-17A and Table 4-17B for the construction and operation periods, respectively.

The technical aquatic macrophyte and attached algae assessment is based on models, scientific literature, and information collected from a proxy reservoir (*i.e.*, Stephens Lake) and the overall certainty associated with the predictions is moderate to high. Overall, there is high certainty regarding the nature and direction of effects and the magnitude of effects predicted for the loss of existing plant beds and reduction in aquatic habitat that produces filamentous algae, and moderate certainty regarding the magnitude of effects predicted for the colonization of flooded areas.

4.3.4.4 Environmental Monitoring and Follow-up

As described in Chapter 8 of the Response to EIS Guidelines, Environmental Monitoring Plans have been developed as part of the Environmental Protection Program for the Project. A comprehensive AEMP will be developed that specifically outlines monitoring to measure the effects of the Project on the aquatic environment, and discusses how results will be used as a basis for adaptive management. The AEMP will include monitoring of the aquatic macrophyte community to verify the results of the macrophyte assessment (*e.g.*, to determine whether aquatic plants colonized the flooded areas as predicted).

Aquatic macrophyte community variables are not considered VECs from an environmental assessment perspective; however, as supporting variables for other AEMP components, aquatic plant community variables do provide important measurement endpoints indicating the suitability of waterbodies to support aquatic life, and indicating potential change within or outside the range of natural variability that may be attributed to the operation of the Project.

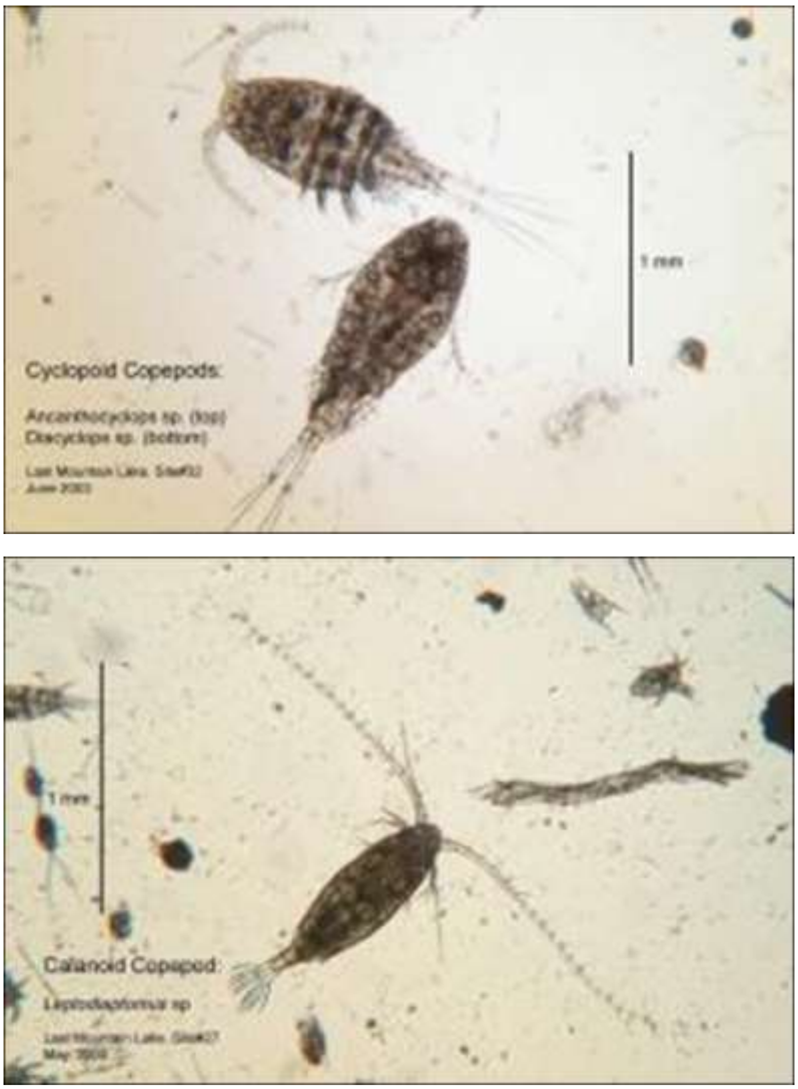
The aquatic macrophyte community AEMP would likely be conducted in conjunction with aquatic habitat monitoring and monitoring activities would likely take place within the specific effects monitoring SEM category. The SEM category encompasses more focussed monitoring components that would be geared towards evaluating effects of the Project in relation to predicted site-specific and/or local effects (*e.g.*, a large increase in aquatic macrophyte cover is expected in the reservoir in the long term in response to the increased availability of aquatic habitat [creation of shallow flooded areas with organic-based or mineral-based substrates]. New vegetation in the upper littoral would likely consist of disturbance-tolerant species due to eroding shoreline, increased frequency of fluctuating water levels, increase in TSS, and overall reduced water clarity). Aquatic plant community monitoring would be conducted annually during in-stream construction and for the first three years of operation; monitoring would then be conducted every three to five years for the first 20–30 years of operation, depending on results obtained.

Reports detailing the outcomes of monitoring programs will be prepared and submitted to MCWS and DFO, in compliance with the *Environment Act* and the *Fisheries Act*, respectively.

4.4 ZOOPLANKTON

4.4.1 Introduction

Zooplankton are very small animals without backbones (invertebrates) living in the water column and are consumed by larval, juvenile, and adult (*e.g.*, cisco) fish. This study includes all zooplankton retained by a 63 µm mesh size. Three important groups in the open water are Cladocera (water fleas), and calanoid and cyclopoid Copepoda (copepods) (Photo 4-6).



Source: Saskatchewan Environment [K. Scott], 2003

Photo 4-6: Representatives of cyclopoid (top panel) and calanoid (bottom panel) copepods

Cladocerans reproduce asexually for the majority of the year in most habitats, enabling this group to increase rapidly in density in response to favourable environmental conditions (*e.g.*, warming water temperatures, increasing food availability). Copepod reproduction is exclusively sexual; their lifecycle is quite prolonged to accommodate several developmental stages. In comparison to cladocerans, copepods are slow reproducers requiring several months to years to complete a lifecycle. As a result, cladocerans are able to take advantage of favourable growing conditions and peak in abundance, while the copepods are not.

Most species of cladocerans and copepods feed by filtering or grazing particles (bacteria, detritus, and phytoplankton) from the water, though there are a few predatory species. The availability and quality of food (*e.g.*, amount and kinds phytoplankton), the number of predators (*e.g.*, other invertebrates, fish), and water residence time affect the abundance of zooplankton; in rapidly flushed lakes and rivers little zooplankton biomass accumulates except in areas where there is little current. Impoundment of rivers to form reservoirs may lead to an increase in zooplankton production.

4.4.2 Approach and Methods

4.4.2.1 Overview to Approach

The approach taken for the zooplankton effects assessment was similar to the general approach taken for other aquatic environment components and was comprised of two major steps:

- A description of the existing aquatic habitat conditions to provide the basis for assessing the potential effects of the Project on these components; and
- An effects assessment in which the predicted post-Project environment was described and changes from existing environment quantified.

An ecosystem-based approach was employed to assess the potential impacts of the Project on the zooplankton community. Information presented incorporates findings from other aquatic environment components (*e.g.*, surface water quality and aquatic habitat). This approach is consistent with the views held by the KCNs, and widely held ecological views, that all components of the aquatic environment are important to maintaining the whole, and that all organisms are interdependent and, therefore, of importance and value.

The environmental setting is described using several sources of information, including: existing published information; and studies conducted specifically as part of the Keeyask environmental studies in 2001 and 2002. Potential Project-related effects on the zooplankton community were assessed using basic models (*i.e.*, simple conceptual models, quantitative models based on changes in habitat area, and qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments). These sources of information and effects assessment approaches are described in the following sections.

4.4.2.2 Study Area

The study area for zooplankton investigations extends along the Nelson River from Split Lake downstream to Stephens Lake in the east (Map 1-2). The magnitude of physical change (*e.g.*, changes in water levels and flows) differs substantially among areas (Section 3.2.2) and, consequently, the zooplankton study area was divided into three areas on the Nelson River as follows:

- Split Lake area (Split Lake and adjoining waterbodies, including Assean Lake and Clark Lake). This area is upstream of any direct Project influence. The zooplankton community in this area was described to provide supporting information for studies of other aquatic biota (Section 5 and Section 6).
- Keeyask area (Nelson River extending from the outlet of Clark Lake to approximately 3 km downstream of Gull Rapids, *i.e.*, hydraulic zone of influence, and tributary streams). Project-related changes to the water regime and direct losses of habitat due to the presence of the GS will occur within this reach (Section 3.2.2). This area was subdivided at Gull Rapids, as the rapids mark a boundary between the reservoir and downstream environment in the post-Project environment.
- Stephens Lake area (Stephens Lake and adjoining waterbodies). This area is immediately downstream of the Keeyask area and the Project will not affect the water regime. Stephens Lake, as the reservoir of the Kettle GS formed in the early 1970s, provides a useful proxy to assist in predicting effects of the Project (Section 1). The zooplankton community in Stephens Lake could also be affected by changes in the immediate upstream environment.

The majority of lower trophic levels investigations were conducted in the Keeyask area, as this area will be directly affected by the Project.

4.4.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for zooplankton are detailed below.

Few zooplankton community studies have been previously conducted in the Aquatic Environment Study Area. Any data collection programs were primarily focussed on the effects of hydroelectric generating stations (*e.g.*, construction and operation of the Kettle GS) or on the effects of the CRD/LWR project and were limited to GS reservoirs along the lower Nelson River, and Split and Stephens lakes.

Zooplankton data were collected in Split and Stephens lakes between 1986 and 1989 as part of the MEMP (Ramsey *et al.* 1989; Janusz 1990b). In 1992 and 2002, limited zooplankton data were collected as part of the Lower Nelson River Forebay Monitoring Program (included Long Spruce and Limestone reservoirs, and the lower Nelson River) (Schneider and Baker 1993; NSC 2012) and again in 2004 as part of the Conawapa GS environmental studies (Burt and Neufeld 2007a). The effects of previous hydroelectric development in northern Manitoba were assessed on the Split Lake RMA as part of the Split Lake Cree PPER (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

As the abundance and composition of zooplankton varies throughout the year due to seasonal variations in food availability and water temperature, zooplankton were collected from Split Lake, Clark Lake, Assean Lake, Gull Lake, and Stephens Lake during four sampling periods (June, July, August, and September/October) in 2001 and 2002 in the study area. The detailed approach and methods for zooplankton community studies conducted in 2001 and 2002 are presented in Appendix 4A.

4.4.2.4 Assessment Approach

Given the complexity of the aquatic ecosystem, models were used for predicting effects of the Project. Within the aquatic assessment, the complexity of models employed depended on: the importance of the issue; availability of information or suitable models; and utility of modelling approaches.

Basic model types used to assess potential Project effects on the zooplankton community were:

- Simple conceptual models (*e.g.*, alteration in off-current areas with respect to nutrient and TSS concentrations leads to an indirect effect on zooplankton abundance). The scientific literature was used to describe and support linkages to the Project.
- Quantitative models based on changes in aquatic habitat area (*e.g.*, calculation of total zooplankton abundance [*i.e.*, 'standing stock'] increase post-impoundment based on the predicted increase in reservoir volume) over the short term and long-term post-Project.
- Qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments. For example, Stephens Lake was used as a surrogate for post-Project conditions in the Keeyask reservoir.

The evaluation of certainty for predicted effects was based in part on the agreement of predicted effects among the various approaches.

4.4.3 Environmental Setting

4.4.3.1 Overview and Regional Context

The environmental setting has been described based on available background data and information collected in the course of the Keeyask environmental studies. The zooplankton community in the study area has been influenced by past hydroelectric development in northern Manitoba (*e.g.*, Kelsey GS, CRD, and LWR).

Thirty-two taxa of zooplankton have been recorded in the study area between 2001 and 2002 (Appendix 4B); however, community composition is discussed below in terms of major groups. There were no invasive zooplankton species observed in the study area (Invasive Species Council of Manitoba 2012).

Zooplankton abundance varied within study area lakes in both study years. Abundances were higher at standing-water sites (secluded bays that were relatively isolated from the flow in the Nelson River) than at flowing-water sites (mainstem). Overall, mean zooplankton abundance for each open water season investigated was highest in Assean Lake, an un-regulated waterbody removed from any influences of the

Nelson River, followed by Split, Clark, Gull, and Stephens lakes. Shorter water residence times and higher turbidity at flowing-water sites reduce the ability for zooplankton to maintain positive net growth rates due to downstream losses. Although zooplankton abundance differed among lakes sampled, the number of different species observed was comparable among areas.

The most diverse zooplankton community was observed in Split Lake during the study period, followed by Assean, Stephens, Clark and Gull lakes. Zooplankton abundance for the majority of sites sampled was typically higher during the summer months, and into the fall for some sites; this pattern was reversed in Gull Lake in 2002, with abundances peaking in the spring and early summer. Copepoda (predominantly cyclopoids) were more abundant than cladocerans in the spring throughout the study area and dominated the community. This was as expected since cladoceran species overwinter in low population densities as either adult females or resting eggs (Wetzel 1983). In early summer, there was a shift in the zooplankton community; cladoceran densities tended to increase and they often predominated throughout the remainder of the open water sampling program.

Zooplankton data were not collected at access road stream crossing locations. Planktonic crustacean zooplankton cannot maintain positive net growth rates in small streams for a number of reasons including downstream losses; as a result, they constitute a relatively unimportant component of the lower trophic level community in these environments (Hynes 1970). The benthic community tends to dominate invertebrate production in small streams (Horne and Goldman 1994).

Zooplankton abundance and composition within the study area were comparable to other waterbodies in northern Manitoba. Lakes within the Burntwood River system including Notigi, Wapisu, Threepoint, Sesep, Wuskwatim, Opegano, and Birch Tree lakes and Wuskwatim Brook, had similar zooplankton seasonal trends and variability as lakes in the study area (Manitoba Hydro and NCN 2003).

Similar to the study area lakes, zooplankton generally peaked within the Burntwood River system in July or August with cladoceran abundances increasing as the open water season progressed. Not unexpectedly, the June sampling period was dominated by copepods as it was in the Aquatic Environment Study Area lakes.

Off-system lakes in each system (*i.e.*, Leftrook Lake on the Burntwood system and Assean Lake in the study area) were observed to have greater zooplankton abundances in comparison to their corresponding on-system lakes. Relatively higher zooplankton abundances in Leftrook and Assean lakes were likely related to lower turbidity and less zooplankton loss due to downstream water movement in these lakes in comparison to their corresponding on-system lakes. In both the Burntwood system and Aquatic Environment Study Area, higher current areas typically had lower zooplankton abundances. In general, water velocity is negatively correlated to zooplankton biomass (Wetzel 2001).

4.4.3.2 Split Lake Area

In the late 1980s, zooplankton biomasses in Split Lake were found to be within the range at other mainstem sampling locations (Ramsey *et al.* 1989). Seasonal mean zooplankton biomass was similar between years and copepods dominated in both years of the study. Inter-annual variability in zooplankton abundance and composition (*i.e.*, the timing of the seasonal maximum and the dominant zooplankton group varied among years) was evident for the majority of locations sampled.

In the Split Lake area during the environmental studies, overall mean zooplankton abundance was greatest in Assean Lake, an unregulated waterbody with relatively little water movement, followed by Split and Clark lakes (Table 4-18). In Assean Lake, the overall mean catch densities in 2001 and 2002 were 47,516 and 42,254 individuals/m^3, respectively. Overall zooplankton densities were notably lower in Split Lake (standing and flowing water sites: 2,929 and 6,380 individuals/m^3 in 2001 and 2002, respectively) and Clark Lake (flowing water sites: 2,672 and 2,845 individuals/m^3 in 2001 and 2002, respectively). If only standing water sites in Split Lake were considered, overall zooplankton density was considerably higher (standing water sites: 7,619 and 14,664 individuals/m^3 in 2001 and 2002, respectively) than when both standing and flowing water sites were considered (Table 4-18). The catch densities within each lake varied between years and among sites and sampling dates (Table 4-18, Figure 4-7, Figure 4-8 and Figure 4-9).

Diversity of the zooplankton community was highest in Split Lake with 30 cladoceran and copepod taxa observed in 2001 and 27 in 2002 (Table 4-19). Total zooplankton abundance was greater in areas of Split Lake with standing water (Figure 4-7). Zooplankton abundance varied among sites and sampling periods, and between years; however, with the exception of early season peaks in abundance at a few sites, zooplankton tended to be more abundant during the summer months. Copepods, predominantly cyclopoids, comprised the majority of the community sampled in the spring with cladocerans increasing in abundance as the open-water season progressed and dominating the community during the summer months and into the fall.

Zooplankton diversity in the Split Lake area was lower in Clark Lake in comparison to the other waterbodies sampled (Table 4-19). As was observed for the majority of sites sampled in Split Lake, zooplankton abundance peaked during the summer months. Cyclopoid copepods were most abundant in the spring and cladocerans became increasingly abundant throughout the summer and fall (Figure 4-8).

Twenty-seven cladoceran and copepod taxa were collected from Assean Lake in 2001, and 23 taxa were collected in 2002 (Table 4-19). Although zooplankton abundance varied among sites and sampling periods, and between years, total abundance was typically higher in late summer and into the fall (Figure 4-9). Copepods, predominantly cyclopoids, comprised the majority of the community sampled in the spring and earlier summer with cladocerans increasing in abundance as the open-water season progressed.

4.4.3.3 Keeyask Area

No data or assessment of the effects of hydroelectric development on the zooplankton community prior to 1997 in the reach of the Nelson River between Clark Lake and Stephens Lake were located in the published literature.

In Gull Lake, the overall mean catch densities in 2001 and 2002 were 779 and 705 individuals/m^3, respectively (Table 4-18). Zooplankton abundance varied between years, and among sampling sites and dates (Table 4-18 and Figure 4-10).

Twenty-one zooplankton taxa were collected from flowing water areas of Gull Lake in 2001 and 2002 (Table 4-19). In 2001, total abundance was higher later in the summer and into the fall; however, this pattern was reversed in 2002 (Figure 4-10). Although the total zooplankton abundance seasonal pattern differed between study years, the composition of the community reflected that observed in other parts of the study area; cyclopoids comprised the majority of the community sampled in the spring and earlier summer with cladocerans contributing to a greater proportion of the community later in the season.

4.4.3.4 Stephens Lake Area

In the late 1980s, the zooplankton communities at the mainstem stations in Stephens Lake were described by Ramsey *et al.* (1989) as being the most depauperate of all those included in the MEMP (*i.e.*, Cross, Split, and Sipiwesk lakes). This result was attributed to the short water retention time, which did not permit zooplankton biomass to accumulate. Copepods were the dominant zooplankton group. As was observed in other lakes on the Nelson River system, higher zooplankton standing stocks were found to occur in backwater areas of Stephens Lake, than at the mainstem locations (Ramsey *et al.* 1989; TetrES Consultants Inc. and NSC 1998). Similar to mainstem locations, copepods dominated the community, accounting for the majority of the mean total biomass in backwater areas.

In Stephens Lake, overall mean zooplankton density was 264 and 761 individuals/m^3 in 2001 and 2002, respectively (Table 4-18). Catch densities varied between years, and among sampling sites and dates (Table 4-18 and Figure 4-11).

During the study period, diversity of zooplankton was highest in 2001 with 24 taxa represented (Table 4-19). With the exception of one site in 2002 that peaked in abundance in the fall, zooplankton were most abundant during the summer months (Figure 4-11). The composition of the community reflected that observed in other parts of the study area.

4.4.3.5 Current Trends

Limited historic zooplankton data were collected from Split and Stephens lakes in the late 1980s as part of the MEMP. Comparison of these data with zooplankton data collected as part of the Keeyask environmental studies for the purpose of assessing current trends is limited. There were differences in sampling and analytical methods employed between studies, and zooplankton abundance and composition varied considerably within waterbodies and between study years. However, qualitative comparisons of zooplankton data over time are presented.

Zooplankton biomass at Stephens Lake mainstem sites in the 1980s was the lowest of all the lakes sampled under the MEMP (*i.e.*, Cross, Split, and Sipiwesk lakes), while zooplankton biomass in Split Lake was higher and found to be comparable to other Nelson River mainstem sampling locations. Of the lakes sampled in the current study area, overall mean zooplankton abundance in the open water season was highest in Assean Lake, an unregulated waterbody removed from any influences of the Nelson River, followed by Split, Clark, Gull, and Stephens lakes. In the 1980s and the current study, higher zooplankton abundances were found to occur in backwater areas (*i.e.*, standing water sampling sites) than at mainstem locations (*i.e.*, flowing water sampling sites). Copepods tended to dominate the community sampled in the late 1980s. In the current study, copepods, predominantly cyclopoids, comprised the majority of the community sampled in the spring and earlier summer, with cladocerans increasing in abundance as the open water season progressed and contributing to a greater proportion of the community later in the season. Inter-annual variability in zooplankton abundance and composition was evident for the majority of locations sampled for both studies.

4.4.4 Project Effects, Mitigation and Monitoring

4.4.4.1 Construction Period

An assessment of potential Project effects on the zooplankton community during the construction period is based on the assessment of construction-related effects to surface water quality (Section 2.5.1, Table 2-12) and phytoplankton (Section 4.2.4.1). The primary potential effect(s) on zooplankton is related to inputs affecting water quality, such as increases in TSS concentrations and related variables (*i.e.*, turbidity) due to instream activities (*e.g.*, cofferdam placement and removal, river impoundment and diversion), and inputs or construction activities that affect dissolved oxygen (DO) concentrations in the lower Nelson River. Additionally, the zooplankton community may respond to any changes in phytoplankton as a result of inputs affecting water quality (Section 4.2.4.1). It is expected that construction effects (*i.e.*, inputs affecting water quality) will be managed through appropriate mitigation measures (Section 2.5.1), thereby reducing the duration and magnitude of any construction-related effects on the zooplankton community.

Currently, zooplankton abundance is relatively low at flowing water sites; zooplankton living in the water column in comparison to microcrustaceans that live associated with substrates (*e.g.*, bottom sediments, aquatic macrophytes) tend to be relatively unimportant in lotic environments as these organisms cannot maintain positive net growth rates (Hynes 1970).

4.4.4.1.1 Upstream of the Outlet of Clark Lake

No construction-related effects on the zooplankton community are expected upstream of the outlet of Clark Lake as there are no linkages between Project construction and surface water quality in Split, Assean, or Clark Lakes (Section 2.5.1).

4.4.4.1.2 Downstream of the Outlet of Clark Lake

The following sub-sections present the assessment of potential effects of construction activities on the zooplankton community in the Keeyask area and downstream.

Changes to Water Quality

Total Suspended Solids, Turbidity, and Water Clarity

Overall, the activities with the greatest potential to increase TSS concentrations in the lower Nelson River during construction of the GS are related to cofferdam placement and removal, and river impoundment and diversion (Section 2.5.1.1). Effects of suspended fine sediments on zooplankton are likely primarily related to its effect on zooplankton behaviour; the addition of suspended inorganic sediments typically decreases the feeding rate/activity (*i.e.*, ingestion and incorporation rates of algae) of filter-feeding zooplankton (*e.g.*, cladocerans such as *Daphnia* and *Bosmina*; calanoid copepods). The negative effect of additional suspended sediments on zooplankton feeding rate may be lessened if organic matter is adsorbed to the sediment particles, as the particles would then be useful as food. Generally, the construction and removal of cofferdams will generate an increase of less than 5 mg/L of TSS above background downstream of Gull Rapids (Section 2.5.1). Larger TSS increases are expected to be of relatively small magnitude and short duration. Peak levels are predicted to be up to 15 mg/L for one day or up to 7 mg/L for one month (Section 2.5.1). Drainage of surface runoff to the Nelson River will be controlled through a Drainage Management Plan (as described in the PD) to minimize the amount of sediment produced and the potential for sediment to enter watercourses. If the TSS concentration in water pumped out of cofferdam and excavation areas and in concrete wash water is greater than 25 mg/L the water will remain in a settling pond until it meets this TSS criterion before being discharged to the Nelson River. As the magnitude and duration of any increases in TSS are typically within the 30-day MWQSOG for PAL (an increase of 5 mg/L above background where background is less than or equal to 25 mg/L), the zooplankton community may be somewhat negatively affected in this downstream environment as their feeding rate may be reduced by suspended sediments, particularly mineral-based sediments (*i.e.*, small, undetectable reductions in zooplankton abundance may occur in affected areas during the construction period).

Dissolved Oxygen

During the latter stages of the Stage II Diversion, when water levels are increased to near full supply level, flooding of organic materials is expected to reduce DO concentrations in flooded areas (Section 2.5.1.2). Additionally, the earlier initiation of ice bridging upstream of Gull Rapids may cause upstream water levels to increase by 0.5–1.5 m during Stage I and Stage II Diversion in the event of a construction design flood. While these water level increases would remain within the range of water levels expected under a similar flow event during Project operation, this occurrence during construction may lead to DO depletion related to decomposition of flooded organic materials similar to that which would occur in the initial period post-impoundment (Section 2.5.1.2). Similar to the operation period, refugia for zooplankton from planktivorous fish predation (*e.g.*, rainbow smelt) may be created over flooded peat by low oxygen conditions (Paterson *et al.* 1997).

Metals and Contaminants (*e.g.*, Hydrocarbons)

Small amounts of metals will be introduced into the aquatic environment in association with construction activities that release sediments, as discussed in Section 2.5.1.6. However, given the proposed mitigation measures to manage sediment levels, these inputs are not expected to cause marked increases in metal levels and, consequently, will have no detectable effect on the zooplankton community.

The presence and levels of hydrocarbons in the aquatic environment could potentially be affected by accidental spills or release of substances containing hydrocarbons (*e.g.*, diesel fuel, gasoline, lubricating oil, *etc.*). Other hazardous substances will also be used during the construction period. As described in Section 2.5.1.6, the release of significant quantities of hazardous substances to the aquatic environment as a result of accidental spills and releases is considered unlikely due to the development and implementation of good management practices.

Phytoplankton

Predicted small reductions in phytoplankton biomass during Section construction (Section 4.2.4.1) may somewhat negatively affect zooplankton abundance during the construction period due to a decrease in a food source for filter feeding zooplankton.

4.4.4.1.3 Net Effects of Construction with Mitigation

Collectively, the above assessment points to the potential for small (*i.e.*, undetectable) decreases in zooplankton abundance during the construction period. Changes in abundance would occur over the short term downstream of the outlet of Clark Lake.

4.4.4.2 Operation Period

The availability and quality of food (*e.g.*, amount and kinds phytoplankton), the number of predators (*e.g.*, other invertebrates, fish), and water residence time affect the abundance of zooplankton; in rapidly flushed lakes and rivers little zooplankton biomass accumulates except in areas where there is minimal water flow. Impoundment of rivers to form reservoirs may lead to an increase in zooplankton production.

4.4.4.2.1 Upstream of the Outlet of Clark Lake

No response is expected. Selection of a 159 m ASL reservoir elevation instead of a higher elevation will avoid Project-related effects as Split Lake area is beyond the upstream extent of the expected hydraulic zone of influence.

4.4.4.2.2 Outlet of Clark Lake to the Keeyask Generating Station

Potential Project Effects and Proposed Mitigation

Operation-related pathways that were assessed for potential effects to the zooplankton community included: changes in surface water quality (*e.g.*, DO concentrations) (Section 2.5.2.2), changes in reservoir water residence time (increase in water level and volume, reduction in water velocity) (Section 3.4.2.2), and changes in the phytoplankton (Section 4.2.4.2) and planktivorous fish communities (Section 5.4.2.2). Summaries of predicted responses of zooplankton to changes resulting from the operation of the Project are presented in Figure 4-5. Where feasible, the effects of these pathways were considered using modelling exercises (quantification of potential effects), empirical information from Stephens Lake and other reservoirs in northern Manitoba, reservoirs in other northern temperate areas, and the scientific literature.

Assessment of Operation-Related Effects

Modelling Approach

Typically, predominantly riverine environments do not support an abundant zooplankton community. In many impoundments, zooplankton density rises in response to increases in the concentration of fine, particulate organic matter, water retention time, and phytoplankton biomass (Henriques 1987). Evidence from other northern Manitoba reservoirs also indicates a small increase in zooplankton abundance because of conversion of river to reservoir habitat (NSC 2012). However, only small increases in mean zooplankton abundance along the mainstem are expected in the Keeyask reservoir as increased water residence time will remain too short to permit a measurable increase in abundance; although total abundance ('standing stock') would increase with the predicted increase in reservoir volume (approximate doubling in comparison to the existing environment) (Section 3.4.2.2). Community composition should remain comparable to the current condition, with a community dominated by small cladocerans (*e.g.*, *Bosmina* spp.) and cyclopoid copepods. The lack of detectable effects may be attributed to high water flushing rates through the mainstem portion of the reservoir (*i.e.*, post-Project water residence time will be in the order of 15 to 30 hours, depending on flow; Section 3.4.2.2), and subsequently, the low accumulation of zooplankton in the reservoir. Short retention times are often associated with high turbulence (turbidity), a mixed waterbody, and a lack of thermal stratification. Zooplankton require a minimum retention time to allow development. If rates of water movement through a reservoir exceed a few millimetres per second, little plankton will develop (Hynes 1970).

Off-current areas could experience small to moderate increases in zooplankton abundance as water residence time in bays is estimated to be substantially longer than in the mainstem and could be up to one month long (Section 3.4.2.2). Post-impoundment conditions may favour bacteria over phytoplankton (Paterson *et al.* 1997). The addition of large amounts of newly flooded terrestrial organic matter may stimulate bacterial activity (increase the flow of carbon to higher trophic levels through the detrital pathway) and increase bacterial biomass in the medium term (5–10 years post-impoundment) instead of phytoplankton. An increase in bacterial biomass could provide a post-flooding food resource for zooplankton leading to an increase in zooplankton density and a shift in community composition to larger daphnids (more effective grazers on bacteria). Additionally, refugia for zooplankton from planktivorous fish predation (*e.g.*, rainbow smelt) may be created over flooded peat by low oxygen conditions (Paterson *et al.* 1997).

Information from Other Reservoirs

Shorter water residence times and higher turbidity at flowing water sites reduce the ability for zooplankton to maintain positive net growth rates due to downstream losses. Presently, mean zooplankton abundances during the open-water period (2001 and 2002) were higher at standing water (secluded bays that were relatively isolated from the flow in the Nelson River) sites than at flowing water (mainstem) ones (Section 4.4.3). Zooplankton abundance is variable and relatively low in study area waterbodies and downstream reservoirs (*i.e.*, Long Spruce and Limestone reservoirs, 2002 and 2004 open water period); although mean densities were higher in Gull Lake (742 individuals/L) and downstream reservoirs (Long Spruce: 2,035 individuals/L; Limestone: 1,037 individuals/L) in comparison to Stephens Lake (512 individuals/L), they were all low in comparison to standing water sites in lacustrine environments (*e.g.*, Assean Lake with greater than 40,000 individuals/L). Total suspended sediments and turbidity decrease along the flow of the Nelson River to the lower Nelson River (NSC 2012), which may contribute to the higher zooplankton densities observed in the downstream reservoirs in comparison to Stephens Lake. Despite the differences in mean zooplankton abundances among the waterbodies, overall community composition was similar between Gull (42% cladocerans, 14% calanoids, 44% cyclopoids) and Stephens lakes (52% cladocerans, 10% calanoids, 37% cyclopoids); cladocerans were more dominant in the downstream reservoirs (60–68%). The dominant cladoceran in each waterbody was *Bosmina* spp.

In the late 1980s, the zooplankton communities at the mainstem stations in Stephens Lake were described by Ramsey *et al.* (1989) as being the most depauperate of all those included in the provincial EMP study (*e.g.*, Cross, Split, and Sipiwesk lakes). This result was attributed to the short water retention time, which did not permit zooplankton biomass to accumulate. Copepods (calanoids, cyclopoids) were the dominant zooplankton group. As was observed in other lakes on the Nelson River system, higher zooplankton standing stocks were found to occur in backwater areas (*i.e.*, off-current) of Stephens Lake rather than at the mainstem locations, which supports the likelihood of an increase in zooplankton abundance in reservoir bays post-impoundment (Ramsey *et al.* 1989; TetrES Consultants Inc. and NSC 1998). Similar to mainstem locations, copepods dominated the community, accounting for the majority of the mean total biomass in backwater areas.

In the first few years of impoundment, large zooplankton populations may develop in flooded terrestrial areas where primary producers (periphyton, phytoplankton) are able to develop, with littoral cladocerans becoming particularly abundant (Northcote and Atagi 1997). Typically, post-impoundment increases in zooplankton abundance or biomass follow the changes in phytoplankton biomass. For example, in the Robert-Bourassa reservoir in Quebec, zooplankton biomass reached a maximum four years after impoundment and then declined; there was a one-year lag between maximum phytoplankton and zooplankton production. The increase in zooplankton was associated with an increase in water residence time (Hayeur 2001).

4.4.4.2.3 Downstream of the Keeyask Generating Station

Downstream effects on water quality are not expected to be substantive as the conditions of the reservoir outflow will not be considerably different from current conditions (Section 2.5.2.3). The major exception is a predicted decrease in TSS at the outflow of the GS. Furthermore, TSS is expected to decrease further as water moves through Stephens Lake and this area of reduced TSS would likely extend approximately 10–12 km downstream of the GS. This improvement in water clarity is expected to result in a small increase in zooplankton abundance in the affected portion of Stephens Lake over the long-term (Figure 4-6). Given the small changes in phytoplankton biomass and zooplankton abundance upstream, no measurable change is expected.

4.4.4.2.4 Net Effects of Operation with Mitigation

The impoundment of the Nelson River at Gull Rapids will produce small to moderate increases in mean zooplankton abundance over the long-term in reservoir bays with longer water residence times. An increase in bacterial biomass could provide a post-flooding food resource for zooplankton leading to an increase in zooplankton density and a shift in community composition to larger daphnids (more effective grazers on bacteria). Additionally, refugia for zooplankton from planktivorous fish predation (*e.g.*, rainbow smelt) may be created over flooded peat by low oxygen conditions.

4.4.4.3 Residual Effects

4.4.4.3.1 Construction Period

No residual effects on zooplankton are expected.

4.4.4.3.2 Operation Period

A small to moderate increase in zooplankton biomass is expected in off-current portions of the reservoir.

4.4.4.3.3 Summary of Residual Effects

The effects of the construction and operation of the Project on zooplankton are expected to be small to moderate, and long-term, and to occur over a small geographic extent. Expected residual effects to the zooplankton community in terms of abundance were assessed and are presented in Table 4-20A and Table 4-20B for the construction and operating periods, respectively.

The technical zooplankton assessment is based on models, scientific literature and information collected from a proxy reservoir (*i.e.*, Stephens Lake), and the overall certainty associated with the predictions is moderate to high. Overall, there is high certainty regarding the nature and direction of effects and the magnitude of effects predicted for the mainstem of the reservoir, and moderate certainty regarding the magnitude of effects in nearshore areas of the reservoir.

4.4.4.4 Environmental Monitoring and Follow-up

As described in Chapter 8 of the Response to EIS Guidelines, Environmental Monitoring Plans have been developed as part of the Environmental Protection Program for the Project. A comprehensive AEMP will be developed that specifically outlines monitoring to measure the effects of the Project on the aquatic environment, and discusses how results will be used as a basis for adaptive management. The AEMP will include monitoring of the zooplankton community to verify the results of the zooplankton assessment.

Zooplankton community variables are not considered VECs from an environmental assessment perspective; however, as supporting variables for other AEMP components, zooplankton community variables do provide important measurement endpoints indicating potential change within or outside the range of natural variability that may be attributed to the operation of the Project. The zooplankton community AEMP would be conducted in conjunction with the surface water quality AEMP.

Monitoring activities for the zooplankton community may be split into two major categories: (1) CM; and (2) SEM. The former is aimed at evaluating effects of the operation of the Project throughout the Aquatic Environment Study Area (*i.e.*, over a broad geographical scale) while the latter encompasses a more focussed monitoring component that will be geared towards evaluating effects of the Project in relation to predicted site-specific and/or local effects (*e.g.*, local effects predicted in reservoir bays with longer water residence times). Zooplankton community monitoring would be conducted annually during instream construction and for the first three years of operation; monitoring would then be conducted every three to five years for the first 20–30 years of operation, depending on results obtained.

Reports detailing the outcomes of monitoring programs will be prepared and submitted to MCWS and DFO, in compliance with the *Environment Act* and the *Fisheries Act*, respectively.

4.5 AQUATIC MACROINVERTEBRATES

4.5.1 Introduction

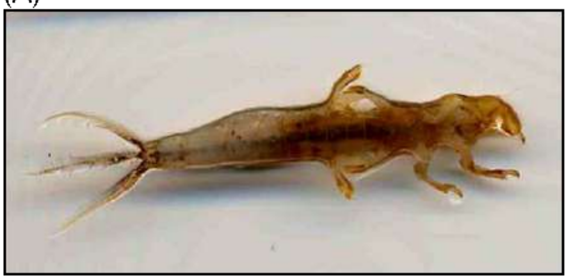
Aquatic macroinvertebrates are small animals without backbones living on or in the substrata of lakes and rivers [*e.g.*, clams (Bivalvia), aquatic earthworms (Oligochaeta), and aquatic insect larvae]. This study includes invertebrates that are retained by a 500 micrometre (µm) mesh size. Macroinvertebrates retained on 500 µm sieves are important food items to vertebrates (particularly fish) and useful bioindicators of environmental change.

Aquatic macroinvertebrates are typically a diverse assemblage, and are adapted to the range of substrate types and water flow regimes (*e.g.*, fast-flowing rivers, sheltered bays in lakes with no discernable flow) found in the aquatic environment. Beds of aquatic vegetation typically harbour the greatest density and diversity (*i.e.*, number of taxa) of macroinvertebrates, with invertebrates living on leaf surfaces (plant-dwelling or epiphytic invertebrates) as well as on and within the sediments beneath the plants (sediment-dwelling or benthic invertebrates, or benthos). These include grazers of attached algae [*e.g.*, snails (Gastropoda), chironomids or non-biting midges (Chironomidae), and mayflies or fishflies (Ephemeroptera)], organisms that consume the organic-rich sediment (*e.g.*, aquatic earthworms), animals that eat the plants [*e.g.*, crayfish (Decapoda)], and a few carnivores [dragonfly nymphs (Anisoptera)] (Photo 4-7). Amphipods, scuds, or freshwater shrimp (Amphipoda), which consume a variety of decaying plant and animal matter, are also present (Photo 4-7). The higher densities of macroinvertebrates in aquatic macrophyte beds reflect higher productivity (via photosynthetic and detrital foodweb pathways) and protection from predators provided by the plants.

Shallow areas (*i.e.*, littoral zone) with mud or mud-sand bottoms provide habitat for filter-feeding clams, sediment-feeding aquatic earthworms, and a variety of insect larvae, many of which have terrestrial adult forms (Photo 4-7). Emergence of larval insects to terrestrial adults results in a loss of numbers and biomass from the aquatic system. Substrata such as sand or gravel usually harbour fewer animals because water currents (*e.g.*, wave action on shorelines) readily disturb these substrata. Sand (0.063–2 mm in diameter) is generally considered to be a poor substrate for macroinvertebrates due to its relative instability (*i.e.*, susceptible to water movement) and the tight packing of particles (*i.e.*, minimal interstitial space) reducing the trapping of organic matter (*i.e.*, detritus) and limiting the presence of DO. In general, diversity and abundance of macroinvertebrates increase with substrate stability and the presence of organic matter. However, sand-dwelling fauna includes some macroinvertebrates typically exhibiting adaptations associated with respiration [*e.g.*, certain types of caddisflies (Trichoptera)]. Generally, in riverine environments invertebrates are more abundant (*i.e.*, show a preference) in shallow water and in gravel or coarser substrates (*i.e.*, cobble) [particularly insects, such as mayflies, stoneflies (Plecoptera), and caddisflies] (Minshall 1984; Jowett *et al.* 1991). Total benthic invertebrate production tends to be relatively low on extremely fine (*e.g.*, silt/clay, sand) and extremely coarse (*e.g.*, boulder, bedrock) substrates, while productivity is typically highest for substrate particles averaging 10 mm in diameter (*i.e.*, coarse gravel)

(Morin 1997). Similar substrate preferences in lentic environments could be expected. In regulated systems, such as the reservoir areas of hydroelectric generating stations, the increased frequency of water level fluctuations tends to reduce macroinvertebrate abundance in regularly exposed areas along the shoreline depending on the GS mode of operation.

(A)



(B)

(C)

(D)

Source: North/South Consultants Inc., 2001

Photo 4-7: Representatives of aquatic macroinvertebrate groups: (A) chironomid larva; (B) ephemeropteran (mayfly) larva; (C) amphipod (scud); and (D) fingernail clam

Deeper areas of lakes are typically depositional environments with fine-textured sediments. Organisms that feed on sediment and detritus (*e.g.*, aquatic earthworms and chironomids) usually dominate in these areas. Many of these animals are adapted to low-oxygen conditions that can develop at greater water depths.

The aquatic macroinvertebrate community within a riverine system is comprised of bottom- and plant-dwelling and drifting organisms. Drifting invertebrates, organisms that move downstream in the water current, are a composite of invertebrates originating from a larger area and diverse array of habitats, but the drifting community tends to be biased towards those groups that drift as a life history strategy (predominantly aquatic insects). Aquatic invertebrate drift is an important and well-documented component of stream structure and function (Brittain and Eikeland 1988). The tendency for aquatic invertebrates to drift, and the distance traveled when drifting, is dependent on life history, and environmental, physical, biological, and chemical cues; and can result from: i) accidental dislodgment from substratum; ii) changes in the physical environment (*i.e.*, discharge, velocity); iii) interactions with other invertebrates; and iv) colonization (Elliott 1971). Drift varies over space (*e.g.*, position in the river) and time (*e.g.*, daily and seasonally). A well-documented temporal feature of aquatic invertebrate drift is the dusk peak, related to a change in light intensity and as an anti-predatory response; temperature and discharge are likely features dictating drift variation over longer periods such as seasons (Shearer *et al.* 2002). Reliable quantification of drift is necessary for investigating some aspects of trophic interaction between invertebrates and drift-feeding fish (Shearer *et al.* 2002). Drift-invertebrate sampling is useful for assessing habitat quality and food availability by examining the abundance and diversity of invertebrate taxa. Non-drifting and drifting macroinvertebrates are an important food source for fish and therefore, their abundance and distribution helps define the importance of an aquatic area as feeding habitat.

4.5.2 Approach and Methods

4.5.2.1 Overview to Approach

The approach taken for the aquatic macroinvertebrate effects assessment was similar to the general approach taken for other aquatic environment components and was comprised of two major steps:

- A description of the existing aquatic habitat conditions to provide the basis for assessing the potential effects of the Project on these components; and
- An effects assessment in which the predicted post-Project environment was described and changes from existing environment quantified.

An ecosystem-based approach was employed to assess the potential impacts of the Project on the aquatic macroinvertebrate community. Information presented incorporates findings from other aquatic environment components (*e.g.*, surface water quality and aquatic habitat). This approach is consistent with views held by the KCNs, and widely held ecological views, that all components of the aquatic environment are important to maintaining the whole, and that all organisms are interdependent and, therefore, of importance and value.

The environmental setting is described using several sources of information, including: existing published information; and studies conducted specifically as part of the Project between 1999 and 2006. Potential Project-related effects on the aquatic macroinvertebrate community were assessed using basic models (*i.e.*, simple conceptual models, quantitative models based on changes in habitat area, and qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments). These sources of information and effects assessment approaches are described in the following sections.

4.5.2.2 Study Area

The study area for aquatic macroinvertebrate investigations extends along the Nelson River from Split Lake downstream to Stephens Lake in the east (Map 1-2). The magnitude of physical change (*e.g.*, changes in water levels and flows) differs substantially among areas (Section 3.2.2) and, consequently, the aquatic macroinvertebrate study area was divided into three areas on the Nelson River as follows:

- Split Lake area (Split Lake and adjoining waterbodies, including Assean Lake and Clark Lake). This area is upstream of any direct Project influence. The aquatic macroinvertebrates in this area were described to provide supporting information for studies of other aquatic biota (Section 5 and Section 6).
- Keeyask area (Nelson River extending from the outlet of Clark Lake to approximately 3 km downstream of Gull Rapids, *i.e.*, hydraulic zone of influence, and tributary streams). Project-related changes to the water regime and direct losses of habitat due to the presence of the GS will occur within this reach (Section 3.2.2). This area was subdivided at Gull Rapids, as the rapids mark a boundary between the reservoir and downstream environment in the post-Project environment.
- Stephens Lake area (Stephens Lake and adjoining waterbodies). This area is immediately downstream of the Keeyask area and the Project will not affect the water regime. Stephens Lake, as the reservoir of the Kettle GS formed in the early 1970s, provides a useful proxy to assist in predicting effects of the Project (Section 1).

The majority of lower trophic levels investigations were conducted in the Keeyask area, as this area will be directly affected by the Project. Aquatic biota was also described as part of the assessment of the north and south access roads stream crossings.

4.5.2.3 Data and Information Sources

Section 1.5 summarizes the overall sources of information used for the Project, including technical studies, scientific publications and local knowledge. Specific sources of information used to characterize the environmental setting for aquatic macroinvertebrates are detailed below.

A number of benthic macroinvertebrate community studies have been previously conducted in the Aquatic Environment Study Area. Data were collected in the early 1960s in response to Vale's nickel mining and refining operation at Thompson, Manitoba. However, the majority of programs were primarily focussed on the effects of hydroelectric generating stations (*e.g.*, construction and operation of the Kettle GS) or on the effects of the CRD/LWR project and were largely limited to GS reservoirs along the lower Nelson River, and Split and Stephens lakes.

Benthic invertebrate data were collected in the Split Lake watershed from 1958 to 1960 and in 1966 to provide information in relation to the nickel and mining operation at Thompson, Manitoba (Beak 1962;

Schlick 1968). In the early 1970s, benthic invertebrates were investigated in the newly formed Stephens Lake (Kettle reservoir) as part of the LWCNRSB program (Crowe 1973). Benthic invertebrate samples were collected from Split and Stephens lakes in the late 1980s as part of the MEMP (Cann 1991). During the 1990s (1990, 1992–1996, 1999) and in 2003, benthic invertebrate data were collected for Manitoba Hydro by KCNs Members together with NSC as part of the Lower Nelson River Forebay Monitoring Program (including Long Spruce and Limestone reservoirs, and the lower Nelson River) (Schneider and Baker 1993; Schneider-Vieira 1994, 1996; Horne 1996, 1997; Zrum and Kennedy 2000; Burt and Neufeld 2007b; NSC 2012). The benthic macroinvertebrate community of Split Lake was described in 1997 and 1998 by TEMA for TCN and Manitoba Hydro (Lawrence and Fazakas 1997; Fazakas and Zrum 1999). The effects of previous hydroelectric development in northern Manitoba were assessed on the Split Lake RMA as part of the Split Lake Cree PPER (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

Detailed sampling to describe the habitat-based abundance, composition and distribution sediment-dwelling aquatic macroinvertebrates (benthos or benthic invertebrates) in the Aquatic Environment Study Area waterbodies was conducted between 1999 and 2006. Sampling to describe the plant-dwelling aquatic macroinvertebrates (epiphytic invertebrates) was conducted in 2001 and 2002 between Birthday and Gull rapids, in 2003 and 2004 for Clark Lake to Gull Rapids, and in 2005 and 2006 for Stephens Lake in conjunction with the aquatic macrophyte abundance and composition program (2001–2006). Two representative areas of Stephens Lake (Ross Wright and O'Neil bays) were surveyed more intensively in 2005 and 2006 (years with higher than average water levels; Section 3.3.2.3) to describe the benthic and epiphytic macroinvertebrate communities in areas that were historically inundated; these two areas were chosen to provide a proxy for the post-impoundment Keeyask reservoir. For Aquatic Environment Study Area waterbodies, the intermittently exposed zone was defined using historic water level percentiles. This area is the shore zone bounded by the 5th and 95th water level percentiles and represents a band along the edge of the waterbody that has experienced exposure, *i.e.*, dewatering, 5 to 95% of the time since 1977 (Section 3.2.4.1).

In riverine environments of the Aquatic Environment Study Area, the flowing water transports a relatively large amount of macroinvertebrate biomass downstream. Drift traps were used to sample drifting macroinvertebrates in tributaries and extensive areas of rapids where sampling by other methods (*e.g.*, dredge or air lift sampler) was not feasible due to logistical (*e.g.*, water depths and water velocities too high for effective sampling) or safety concerns (*i.e.*, high flows immediately upstream and downstream of Gull Rapids). Drifting macroinvertebrates were sampled at various locations along the Nelson River during the 2001 to 2004 open water seasons to gain an overall understanding of the spatial and temporal differences in abundance and distribution of biomass within the study area, and provide the basis for assessing potential changes in production from specific areas (*i.e.*, Birthday and Gull rapids, and Stephens Lake) associated with the proposed Keeyask hydroelectric development.

The aquatic macroinvertebrate community was sampled near the proposed access road ROW during the fall, 2004. Aquatic invertebrates were collected in the vicinity of all stream crossings using a kick net to provide a qualitative description of the community.

The detailed approach and methods for aquatic macroinvertebrate community studies conducted between 1997 and 2006 are presented in Appendix 4A.

4.5.2.4 Assessment Approach

Given the complexity of the aquatic ecosystem, models were used for predicting effects of the Project. Within the aquatic assessment, the complexity of models employed depended on: the importance of the issue; availability of information or suitable models; and utility of modelling approaches.

Basic model types used to assess potential Project effects on the aquatic macroinvertebrate community were:

- Simple conceptual models (*e.g.*, benthic invertebrate abundance will likely increase with an increase in wetted habitat resulting from reservoir formation). The scientific literature was used to describe and support linkages to the Project.
- Quantitative models based on changes in aquatic habitat area (*e.g.*, calculation of benthic invertebrate numbers based on specific areas of habitat types that had been sampled in the existing environment) over the short-term and long-term post-Project.
- Qualitative empirical models based on observed changes in the environment following similar developments in other Manitoba settings and in northern environments. For example, Stephens Lake was used as a surrogate for post-Project conditions in the Keeyask reservoir (Section 3.2.4).

A quantitative habitat-based model was used to estimate the abundance of sediment- and plant-dwelling macroinvertebrates in the newly created Keeyask reservoir at four time steps (Years 1, 5, 15, and 30) post-impoundment (Appendix 4A). The model used the mean abundance (individuals/m^2) of macroinvertebrates from baseline studies in the study area as an estimate of macroinvertebrate abundance at defined habitat types in the existing environment and as a predictor of macroinvertebrates in the same habitat types post-Project. An abundance estimate was generated for some habitat types that were not sampled because of methods constraints (*e.g.*, medium water velocity habitats) or because they were uncommon in the existing environment (*e.g.*, deep water, organic substrate habitats) using surrogate values from similar habitat types that were sampled or other comparable areas in northern Manitoba (*e.g.*, Wuskwatim area, Stephens Lake). The area of each habitat type was estimated for the Nelson River between the outflow of Clark Lake and the Keeyask GS location in the existing environment using Manitoba Hydro's shoreline data (the spatial extent of habitat types was modelled at 95th percentile flow conditions) and in Year 30 post-Project using the predicted shoreline at a water level elevation at the face of the dam of 158 m ASL for minimum operating level (MOL) or at 159 m ASL under 95th percentile flow conditions for FSL (Section 3.2.4; Appendix 3D). The Year 30 habitat areas were modified for the intermediate time steps (*i.e.*, Years 1, 5, and 15) to account for shoreline erosion, peat disintegration and transport, and loss and subsequent establishment of aquatic plant beds (Appendix 3D), and modelled DO and TSS concentrations (Appendix 4A). At all post-Project time steps, the model accounted for reductions in the quality of aquatic habitat for macroinvertebrates due to weekly cycling, *i.e.*, mode of GS operation (Appendix 3D).

The evaluation of certainty for predicted effects was based in part on the agreement of predicted effects among the various approaches.

4.5.3 Environmental Setting

4.5.3.1 Overview and Regional Context

The environmental setting has been described based on available background data and information collected in the course of the Keeyask environmental studies. The aquatic macroinvertebrate community in the study area has been influenced by past hydroelectric development in northern Manitoba (*e.g.*, Kelsey GS, CRD, and LWR).

No species are listed on Schedule 1 of SARA and none have been assessed as "at risk" by COSEWIC. The Manitoba Conservation Data Centre does not list any S1 or S2 species for the area (Manitoba Conservation Data Centre 2012a; Manitoba Conservation Data Centre 2012b). None of the species identified are listed as invasive on the Invasive Species Council of Manitoba website (Invasive Species Council of Manitoba 2012).

4.5.3.1.1 Split Lake Area

Within the Split Lake area, Assean Lake is removed from any influences of the Nelson River. Except for the mouth of the Assean River, the hydrology of the watershed has not been affected by hydroelectric development. For this reason, the macroinvertebrate community of the Assean watershed is described separately from other waterbodies in the Split Lake area.

Forty-nine macroinvertebrate taxa were observed in the sediment- and plant-dwelling communities investigated between 2001 and 2004 in the Split Lake area (Split and Clark lakes, including the York Landing Arm of Split Lake) (Table 4-21). Drifting invertebrates were not investigated in this area. When all study years were considered, the sediment-dwelling community appeared to be more diverse (44 taxa) than the plant-dwelling community (25 taxa). This pattern of higher taxa richness in sediment samples was particularly apparent for the aquatic insects (Appendix 4B).

The mean overall sediment-dwelling macroinvertebrate (benthos) abundance for aquatic habitats sampled in the Split Lake area was 3,319 individuals/m^2. The shallow water habitats supported a lower mean abundance of benthos than those described for the deep water. When the same type of aquatic habitat in terms of water level and flow, and substrate (compaction, composition, aquatic plants) was sampled for sediment- and plant-dwelling macroinvertebrates, a higher abundance of macroinvertebrates was observed in the sediments (2,316 individuals/m^2) than living on the plants (181 individuals/m^2). Chironomids dominated the community for both; however, mayflies and fingernail clams were more common in the sediments while snails and aquatic earthworms contributed more to the community in the plants.

Within the Assean watershed, 55 macroinvertebrate taxa were observed in the sediment-dwelling and drifting communities investigated between 2001 and 2004 (Table 4-21). When all study years were considered, the sediment-dwelling community appeared to be more diverse (40 taxa) than that represented in the drifting component (30 taxa); however macroinvertebrates were identified to a lower taxonomic level in 2003 and 2004 and this resulted in a step-trend increase to the number of taxa observed in comparison to 2001 and 2002. When sediment-dwelling sites were limited to 2001 and 2002 data, however, the number of taxa present was actually lower (19 taxa) (Table 4-21). Approximately 58% more taxa were recorded in drift samples than in sediment samples. This pattern of higher taxa richness in drift samples was particularly apparent for the aquatic insects (Appendix 4B).

The mean overall sediment-dwelling macroinvertebrate abundance for aquatic habitats sampled in Assean Lake was 2,207 individuals/m². The shallow habitats supported a higher mean abundance of benthos than deep habitats.

4.5.3.1.2 Keeyask Area

Within the Keeyask area, 93 macroinvertebrate taxa were observed in the sediment- and plant-dwelling, and drifting communities investigated between 1999 and 2004 (Table 4-21). When all study years were considered, the drifting community appeared to be more diverse (85 taxa) than the sediment- (43 taxa) and plant-dwelling (56 taxa) communities. The increased diversity observed in drift samples may be reflective of the higher degree of heterogeneity found in the Keeyask area aquatic habitats. This pattern of higher taxa richness in drift samples was particularly apparent for the aquatic insects (Appendix 4B). Drifting invertebrates may be derived not only from the sediment- and plant-dwelling communities in habitats immediately upstream of drift sampling locations, but also from habitats in the river channel and tributaries considerable distances upstream. Drift samples included a greater variety of taxa than other macroinvertebrate samples presumably because they integrate over a much greater spatial scale and also over time (Shearer *et al.* 2003).

The mean overall sediment-dwelling macroinvertebrate (benthos) abundance for aquatic habitats sampled in the Keeyask area was 3,539 individuals/m². The shallow water habitats supported a higher mean abundance of benthos than those described for deep water. When the same type of aquatic habitat was sampled for sediment- and plant-dwelling macroinvertebrates, there were four to ten times more macroinvertebrates observed in the sediments than living on the plants. Chironomids, aquatic earthworms, and amphipods were commonly found in both the sediments and associated with the plants. Aquatic earthworms and amphipods tended to contribute more to the plant-dwelling community, whereas mayflies and fingernail clams were additional important groups in the sediment-dwelling community.

Rapids in the Keeyask area provide areas of increased drifting invertebrate production with increased drift densities often observed in the aquatic habitats sampled downstream of Birthday and Gull rapids. Aquatic insects (specifically mayflies, caddisflies, and chironomids) were typically the most abundant drifting invertebrates collected in drift traps. The greatest drifting invertebrate densities in the study area were observed upstream of Gull Rapids (at the downstream end of Gull Lake), with the next highest densities observed downstream of Birthday Rapids, downstream of Gull Rapids, and upstream of Birthday Rapids. Therefore, it may be inferred that the majority of drifting invertebrates in the study area were produced by the Nelson River aquatic habitats between Birthday and Gull rapids, including Gull Lake, and within Birthday and Gull Rapids themselves. Production of drifting invertebrates from Gull Rapids is likely an important input to Stephens Lake; however, these rapids appear to produce overall fewer drifting invertebrates than the Nelson River between Birthday Rapids and the downstream extent of Gull Lake. Drifting invertebrates are an important food source for fish and therefore, their abundance and distribution helps define the importance of an aquatic area as feeding habitat. The drifting community is also important in terms of providing individuals to colonize downstream areas and thereby contributing to the composition and abundance of downstream communities.

4.5.3.1.3 Stephens Lake Area

Fifty-four taxa were observed in the macroinvertebrate communities investigated between 2001 and 2004 in the Stephens Lake area (Table 4-21). When all study years and areas of Stephens Lake were considered (2001 to 2006), including the two areas (Ross Wright and O'Neil bays) that were surveyed more intensively in 2005 and 2006, the diversity of the drifting community downstream of Stephens Lake (downstream of the Kettle GS) was slightly higher (40 taxa) than that of either the sediment- (32 taxa) or plant-dwelling (34 taxa) components. This pattern of higher taxa richness in drift samples was somewhat more apparent for the crustaceans and aquatic insects (Appendix 4B).

The mean overall sediment-dwelling macroinvertebrate (benthos) abundance for aquatic habitats sampled in the Stephens Lake area was 2,621 individuals/m^2. The shallow water habitats supported a higher mean abundance of benthos than those described for deep water. When the same type of aquatic habitat was sampled for both benthos and plant-dwelling macroinvertebrates, there were 2–27 times more macroinvertebrates observed in the sediments than living on the plants. Chironomids, snails, aquatic earthworms, and amphipods were commonly found in both the sediments and associated with the plants. Although present in the plant-dwelling community, chironomids were more abundant in the sediment, where mayflies were also abundant. Hydra (Hydrozoa) and clam shrimp (Diplostraca) were important components of the plant-dwelling community.

As at drift trap locations in the Keeyask area, aquatic insects (specifically mayflies, caddisflies, and chironomids) were typically the most abundant invertebrates collected in drift traps downstream of the Kettle GS. Overall, drift trap locations downstream of the Kettle GS were the least productive in terms of invertebrate drift density within the study area.

4.5.3.1.4 Access Road Area

The presence of aquatic invertebrates was described at five stream crossings along the proposed Keeyask GS access road, two on the north side of the Nelson River and three on the south side. Aquatic insects dominated the community at each stream crossing, comprising between 43 and 73% of the taxa observed. Caddisflies, dipterans, and mayflies typically dominated the community; however, snails were occasionally relatively common at some sampling locations. The lowest diversity (7 taxa) was observed at SC-5, a small unnamed tributary of Stephens Lake on the south side of the Nelson River that provides drainage to bog and fen areas, including a small lake upstream of the crossing. SC-5 was only assessed at the crossing site, which likely contributed to the relatively low number of taxa observed. The highest diversity of aquatic invertebrate taxa (33 taxa) was observed at SC-7. SC-7 crosses Looking Back Creek approximately 4 km upstream of Stephens Lake on the north side of the Nelson River. The crossing site is at the lower reaches of the creek with 95% of the 126 km^2 drainage area above the crossing. Aquatic habitat consisted entirely of run/glide habitat with a high amount and diversity of cover, including over stream vegetation, woody debris, cutbank, instream vegetation, and boulder. The diversity of habitat and size of the stream likely contributed to the greater diversity of aquatic invertebrates observed in comparison to other stream crossings.

4.5.3.1.5 Regional Context

The overall mean number of sediment-dwelling macroinvertebrates (benthos) observed for selected northern Manitoba waterbodies from the Churchill, Rat/Burntwood, and Lower Nelson river systems was variable (Table 4-22). Mean abundances ranged from 1,912 individuals/m² in the Rat River along the Churchill River diversion route to notably higher abundances of 9,529 individuals/m² in the Churchill River, prior to weir construction, and 8,439 individuals/m² in Leftrook Lake, a waterbody not on CRD that has not been impacted by past hydroelectric regulation. With the exception of chironomids and a few taxa of mayflies, insect larvae were not common in these areas. Instead, groups typical of larger rivers and lakes, such as amphipods, fingernail clams, snails, and aquatic earthworms predominated. With a few exceptions, the majority of habitat types investigated could be considered representative of relatively healthy and diverse aquatic habitat. Taxa expected to be observed in intermittently exposed, shallow water, and deep water habitats were present, and their relative proportions were similar to other waterbodies. At some sites, the taxa present were representative of a heterogeneous substrate (*e.g.*, a mixture of silt/clay-based, and gravel and cobble substrates), which is often found in the transitional shallow areas of larger rivers and lakes.

Abundances of sediment-dwelling macroinvertebrates in the study area along the Nelson River were within the range observed for waterbodies along the Churchill, Rat/Burntwood, and Lower Nelson river systems. In the Split Lake area, overall mean abundances were 2,207 and 3,319 individuals/m² in Assean Lake and Split Lake (includes Split and Clark lakes, and the York Landing Arm of Split Lake), respectively. Assean Lake was more comparable to Leftrook Lake in that the hydrology of the watershed has not been affected by hydroelectric development; both have relatively shallow water depths; and aquatic macrophyte growth was dense in patches and beds were abundant in the shallow 'marshy' bays. However, the abundance of benthos in Assean Lake was notably lower than in Leftrook Lake; chironomids and fingernail clams were predominant in both waterbodies, but mayflies, stoneflies, and caddisflies (particularly mayflies) were relatively more common in Assean Lake while amphipods were more common in Leftrook Lake. Compared to other lakes in northern Manitoba, the abundance of benthos in Split Lake was higher. Similar to Assean Lake, the aquatic invertebrate community composition in Split Lake was similar to other northern Manitoba lakes, but mayflies, stoneflies, and caddisflies (particularly mayflies) contributed to a greater proportion of the community observed.

In the Keeyask area of the Nelson River, overall mean abundance was 3,539 individuals/m², which was comparable to the lower Churchill River (post-weir) and higher than some reaches of the Rat, Burntwood, and lower Nelson rivers that were investigated. Similar to other reaches in the study area, mayflies, stoneflies, and caddisflies (particularly mayflies and occasionally caddisflies) played a larger role in the benthic invertebrate community composition.

Construction of the Kettle GS in the early 1970s resulted in extensive flooding immediately upstream of the GS. Moose Nose Lake (north arm) and several other small lakes that previously drained into the Nelson River became continuous with the river to form Stephens Lake. The overall abundance of benthos in Stephens Lake (2,621 individuals/m²) was comparable to, if not slightly higher than, the abundances of benthos in selected lakes along the diversion route and the lower Nelson River. Chironomids, amphipods, aquatic earthworms, and snails contributed to the community of Stephens

Lake. Fingernail clams were relatively uncommon, but mayflies were an important component of the sediment-dwelling macroinvertebrate community.

4.5.3.2 Split Lake Area

4.5.3.2.1 Sediment-Dwelling Macroinvertebrates

Beak (1962) studied the benthic invertebrate (sediment-dwelling) community of the Split Lake watershed from 1958 to 1960 and reported that the mayflies *Hexagenia* spp. and *Ephemera* spp., and the amphipod *Pontoporeia affinis*, were found in comparatively large numbers in most of the samples collected at 14 stations on the Burntwood River. The number of chironomids, on the other hand, was unexpectedly low. During the summer of 1966, amphipods were most abundant in shallow water (less than 6.5 m deep) and were the most widely distributed group throughout Split Lake (Schlick (1968). In addition to amphipods, mayflies, chironomids, fingernail clams (Pisidiidae), and snails were also commonly observed.

During the MEMP in the late 1980s, benthic invertebrates were sampled at the mid-lake stations used for the collection of limnological data (Cann 1991). The composition of the benthic community varied year to year. When sediment-dwelling **meiofauna** [*e.g.*, nematodes (Nemata), copepods, cladocerans] were excluded from the analysis, seed shrimp (Ostracoda) were the most relatively abundant (comprised 27% of the community), followed by amphipods (22%), clams (19%), Diptera (primarily chironomids) (15%), snails (7%), and mayflies (5%).

As a part of the PPER, TCN Members indicated that mayflies disappeared from Split Lake after implementation of the CRD/LWR project (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). Whether this decline relates to hydroelectric development or other factors is not clear as Giberson *et al.* (1991, 1992) showed a dramatic decline in mayfly populations in Southern Indian Lake following hydroelectric development that was largely related to air temperature.

CRD/LWR resulted in increased TSS loading to Split Lake, and increased sedimentation in some regions of the lake, so it is likely that benthic invertebrate community composition within some reaches of the Burntwood River and Split Lake was altered by CRD/LWR. Although substantial recent benthic invertebrate data (1997–2004) are available for Split and Clark lakes, comparison of these data with those of studies conducted in the 1980s and before CRD/LWR are challenging because of differences in methods and approach.

The sediment-dwelling macroinvertebrate community (benthos) was quantitatively described for nine aquatic habitat types in the Split Lake area (Split and Clark lakes, including the York Landing Arm of Split Lake) between 1997 and 2004 (Table 4-23). Forty-four macroinvertebrate taxa were observed in the Split Lake area benthos between 2001 and 2004 (Table 4-21).

The mean overall sediment-dwelling macroinvertebrate abundance for aquatic habitats sampled in the Split Lake area was 3,319 individuals/m^2. The shallow water habitats supported a lower mean abundance of benthos (2,919 individuals/m^2) than those described for the deep water habitats (3,664 individuals/m^2). Within the shallow area, mean abundance of benthos was notably higher in the intermittently exposed portion (4,201 individuals/m^2) than in areas predominantly wetted (2,449 individuals/m^2). Shallow areas with aquatic plants harboured a greater abundance of benthos

(3,395 individuals/m^2) compared to areas devoid of vegetation (2,599 individuals/m^2). Mean total abundance for specific aquatic habitats ranged from 2,192 individuals/m^2 in the shallow, intermittently exposed aquatic habitat with low water velocity, soft mineral-based substrate, and aquatic plants present (S-IEZ-L-S-M-RV) to 5,025 individuals/m^2 in comparable habitat, but with standing water (S-IEZ-ST-S-M-RV) (Table 4-23, Figure 4-12). There was substantial variability in abundances within habitat types and among replicates from individual sites.

Within the shallow environment, the composition of the sediment-dwelling community differed among specific aquatic habitat types (Table 4-23, Figure 4-12). In areas with aquatic plants present, fingernail clams and non-biting midges tended to be the most common taxa, with fingernail clams predominant in plant beds with water movement (low water velocity) and midges in beds in standing water areas. Mayflies and snails contributed to a higher proportion of the community observed in beds found in predominantly wetted areas compared to beds in intermittently exposed areas. In deep aquatic habitats, molluscs (snails and fingernail clams), mayflies, and amphipods dominated the community. Snails, fingernail clams, mayflies, and amphipods were all relatively common in calm, deeper water areas. In deeper areas with water movement, fingernail clams overwhelmingly dominated the community (47%).

The benthos was quantitatively described for three aquatic habitat types in Assean Lake between 2001 and 2004 (Table 4-24). Forty macroinvertebrate taxa were observed in the Assean Lake benthos between 2001 and 2004 (Table 4-21). An historical water level record is not available for the Assean watershed so it was not possible to determine the extent of the intermittently exposed and predominantly wetted portions of aquatic habitat.

The mean overall sediment-dwelling macroinvertebrate abundance for aquatic habitats sampled in Assean Lake was 2,207 individuals/m^2. The shallow habitats supported a higher mean abundance of benthos (3,320 individuals/m^2) than the deep habitats (1,012 individuals/m^2). Shallow areas with aquatic plants harboured a greater abundance of benthos (4,217 individuals/m^2) in comparison to areas devoid of vegetation (1,851 individuals/m^2) (Table 4-24, Figure 4-13). There was substantial variability in abundances within habitat types and among replicates from individual sites.

Within the shallow environment, the composition of the benthos was comparable between specific aquatic habitat types (Table 4-24, Figure 4-13). Chironomids were overwhelmingly the most common taxa observed in areas with (52%) and without (46%) aquatic plants, followed by fingernail clams. Mayflies contributed to a higher proportion of the community observed in shallow water environments without aquatic plants compared to areas with plants. In the deep aquatic habitat, chironomids remained dominant; however, fingernail clams and mayflies contributed to a greater proportion of the community observed in comparison to the shallow habitat.

4.5.3.2.2 Plant-Dwelling Macroinvertebrates

Quantitative surveys of the plant-dwelling macroinvertebrate community were undertaken in Clark Lake in 2003 and 2004 as part of the environmental studies and the community was described for one shallow aquatic habitat type with aquatic plants (Table 4-25); twenty-five macroinvertebrate taxa were observed (Table 4-21).

Shallow areas with aquatic plants harboured relatively few plant-dwelling macroinvertebrates (181 individuals/m^2) (Table 4-25). Chironomids were the most common taxon observed (38%), but aquatic earthworms (29%) and snails (25%) also contributed to the community. Aquatic plants were of relatively low density in the areas sampled to describe plant-dwelling macroinvertebrates (see Appendix 4A, Section 4A.2.3.2.2 for explanation of sampling location selection). The predominant species were pondweeds (*Potamogeton* spp.) and two species more tolerant of periodic episodes of dewatering and ice scour stress, common spikerush, and vernal water-starwort (Table 4-26).

When the same type of aquatic habitat was sampled for benthos and plant-dwelling macroinvertebrates, a notably higher abundance of macroinvertebrates was observed in the sediments (2,316 individuals/m^2) than living on the plants (181 individuals/m^2). Chironomids dominated the community for both; however, mayflies and fingernail clams were more common in the sediments while snails and aquatic earthworms contributed more to the community in the plants.

4.5.3.2.3 Drifting Macroinvertebrates

The Assean River drifting invertebrate community consisted of 30 taxa in 2001 and 2002, which was comparable to other reaches of the study area for the same sampling period (Table 4-21). Aquatic insects were the most common invertebrates observed in the drift in both years with 17 taxa represented; crustaceans were also relatively common with four taxa observed (Appendix 4B).

4.5.3.3 Keeyask Area

No data or assessment of the effects of hydroelectric development on the aquatic macroinvertebrate community prior to 1997 in the reach of the Nelson River between Clark Lake and Stephens Lake were located in the published literature.

4.5.3.3.1 Sediment-Dwelling Macroinvertebrates

The sediment-dwelling macroinvertebrate community was quantitatively described for 11 aquatic habitat types in the Keeyask area (Nelson River mainstem, backwater inlets, and Gull Lake) between 1999 and 2004 (Table 4-27). Forty-three macroinvertebrate taxa were observed in the benthos between 1999 and 2004 (Table 4-21).

The mean overall sediment-dwelling macroinvertebrate abundance for aquatic habitats sampled in the Keeyask area was 3,539 individuals/m^2. The shallow water habitats supported a higher mean abundance of benthos (3,921 individuals/m^2) than those described for the deep (2,693 individuals/m^2). Within the shallow habitat, including backwater inlet habitat, mean abundance of benthos was notably higher in the intermittently exposed portion (5,059 individuals/m^2) than in areas predominantly wetted (1,874 individuals/m^2). Mainstem and backwater inlet shallow areas with aquatic plants harboured a greater abundance of benthos (4,505 individuals/m^2) in comparison to areas devoid of vegetation (2,754 individuals/m^2). Mean total abundance for specific aquatic habitats ranged between 917 individuals/m^2 in the deep areas with standing water and soft mineral-based substrate (D-ST-S-M-NP) and 5,900 individuals/m^2 in shallow, intermittently exposed aquatic habitat with standing water, soft mineral-based substrate, and aquatic plants present (S-IEZ-ST-S-M-RV) (Table 4-27, Figure 4-14). There was substantial variability in abundances within habitat types and among replicates from individual sites.

Within the shallow environment, the composition of the sediment-dwelling community differed among specific aquatic habitat types (Table 4-27, Figure 4-14). In areas with aquatic plants present, chironomids were the most commonly observed invertebrate, with the exception of plant beds in predominantly wetted areas with standing water, where mayflies (32%) and fingernail clams (24%) contributed to a greater proportion of the community. The composition of the benthos in shallow areas devoid of plants was more variable compared to shallow areas with plant beds. The intermittently exposed areas were dominated by amphipods (48%) in the backwater inlet habitat and chironomids (66%) in the mainstem. Mayflies (39%) were most common in predominantly wetted areas with standing water and fingernail clams (46%) in areas with water movement (low water velocity). In deep aquatic habitats, molluscs (snails and fingernail clams), mayflies, chironomids, and caddisflies dominated the community. Mayflies (38%), fingernail clams (25%), and chironomids (24%) were all relatively common in calm, deeper water areas. In deeper areas with water movement, snails and fingernail clams dominated the community and caddisflies (14%) were relatively common.

4.5.3.3.2 Plant-Dwelling Macroinvertebrates

Quantitative surveys of the plant-dwelling macroinvertebrate community were undertaken in the Keeyask area between 2001 and 2004 and the community was described for three shallow aquatic habitat types with aquatic plants present (Table 4-28). A total of 56 macroinvertebrate taxa were observed (Table 4-21).

Shallow areas with aquatic plants harboured relatively few plant-dwelling macroinvertebrates. Mean total abundance for specific aquatic habitats ranged from 367 individuals/m^2 in shallow aquatic habitat with standing water (S-ST-S-M-RV) to 600 individuals/m^2 in shallow aquatic habitat with low water velocity (S-L-S-M-RV) (Table 4-28). There was substantial variability in abundances within habitat types and among replicates from individual sites.

Shallow habitat with aquatic plants was dominated by snails, chironomids, aquatic earthworms, and amphipods. Snails and chironomids were common in all plant beds sampled. Aquatic earthworms contributed to a greater proportion of the community in intermittently exposed, backwater inlet habitat (26%) and predominantly wetted habitat with water movement (25%), whereas amphipods were more common in predominantly wetted habitat with standing water (17%). Aquatic plants were of relatively low to medium density in the areas sampled to describe plant-dwelling macroinvertebrates. Plants were most abundant in the intermittently exposed zone of the backwater inlets sampled. The community in this type of habitat was dominated by northern watermilfoil and was shared with other species including the more amphibious common spikerush. In shallow habitat that was predominantly wetted, the community was dominated by pondweeds, particularly *Potamogeton* spp. Common spike rush and star duckweed were much more common in shallow habitat with standing water than in those classified as having low water velocity (Table 4-29).

When the same type of aquatic habitat was sampled for benthos and plant-dwelling macroinvertebrates, there were four to ten times more macroinvertebrates observed in the sediments than living on the plants. Chironomids, aquatic earthworms, and amphipods were commonly found in both the sediments and associated with the plants; however, aquatic earthworms and amphipods tended to contribute more to the community in the plants, and mayflies and fingernail clams were additional important members of the sediment-dwelling community.

4.5.3.3.3 Drifting Macroinvertebrates

In riverine environments of the study area, a relatively large amount of macroinvertebrate biomass is transported downstream by flowing water. Drift traps were used to sample drifting macroinvertebrates in tributaries and extensive areas of rapids where sampling by other methods (*e.g.*, dredge or air lift sampler) was otherwise not feasible due to logistical (*e.g.*, water depths and water velocities too high for effective sampling) or safety concerns (*i.e.*, high flows immediately upstream and downstream of Gull Rapids).

Eighty-five macroinvertebrate taxa, including semi-aquatic and terrestrial insects, were captured in drift traps set throughout the Keeyask area (Nelson River mainstem, upstream of Birthday Rapids to downstream of Gull Rapids), including smaller tributaries to the Nelson River mainstem, between 2001 and 2004 (Table 4-21). Macroinvertebrates were identified to a lower taxonomic level in 2003 and 2004 and this resulted in a step-trend increase to the number of taxa observed in comparison to 2001 and 2002. Within the Nelson River mainstem, 83 taxa were observed, with greater diversity downstream of Birthday (65 taxa) and Gull rapids (69 taxa). Tributaries were only assessed in 2001and 2002 and 27 taxa were present; when mainstem sites were limited to 2001 and 2002 data, the total number of taxa present in the mainstem was only slightly higher than in the tributaries (Table 4-21).

In 2003 and 2004, aquatic insects (specifically mayflies, caddisflies, and chironomids) were the most abundant drifting invertebrates collected in drift traps, representing 86 to 98% of the total mean drift trap catch in the Keeyask area (Table 4-30). Mayflies dominated the drifting invertebrate community in the majority of aquatic habitats sampled, with the exception of downstream of Gull Rapids where dipterans (predominantly chironomids) and aquatic earthworms occasionally dominated the drift trap catch (Table 4-30).

Overall, the drift traps upstream of Gull Rapids (at the downstream end of Gull Lake) were the most productive in terms of mean drifting invertebrate density (50 individuals/100 m^3) within the study area, followed by traps downstream of Birthday Rapids (35 individuals/100 m^3), downstream of Gull Rapids (21 individuals/100 m^3), and upstream of Birthday Rapids (9 individuals/100 m^3) (Table 4-30). From this, it may be inferred that the majority of drifting invertebrates in the study area were produced by the Nelson River aquatic habitats between Birthday and Gull rapids, including Gull Lake, and within Birthday and Gull Rapids themselves. Production of drifting invertebrates from Gull Rapids is likely an important input to Stephens Lake; however, these rapids appear to produce overall fewer drifting invertebrates than does the Nelson River between Birthday Rapids and the downstream extent of Gull Lake. Drifting invertebrates are important as a food source for fish and therefore, their abundance and distribution helps define the importance of an aquatic area as feeding habitat. The drifting community is also important in terms of providing individuals to colonize downstream areas and thereby contributing to the composition and abundance of downstream communities. Overall, drifting invertebrate density tended to peak during the summer months (late July to early August).

4.5.3.4 Stephens Lake Area

4.5.3.4.1 Sediment-Dwelling Macroinvertebrates

Benthic invertebrate studies in Stephens Lake have focussed on the deeper water areas within the central portions of basins (Crowe 1973; Cann 1991). Crowe (1973) examined the pre-CRD/LWR benthic invertebrate community of the newly formed Stephens Lake (Kettle reservoir) in 1972 and found that chironomids were the most abundant benthic invertebrates, though aquatic earthworms, snails, and mayflies were also collected.

During the MEMP, benthic invertebrates were sampled at the mid-lake stations used for the collection of limnological data (Cann 1991). The composition of the benthic community varied year to year. When sediment-dwelling meiofauna (*e.g.*, nematodes, copepods, cladocerans) were excluded from the analysis, Diptera (primarily chironomids) were the most relatively abundant (comprised approximately 45% of the community), followed by aquatic earthworms (17%), amphipods (15%), seed shrimp (14%), clams and snails (6%), and mayflies (2%).

Elders from TCN have stated that they have noticed a decrease in the number of mayflies and that they feel that aquatic insects are emerging earlier, possibly due to climate change. They have also noticed other biological changes including waterfowl arriving earlier, moose breeding earlier, and bears breeding at different times (about a month earlier) (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

Although substantial recent benthic invertebrate data (2001–2006) are available for Stephens Lake, comparison of these data with those of studies conducted in the 1980s and before CRD/LWR are challenging because of differences in methods and approach.

The sediment-dwelling macroinvertebrate community was quantitatively described for nine aquatic habitat types in the Stephens Lake area between 2001 and 2006 (Table 4-31). Thirty-two macroinvertebrate taxa were observed in the benthos between 2001 and 2004 (Table 4-21).

The mean overall sediment-dwelling macroinvertebrate abundance for aquatic habitats sampled in the Stephens Lake area was 2,621 individuals/m^2. The shallow habitats supported a slightly higher mean abundance of benthos (2,739 individuals/m^2) than those described for the deep habitats (2,366 individuals/m^2). Within the shallow habitats, mean abundance of benthos was higher in the intermittently exposed portion (3,874 individuals/m^2) than in areas predominantly wetted (2,249 individuals/m^2). Shallow areas with aquatic plants harboured a slightly lower abundance of benthos (2,665 individuals/m^2) compared to areas devoid of vegetation (2,757 individuals/m^2). With respect to substrate composition, sediment-dwelling macroinvertebrates were most abundant in organic-based substrates in the shallow environment (4,318 individuals/m^2). Mean total abundance for specific aquatic habitats ranged from 1,611 individuals/m^2 in predominantly wetted shallow areas with standing water and soft mineral-based substrate that were devoid of aquatic plants (S-ST-S-M-NP) to 8,331 individuals/m^2 in intermittently exposed shallow areas with standing water and soft organic-based substrate that were devoid of aquatic plants (S-IEZ-ST-S-O-NP) (Table 4-31, Figure 4-15). There was substantial variability in abundances within habitat types and among replicates from individual sites.

Within the shallow aquatic environment, the composition of the sediment-dwelling community differed among specific aquatic habitat types (Table 4-31, Figure 4-15). In areas with aquatic plants present, chironomids were most commonly observed. Mayflies and amphipods contributed to a higher proportion of the community observed in beds found in predominantly wetted areas, particularly in areas with soft, organic-based substrate. Chironomids were also predominant in areas devoid of vegetation. In the shallow areas without plants, mayflies were more common in predominantly wetted areas and amphipods preferred organic sediments. In deep aquatic habitats, the community was dominated by chironomids, mayflies, and amphipods, which were all relatively common in deeper water areas with mineral-based sediments. However, in deeper areas with organic sediments, amphipods overwhelmingly dominated the community (68%).

Areas with Winter Dissolved Oxygen Depletion

Site-specific analysis was conducted on the sediment-dwelling community in two areas of Stephens Lake (Ross Wright and O'Neil bays) that were historically inundated. In these bays, pockets of low DO concentrations have been observed during the winter (under ice-cover), most notably over organic substrates. Concentrations below Manitoba water quality objectives for the protection of cool- and cold-water species have been observed in this area and in extreme instances anoxia has developed in some shallow areas (Section 2.4.2.5). These two areas were chosen as proxies for the post-impoundment Keeyask reservoir and were surveyed intensively in the fall of 2006 to describe the benthic community.

The shallow, standing water habitat with organic-based substrates in Ross Wright and O'Neil bays where DO depletion was observed during the previous winter supported a substantially higher mean abundance of benthos (6,426 individuals/m^2) than either a comparable habitat type that did not experience low DO conditions (1,930 individuals/m^2) or a habitat with silt/clay-based substrate and adequate DO (2,178 individuals/m^2) (Figure 4-16).

In the habitat with organic-based substrates that experienced DO depletion, chironomids overwhelmingly dominated the community (75%) (Figure 4-16). Other major groups of invertebrates, such as mayflies, amphipods, aquatic earthworms, and fingernail clams, were present, but at proportionately low densities. Mayflies contributed to a higher proportion of the community in habitats where winter DO depletion did not occur, with amphipods being proportionately more abundant in areas with organic-based substrate and aquatic earthworms in areas with silt/clay-based substrate (Figure 4-16).

4.5.3.4.2 Plant-Dwelling Macroinvertebrates

Quantitative surveys of the plant-dwelling macroinvertebrate community were undertaken in 2005 and 2006 (years with higher than average water levels; Section 3.3.2.3) in two areas (Ross Wright and O'Neil bays) of Stephens Lake that were historically inundated. These two areas were chosen to provide a proxy for the post-impoundment Keeyask reservoir. The plant-dwelling community was described for four shallow aquatic habitat types with aquatic plants present (Table 4-32). A total of 34 macroinvertebrate taxa were observed.

Shallow water areas with aquatic plants harboured relatively few plant-dwelling macroinvertebrates. Mean total abundance for specific aquatic habitats ranged from 90 individuals/m^2 in predominantly wetted shallow habitat with mineral-based sediments (S-ST-S-M-RV) to 859 individuals/m^2 in comparable intermittently exposed habitat (S-IEZ-ST-S-M-RV) (Table 4-32). There was substantial variability in abundances within habitat types and among replicates from individual sites.

Hydra, chironomids, aquatic earthworms, snails, and amphipods dominated the shallow water environment with aquatic plants. Hydra and chironomids were common in the majority of plant beds sampled. Aquatic earthworms contributed to a greater proportion of the community in intermittently exposed habitats, whereas snails and amphipods were more common in predominantly wetted habitat with organic-based sediments. Aquatic plants ranged from relatively low to high density in the areas sampled to describe plant-dwelling macroinvertebrates (Table 4-33). Aquatic plants were most abundant in the shallow areas sampled with soft, mineral-based sediments. The intermittently exposed portion of this type of habitat was dominated by Richardson's pondweed; however, northern watermilfoil was also relatively common and *Stukenia* spp. and aquatic moss species were also found. The community sampled in the predominantly wetted portion was almost exclusively Richardson's pondweed. In shallow habitat with soft, organic-based sediments, most typically observed at the terminal ends of inundated bays (*e.g.*, Ross Wright Bay), the community composition shifted and northern watermilfoil was predominant with water smartweed also being abundant in localized areas.

When the same type of aquatic habitat was sampled for both benthos and plant-dwelling macroinvertebrates, there were two and 27 times more macroinvertebrates observed in the sediments than living on the plants. Chironomids, snails, aquatic earthworms, and amphipods were commonly found in both the sediments and living on the plants. Chironomids and mayflies generally contributed more to the sediment-dwelling community in comparison to the plant-dwelling community. Hydra and clam shrimp were notably important components of the plant-dwelling community.

4.5.3.4.3 Drifting Macroinvertebrates

Forty macroinvertebrate taxa, including semi-aquatic and terrestrial insects, were captured in drift traps set downstream of the Kettle GS in 2003 and 2004 (Table 4-21). Aquatic insects (specifically mayflies, caddisflies, and chironomids) were the most abundant drifting invertebrates collected in drift traps downstream of the Kettle GS, representing 91% of the total mean drift trap catch (Table 4-30). Similar to drift traps downstream of Gull Rapids, dipterans (predominantly chironomids) dominated the drift trap community and mayflies were relatively less dense (Table 4-30).

Overall, the drift traps downstream of the Kettle GS (*i.e.*, Stephens Lake) were the least productive in terms of mean drifting invertebrate density (7 individuals/100 m^3) within the study area (Table 4-30). The relatively low mean drifting invertebrate density downstream of the Kettle GS may be the result of: drift traps being located approximately 1 km downstream of the GS structure, thereby potentially sampling the drifting invertebrates predominantly originating from this relatively short section of the river only rather than also from Stephens Lake, monthly average discharge (cubic metres per second [cms]) being lower in comparison to the Keeyask area, and/or the majority of drift traps being located in areas with relatively slower water velocities in comparison to those located in the Keeyask area. However, the lack of drifting invertebrate density information from immediately upstream of the Kettle GS in Stephens Lake makes it difficult to determine whether this low downstream density is contributed to by sampling location and/or a paucity of invertebrates originating from Stephens Lake and drifting through the Kettle GS. Similar to other drift trap locations along the Nelson River mainstem, drifting invertebrate density tended to peak during the summer months (late July to early August).

4.5.3.5 Access Road Area

The presence of aquatic invertebrates was described using kick net samples at five stream crossings along the proposed Keeyask GS access road, two on the north side of the Nelson River and three on the south side (Map 1-4). The construction of the north access road was assessed in the KIP EA. The current assessment considers the construction and operation of the south access road and the operation of the north access road stream crossings.

4.5.3.5.1 North Side

Seventeen aquatic invertebrate taxa were observed in samples at SC-1, an unnamed tributary of the South Moswakot River on the north side of the Nelson River. Aquatic insects were the most common invertebrates with 11 taxa represented; caddisflies and dipterans dominated the insect community with four and three taxa, respectively. As described in the KIP EA, this stream will be crossed by a culvert with riprap to stabilize the banks on either side. No alterations to habitat outside of the crossing location are expected.

The highest diversity of aquatic invertebrate taxa (33 taxa) was observed at SC-2. SC-2 crosses Looking Back Creek approximately 4 km upstream of Stephens Lake on the north side of the Nelson River. The crossing site is located in the lower reaches of the creek with 95% of the 126 km^2 drainage area above the crossing. Aquatic insects dominated the community sampled with 24 taxa represented. Mayflies and caddisflies were the most common insects, each with six taxa represented, followed by snails (four taxa), beetles (Coleoptera) (four taxa), and dipterans (three taxa). SC-2 is the second of only two ROW crossings where black flies (Simuliidae) were observed; they were also observed at SC-5. Aquatic habitat consisted entirely of run/glide habitat with a high amount and diversity of cover, including over stream vegetation, woody debris, cutbank, in stream vegetation, and boulder. The diversity of habitat and size of the stream likely contributed to the greater diversity of aquatic invertebrates observed in comparison to other stream crossings. As described in the KIP EA, this stream will be crossed by a clear span bridge with no effect on aquatic habitat.

4.5.3.5.2 South Side

Aquatic invertebrates from 22 taxa were identified from Gull Rapids Creek (SC-3), a small seasonal tributary of the Nelson River on the south side that provides drainage to bog and fen areas including a small lake upstream of the crossing. Aquatic insects dominated the community sampled with 14 taxa observed; dipterans and caddisflies were the most common.

The fewest aquatic invertebrate taxa (seven taxa) were observed from SC-4, a small unnamed south side tributary of Stephens Lake that provides drainage to bog and fen areas including a small lake upstream of the crossing. Three aquatic insect and two Annelida (aquatic earthworms and leeches) taxa were represented in the samples. SC-5 was only assessed at the crossing site, which likely contributed to the relatively low number of taxa observed.

Aquatic invertebrates from 13 taxa were identified in kick net samples from Gillrat Lake Creek (SC-5), a relatively small first-order stream, with virtually the entire watershed located upstream of the crossing location. The stream drains Gillrat Lake and flows into Stephens Lake. Aquatic insects dominated the community sampled with nine taxa represented; caddisflies and dipterans were the most common. SC-6 is one of only two ROW crossings where black flies were observed. Aquatic habitat was diverse in this small creek, consisting primarily of runs with lesser amounts of pools and riffles. Black fly larvae and pupae are found wherever there is permanent or semi-permanent running water, which the larval and pupal stages require for their development (Peterson 1996).

4.5.3.6 Current Trends

Historic benthic invertebrate (sediment-dwelling) community data were located for Split and Stephens lakes; there are no historic data for the Keeyask area with which to compare in order to discern long-term trends. The Split and Stephens lakes data were collected pre-CRD/LWR (late 1950s and 1960s in Split Lake, and early 1970s in Stephens Lake) and in the late 1980s as part of the MEMP. Comparison of these data sets with benthic invertebrate data collected for the Project with the specific purpose of assessing current trends is limited for the following reasons:

- There were differences in approach (*e.g.*, surveys were conducted at different times/seasons of the year) and sampling methods (*e.g.*, different sampling, identification, and enumeration techniques were used in each of the studies or not adequately reported so that methods are unclear) employed.
- The short time span of each survey (one to three years) was not adequate to account for normal year-to-year variability in abundance.
- Benthic invertebrate abundance and composition varied considerably within waterbodies and among study years.

However, qualitative comparisons of benthic invertebrate data over time are presented.

Generally, in Split and Stephens lakes over time, most benthic invertebrate organisms occurred throughout the lake; the difference among aquatic habitats was with respect to the relative abundance of the various invertebrate groups.

The benthic invertebrate community in Split Lake during summer sampling in the late 1950s and 1960s consisted of amphipods, mayflies, chironomids, fingernail clams, and snails. These invertebrate groups were also collected during the present study, though the order of relative abundance was different. Chironomids dominated the shallow water community of the present study; however, mayflies, fingernail clams, snails, and aquatic earthworms were also relatively abundant. Snails, fingernail clams, mayflies, and amphipods were all relatively common in deeper water areas of Split Lake. The composition of the benthic community varied year to year during the MEMP in the late 1980s, but was dominated by nematodes, with amphipods and fingernail clams being of secondary importance. A relatively large number of nematodes was recorded in comparison to the pre-CRD/LWR results and those of the present study, due in part to the smaller mesh size (400 μm) used to sieve the benthic samples, which would have selectively retained more of the small nematodes. When nematodes were excluded from the analysis, seed shrimp, amphipods, and clams dominated the community. The relatively low abundance of mayflies in the late 1980s has been related to the implementation of CRD/LWR by local knowledge. Whether this temporary decline relates to hydroelectric development or other factors is unclear, though, as Giberson *et al.* (1991, 1992) showed a dramatic decline in mayfly populations in Southern Indian Lake following hydroelectric development that was strongly correlated with air temperatures during the summer period, suggesting that weather, rather than hydroelectric development, was largely responsible for controlling the mayfly population abundance.

In the early 1970s (pre-CRD/LWR), the benthic invertebrate community in deep water areas of Stephens Lake was dominated by chironomids; aquatic earthworms, snails, and mayflies were also present. These invertebrate groups were also observed during the present study. In shallow and deep aquatic habitats, chironomids, mayflies, and amphipods dominated the community. However, there were differences among specific types of aquatic habitat with respect to the relative abundance of the various invertebrate groups. As for Split Lake, the composition of the benthic community varied year to year during the MEMP in the late 1980s. When the meiofauna (*e.g.*, nematodes, copepods, cladocerans selectively retained by the smaller mesh size) were excluded from the analysis, chironomids dominated the community at the mid-lake stations, followed by aquatic earthworms, amphipods, and seed shrimp; mayflies only comprised 2% of the community. Elders from TCN have noted the relatively low abundance of mayflies in the late 1980s and they also felt that aquatic insects were emerging earlier, possibly due to climate change.

4.5.4 Project Effects, Mitigation and Monitoring

4.5.4.1 Construction Period

The following section considers potential effects related to the construction of the GS and south access road, and operation of the construction camp and north and south access roads during the construction period. The construction of the north access road was assessed in the KIP EA (Keeyask Hydropower Partnership Ltd. 2009).

An assessment of potential Project effects on the aquatic macroinvertebrate community during the construction period is based on the assessment of construction-related effects to surface water quality (Section 2.5.1, Table 2-12), physical attributes of aquatic habitat (Section 3.4.1), and aquatic macrophytes (Section 4.3.4.1). The primary potential effect(s) on aquatic macroinvertebrates is related to inputs affecting water quality, such as increases in TSS concentrations and related variables (*i.e.*, turbidity) due to in-stream activities (*e.g.*, cofferdam placement and removal, river impoundment and diversion) and inputs or construction activities that affect DO concentrations in the lower Nelson River. Predicted increases in TSS will alter downstream substrate due to sedimentation, and this could influence aquatic macroinvertebrates in affected areas. Cofferdam placement and dewatering of the area within cofferdams would affect any aquatic macroinvertebrates in the immediate vicinity of any works (the majority of aquatic habitat affected during construction will also affected by the permanent works; some construction works will remain in place and be submerged during impoundment). The aquatic macroinvertebrate community (*i.e.*, plant-dwelling macroinvertebrates) would respond to any changes in aquatic macrophytes as a result of inputs affecting water quality and sedimentation. Additionally, some aquatic habitat disruption will occur during construction of stream crossings to accommodate the south access road and may affect aquatic macroinvertebrates at the crossings. It is expected that construction effects (*e.g.*, inputs affecting water quality) will be managed through appropriate mitigation measures (Section 2.5.1; Section 3.4.1), thereby reducing the duration and magnitude of any construction-related effects on the aquatic macroinvertebrate community.

4.5.4.1.1 Upstream of the Outlet of Clark Lake

No construction-related effects on the aquatic macroinvertebrate community are expected upstream of the outlet of Clark Lake as there are no linkages between Project construction and surface water quality (Section 2.5.1) or aquatic habitat (Section 3.4.1) in Split, Assean, or Clark lakes.

4.5.4.1.2 Downstream of the Outlet of Clark Lake

The following sub-sections present the assessment of potential effects of construction activities on the aquatic macroinvertebrate community in the Keeyask area and downstream.

Changes to Water Quality

Total Suspended Solids, Turbidity, and Water Clarity

The current variability of river flow results in variations in the concentrations of suspended sediments and their deposition in the study area. As a result, the current aquatic macroinvertebrate community should be able to withstand very short-term increases (*i.e.*, days to a few weeks) in suspended and benthic sediments with small, long-term negative effects. Overall, the activities with the greatest potential to increase TSS concentrations in the lower Nelson River during construction of the GS are related to cofferdam placement and removal, and river impoundment and diversion (Section 2.5.1.1). Prolonged (*i.e.*, months), low to moderate increases in suspended fine sediments beyond the current range of concentrations may affect aquatic macroinvertebrates in the following ways: abrasion of/deposition on respiratory surfaces (*i.e.*, gills) (*e.g.*, a reduction in certain types of mayflies); interference of food intake for filter-feeders (*e.g.*, a reduction in certain types of caddisflies and fingernail clams); and increased rates of invertebrate drift due to changes in feeding efficiency and behaviour (*e.g.*, a temporary reduction in aquatic insect abundance in areas exposed to increases in TSS).

Generally, the construction and removal of cofferdams will generate an increase of less than 5 mg/L of TSS above background downstream of Gull Rapids (Section 2.5.1). Larger TSS increases are expected to be of relatively small magnitude and short duration. Peak levels are predicted to be up to 15 mg/L for one day or up to 7 mg/L for one month (Section 2.5.1). Drainage of surface runoff to the Nelson River will be controlled through a Drainage Management Plan (as described in the PD SV) to minimize the amount of sediment produced and the potential for sediment to enter watercourses. If the TSS concentration in water pumped out of cofferdam and excavation areas and in concrete wash water is greater than 25 mg/L the water will remain in a settling pond until it meets this TSS criterion before being discharged to the Nelson River. As the magnitude and duration of any increases in TSS are typically within the 30-day MWQSOG for PAL (an increase of 5 mg/L above background where background is less than or equal to 25 mg/L), the aquatic macroinvertebrate community may be negatively affected in this downstream environment (*i.e.*, small, undetectable reductions in aquatic macroinvertebrate distribution and/or abundance may occur in affected areas during the construction period). Additionally, these concentrations are well below levels that been described as being "low risk" to fish and their habitat (25–100 mg/L; DFO in Birtwell 1999).Under water Excavated Material Placement Areas (EMPAs) in the reservoir will be armoured and of limited elevation to prevent erosion by flowing water. In shallow areas of the reservoir, they will be placed in areas where they do not exacerbate the depletion of DO.

Dissolved Oxygen

During the latter stages of the Stage II Diversion, when water levels are increased to near full supply level, flooding of organic materials is expected to reduce DO concentrations in flooded areas (Section 2.5.1.2). Additionally, the earlier initiation of ice bridging upstream of Gull Rapids may cause upstream water levels to increase by 0.5–1.5 m during Stage I and Stage II Diversion in the event of a construction design flood. While these water level increases would remain within the range of water levels expected under a similar flow event during Project operation, this occurrence during construction may lead to DO depletion related to decomposition of flooded organic materials similar to that which would occur in the initial period post-impoundment (Section 2.5.1.2). These effects (*i.e.*, due to reservoir impoundment) are discussed in detail in the assessment of operation-related effects on surface water quality (Section 2.5.2.2) and aquatic macroinvertebrates (Section 4.5.4.2).

Metals and Contaminants (*e.g.*, Hydrocarbons)

Small amounts of metals will be introduced into the aquatic environment in association with construction activities that release sediments, as discussed in Section 2.5.1.6. However, given the proposed mitigation measures to manage sediment levels, these inputs are not expected to cause marked increases in metal levels and, consequently, will have no detectable effect on the aquatic macroinvertebrate community.

The presence and levels of hydrocarbons in the aquatic environment could potentially be affected by accidental spills or release of substances containing hydrocarbons (*e.g.*, diesel fuel, gasoline, lubricating oil, *etc.*). Other hazardous substances will also be used during the construction period. As described in Section 2.5.1.6, the release of significant quantities of hazardous substances to the aquatic environment as a result of accidental spills and releases is considered unlikely due to the development and implementation of good management practices.

Alteration and Destruction of Aquatic Habitat

Downstream Sedimentation

It is predicted that approximately 30 % of the additional sediment resulting from shore erosion during Stage I and II Diversions will be deposited in Stephens Lake before it reaches the Kettle GS (Section 2.5.1.1.3); most of the deposition is expected to occur near the entrance of Stephens Lake, downstream of Gull Rapids (Section 3.4.1.5). This additional sedimentation could negatively influence the aquatic macroinvertebrate community in the affected area depending on the size of sediment particles, the type of substrate (*e.g.*, greater negative potential if coarser substrate affected), the spatial extent (*e.g.*, greater negative potential as percent surface cover increases), and depth of deposited sediments (*e.g.*, greater negative potential if depth of sediments exceeds 5 cm), the rate of deposition, and if deposited sediments are stable or transient (*e.g.*, washed away with the next higher flow event). Cumulative sediment input from all construction sources, over a four-year period for instream work, is expected to result in a depth of deposited sediments less than 0.6 cm (very low rate of deposition) through the south arm of Stephens Lake. Deposited material will likely be a combination of silt, sand, and coarser material, and is unlikely to be remobilized during the GS operating period. A small increase in sediment may reduce population densities because of a reduction in habitat space (*e.g.*, an increase in substrate embeddedness); however, community structure (*i.e.*, community composition) may not change (Lenat *et al.* 1979). An increase in the volume of fine sediments may favour certain taxa over others; for example, some chironomids use fine sediments in the construction of cases and tubes, aquatic earthworms and fingernail clams are often associated with fine sediment, and specific mayflies (*Hexagenia limbata*) are more common in silt deposits, into which they burrow. Some types of mayflies, stoneflies, and caddisflies are often particularly affected by sedimentation due the inhibitory effects of fine sediments on attached algae as a food source, density of prey items, available oxygen for respiration, and interstitial space (*i.e.*, spaces between coarser particles) for refuge. When the substrate is degraded by fine sediment, there will be a point where the macroinvertebrate community will become less diverse and numerically dominated by fine sediment tolerant taxa, such as chironomids. However, based on the low rate of deposition and resultant minimal depth of deposited sediments over the four years of instream work, downstream sedimentation is not expected to have a measurable effect on the aquatic macroinvertebrate community during the construction period.

Loss of Aquatic Habitat in Footprint of Supporting Infrastructure

The construction of cofferdams will result in the temporary loss of aquatic habitat in Gull Rapids (Section 3.4.1.1). The benthic macroinvertebrate community occupying hard substrate, faster-flowing water aquatic habitat types will be directly affected by the loss of habitat due to either cofferdam footprint or dewatered area; additionally, starting during construction, there would be a site-specific decrease in the production of drifting invertebrates (predominantly aquatic insects) from these areas.

Starting during construction and continuing through operation, the positioning of EMPAs within the reservoir may increase aquatic habitat diversity in affected areas.

Aquatic Macrophytes

Predicted moderate reductions in the production of drifting, non-vascular plant (filamentous algae) biomass originating from Gull Rapids during construction may further negatively affect aquatic macroinvertebrate distribution and/or abundance due to a decrease in available habitat.

4.5.4.1.3 South Access Road Stream Crossings

No response is expected due to the input of sediments into natural watercourses as effects to surface water quality are predicted to be small due to the application of various mitigation measures (Section 2.5.1.7).

At each of the three stream crossings, the footprint of the road, combined with the installation of the culvert(s), will result in several changes in aquatic habitat (Section 3.4.1.6). A portion of the benthic macroinvertebrate community that occurs at proposed south access road stream crossings will be lost due to infilling of a relatively small amount of aquatic habitat at crossings. Potential effects to benthic invertebrates at the stream crossings will be addressed by following the "Manitoba Stream Crossing Guidelines for Protection of Fish and Fish Habitat" and other pertinent regulatory guidelines.

4.5.4.1.4 Net Effects of Construction with Mitigation

Collectively, the above assessment points to the potential for small decreases in aquatic macroinvertebrate distribution and/or abundance during the construction period. Changes in distribution and/or abundance would occur over the short- to long-term downstream of the outlet of Clark Lake. Additionally, the construction of cofferdams will result in a moderate reduction in the production of drifting invertebrates (predominantly larval insects) originating from Gull Rapids. The decrease in drifting invertebrates is expected to be long-term due to effects continuing through the operation period.

Access road stream crossings will result in the permanent loss of benthic macroinvertebrates in the immediate footprint of the access road and culvert(s).

4.5.4.2 Operation Period

The aquatic macroinvertebrates are typically a diverse assemblage, and are adapted to the range of substrate types and water flow regimes (*e.g.*, fast-flowing rivers, sheltered bays in lakes with no discernable flow) found in the aquatic environment. The impoundment of rivers often produces large changes in the macroinvertebrate community, both within the reservoir and downstream of the GS. Generally, changes within reservoirs are consistent with organic enrichment and a transition from riverine to lacustrine-type habitat (Henriques 1987). In regulated systems, such as the reservoir areas of hydroelectric generating stations, the increased frequency of water level fluctuations tends to reduce macroinvertebrate abundance in shallow, regularly exposed areas along the shoreline (*i.e.*, upper littoral zone) (*e.g.*, Hunt and Jones 1972).

4.5.4.2.1 Upstream of the Outlet of Clark Lake

No response is expected. Selection of a 159 m ASL reservoir elevation instead of a higher elevation will avoid Project-related effects as Split Lake area is beyond the upstream extent of the expected hydraulic zone of influence.

4.5.4.2.2 Outlet of Clark Lake to the Keeyask Generating Station

Potential Project Effects and Proposed Mitigation

Operation-related pathways that were assessed for potential effects to aquatic macroinvertebrate distribution, abundance, and/or community composition included: flooding (loss of existing habitats, creation of new habitats); reduction in medium and high water velocity habitats; conversion of existing hard substrates (gravel, cobble, boulder) to silt/clay due to sedimentation in Gull Lake; increase in the frequency of water level fluctuations; conversion of tributary habitat to bays; and, a reduction in the extent and severity of ice scour (Section 3.4.2.2); and, changes in surface water quality in off-current areas, in particular bays (Section 2.5.2.2). Summaries of predicted responses of aquatic macroinvertebrates to changes resulting from the operation of the Project are presented in Figure 4-5. Where feasible, the effects of these pathways were considered using modelling exercises (quantification of potential effects), empirical information from Stephens Lake and other reservoirs in northern Manitoba, reservoirs in other northern temperate areas, and the scientific literature.

Assessment of Operation-Related Effects

Modelling Approach

Post-impoundment, the newly flooded areas created and the expansion of deep-water habitat (standing-low water velocity, soft, mineral-based substrates) as water levels increase are expected to provide suitable habitat for benthic macroinvertebrates and result in an approximate area-wide, two- to three-fold increase in total benthic macroinvertebrate abundance (individuals/habitat type; Table 4-34). However, in the medium term (*i.e.*, 10–15 years post-impoundment), DO depletion in bays within the shallow, flooded areas, and potentially longer (*i.e.*, longer than 25 years) in highly isolated areas where organic substrates persist, is expected to limit invertebrate colonization to a few resilient groups (*e.g.*, chironomids). As aquatic plant beds are not expected to begin to develop in the downstream portion of the new reservoir (reaches 5-9A) until between 5 and 15 years after impoundment (Section 3.4.2.2; Table 4-16), the plant-dwelling macroinvertebrate community would also be mostly absent during this time (Table 4-35). Following impoundment, most groups of macroinvertebrates should be represented, but their relative importance will likely be influenced by the extent and frequency of water level fluctuation, DO concentrations, food availability, and substrate suitability (*i.e.*, preferences of groups).

Model results indicate an expected area-wide, large increase (a direct gain of approximately 3,363 ha) in aquatic habitat in the long-term when the reservoir is at MOL (Appendix 3D, Table 3D-1). A larger increase (a direct gain of approximately 5,176 ha) is modelled for the reservoir at FSL; however, a portion of the upper littoral habitat has the potential to be degraded in quality (*i.e.*, increased potential for desiccation and freezing) for both sediment- and plant-dwelling macroinvertebrates due to increased frequency of water level fluctuations, the extent of which will depend on water level cycling at FSL (Appendix 3D; Appendix 4A, Table 4A-1 and Table 4A-2). The quality of the upper littoral zone will also be influenced by the type of substrate affected by water level fluctuation; mineral-based substrate tends to freeze solid to some depth (degraded quality for benthic macroinvertebrates), whereas organic-based substrate typically freezes only at the surface, if at all (better quality for benthic macroinvertebrates) (Koskenniemi 1994). When the GS is operating in peaking mode, water levels in the 19 km section of the reservoir upstream of the powerhouse could fluctuate by as much as 1.0 m per day (excluding wind effects); the magnitude of water level variation would diminish near the upstream boundary of the hydraulic zone of influence (Section 3.4.2.2). Additionally, three reservoir zones are anticipated post-impoundment: riverine (higher flow and lower residence time); transitional (reduced flow and increased residence time); and lacustrine (low flow and greatest residence time).

An area-wide, large increase in the abundance of benthic macroinvertebrates (three- to four-fold) is expected in the long-term in response to the increased availability of aquatic habitat (Table 4-34). However, estimates for the plant-dwelling macroinvertebrate community vary between an approximate 78 and 48% reduction in abundance depending on water level cycling under the peaking mode of operation in the future reservoir (Table 4-35). Under a base loaded mode of operation (assuming no cycling of water level with the reservoir at FSL), the reduction in plant-dwelling macroinvertebrates is only 19% relative to the existing environment; this is due to the upper littoral habitat no longer being degraded in quality (*i.e.*, all habitat is permanently wetted) (Table 4-35). The increase in benthic macroinvertebrate abundance may be accompanied by a change in the community composition from that typical of riverine aquatic habitat to one more characteristic of slower-flowing water (*i.e.*, resembling portions of Stephens Lake). However, a community characteristic of existing faster waters will not disappear entirely as there will likely be a longitudinal variation in the benthic macroinvertebrate community reflecting the change from more riverine (upper portion of reservoir), to transitional (middle portion), to lacustrine (lower reservoir) (Northcote and Atagi 1997).

Flooding and peat disintegration are expected to cause decreases in DO concentrations in portions of shallow, flooded bays of the reservoir with poor mixing and long water residence times in the open water and ice-cover seasons (Section 2.5.2.2). The effects are expected to be medium term in duration (*i.e.*, the first 10–15 years post-impoundment), but in highly isolated shallow areas where organic substrates persist and/or where floating peat islands are present, the duration of effects may be longer (*i.e.*, longer than 25 years). The majority of the reservoir is expected to remain well-oxygenated throughout the year due to high water volumes/flows and short water residence times. The area over which the most stringent PAL water quality objective (chronic objective of 6.5 mg/L) is expected to be met in summer would vary according to the mode of operation (water level fluctuations) and wind speeds, but is expected to include the mainstem of the reservoir (including the immediate reservoir near the GS) and substantial portions of the flooded bays. Localized depletion of oxygen may occur where substantive areas of peat islands possibly will accumulate, particularly in shallow, flooded areas. Greater effects to DO in the Keeyask area will occur in winter, where a larger area will be affected, the magnitude of DO depletion will be greatest, and the duration of the effects would be longest. In winter, the area over which the most stringent PAL water quality objectives would be met in the reservoir is estimated as 62–69 km^2 (approximately 66–74% of the total reservoir area), depending on mode of operation. Anoxic and hypoxic conditions are expected to develop in shallow areas over flooded terrestrial habitat with limited mixing with the mainstem during the ice-cover season. As the ice-cover season is long in the Aquatic Environment Study Area, these low DO conditions are expected to occur for a number of months. Most invertebrate taxa tolerate all but very low DO levels (less than or equal to 10% saturation); however effects of low DO concentrations are typically observed at concentrations of less than or equal to 5–6.5 mg/L (Dauer 1993; Lowell and Culp 1999; Chambers *et al.* 2000; Dunnigan *et al.* 2004). Of the insects, mayflies demonstrate the highest sensitivity to low DO conditions [lethal effects observed at less than or equal to 20% saturation (Dauer 1993) or less than 1 mg/L (Winter *et al.* 1996)], while chironomids are more tolerant (Connolly *et al.* 2004). Initially, lower invertebrate biomass, abundance (particularly mayflies, stoneflies, and caddisflies, collectively referred to as EPT), and richness (particularly EPT) are anticipated in areas with poor DO conditions, with a community dominated by chironomids. A summer DO and water temperature regime of greater than or equal to 8 mg/L and greater than or equal to 10°C is adequate to sustain mayfly nymphs without limiting their survival. A reduction in DO at a high temperature would be more harmful to nymphs than the same degree of hypoxia at a lower temperature (Winter *et al.* 1996). During periods of ice-cover, water temperatures typically approach 0°C; such low temperatures may reduce invertebrate metabolism (and thus, oxygen demand) sufficiently to somewhat lessen the negative impacts of low DO concentrations during the winter (Lowell and Culp 1999).

The reduction in fast water (high velocity) and hard substrate at rapids due to flooding (up to 60% reduction in high velocity, hard substrate habitat; Appendix 3D, Table 3D-1) will result in a reduction in the abundance of macroinvertebrates favouring this type of aquatic habitat and likely contribute to a decline in the production of drifting invertebrates from within Birthday Rapids. Although water velocity is being reduced and water depth is increasing in areas of the reservoir, water flow is expected to be adequate (relatively short water residence time; low to high velocity aquatic habitat present) through the mainstem to produce and maintain a somewhat comparable density of drifting invertebrates (Section 3.4.2.2; Appendix 3D, Table 3D-1), particularly in the transitional and riverine environments of the middle and upper reservoir. In addition, there is typically a positive correlation between discharge (cms) and invertebrate drift (Svendsen *et al.* 2004) and future environment flows are expected to be quite similar to what has occurred over the past 30 years (existing environment) (PE SV, Section 4.4.2.1).

Information from Other Reservoirs

Presently, abundances of aquatic macroinvertebrates in the study area along the Nelson River are within the range observed for waterbodies along the Churchill, Rat/Burntwood, and Lower Nelson river systems. In the Keeyask area of the Nelson River, overall mean abundance was 3,539 individuals/m^2, which was comparable to the lower Churchill River (post-weir; 1999–2005) and the riverine and transitional portions of both the Long Spruce and Limestone reservoirs (2003 only; NSC 2012) and slightly higher than that observed in Stephens Lake. The abundance of macroinvertebrates (specifically, amphipods) in the lower portions of both the Long Spruce and Limestone reservoirs was higher (4,200 and 5,500 individuals/m^2, respectively), suggesting an increase in productivity due to impoundment (NSC 2012). Within the Keeyask area, chironomids, aquatic earthworms, and amphipods were commonly found in both the sediments and associated with the plants; however, aquatic earthworms and amphipods tended to contribute more to the community in the plants, and mayflies and fingernail clams were additional important members of the sediment-dwelling community. Similarly to the Keeyask area, chironomids, amphipods, oligochaetes, and snails contributed to the community of Stephens Lake; however, fingernail clams were relatively uncommon while mayflies were a considerably more important component of the sediment-dwelling macroinvertebrate community.

A more prolonged exposure to sediment deposition will result in a reduction in the abundance of certain types of invertebrates through the abrasion of respiratory surfaces (*i.e.*, gills) (*e.g.*, a reduction in certain types of mayflies), interference of food intake for filter-feeders (*e.g.*, a reduction in certain types of caddisflies and fingernail clams), and increased rates of invertebrate drift (*e.g.*, a reduction in aquatic insect abundance), but will also influence the structure of benthic macroinvertebrate communities. Numerous stream-dwelling mayflies, caddisflies, and stoneflies prefer coarse streambed substrates and are harmed by intrusions of fine sediments, while other groups of invertebrates (*e.g.*, chironomids) are more tolerant of fine sediment deposition onto existing gravel and cobble substrates.

Amphipods generally occur in greater numbers within slower-moving water and seem to prefer flooded terrestrial habitat; therefore, an increase in their abundance following impoundment could be expected, particularly in the more lacustrine downstream portion of the Keeyask reservoir. Greater abundances of chironomids and certain types of mayflies [primarily Ephemeridae (burrowing mayflies)] may be expected due to the establishment of silt or clay bottoms (preferred habitat of burrowing mayflies; Merritt and

Cummins 1996) in portions of the lower region of the reservoir where other conditions are also suitable (*e.g.*, adequate DO concentrations), as was observed in each of the lower Nelson River reservoirs following impoundment (NSC 2012).

Typically, significant variation in drifting invertebrate density occurs among stream reaches, within reaches, and over time. Drifting invertebrates may originate from the aquatic macroinvertebrate community in habitat immediately upstream of sampling locations, but also from habitat in the mainstem and tributaries considerable distances upstream (Shearer *et al.* 2003). Factors associated with variation in the magnitude of drift include water velocity, light (*i.e.*, daytime vs. nighttime), and substrate type. However, for most types of macroinvertebrates, the relative and interactive effects of these different factors are not well understood as most streams and rivers are made up of many habitat types that differ in terms of the above attributes and others (Baker and Hawkins 1990). In general, there appears to be a positive correlation between stream discharge (cms) and invertebrate drift (Svendsen *et al.* 2004). However, water velocity does not seem to describe a simple linear response, but rather velocity seems to limit the range of drift densities possible. At slow flows, a wide range of drift densities (relatively low to high) is possible, but with increasing water velocity, the highest observed drift densities decrease in magnitude. At any particular water velocity, significant variation typically occurs, most likely associated with other habitat attributes (*e.g.*, substrate type) (Baker and Hawkins 1990).

Overall, the drift traps upstream of Gull Rapids (at the downstream end of Gull Lake) were the most productive in terms of drifting invertebrate density within the study area, followed by traps downstream of Birthday Rapids, downstream of Gull Rapids, and upstream of Birthday Rapids. From this, it may be inferred that the majority of drifting invertebrates in the study area was produced by the Nelson River aquatic habitats between Birthday and Gull rapids, including Gull Lake. Relatively higher production of drifting invertebrates from within this portion of the Nelson River may be contributed to by the relatively high aquatic habitat diversity in comparison to rapids alone, the greater proportion of low-medium water velocity habitat (0.2–1.5 m/s), and substrate made up of proportionately more gravel and cobble (Map 3-8 and Map 3-14). In terms of water velocity suitability for benthic invertebrates in rivers, preferences (*i.e.*, optima) have been shown to occur between 0.9 m/s and 1.3 m/s, both of which are medium water velocities (0.5–1.5 m/s) (Jowett *et al.* 1991). Generally, in riverine environments invertebrates are more abundant (*i.e.*, show a preference) in shallow water and in gravel or coarser substrates (particularly mayflies, stoneflies, and caddisflies) (Minshall 1984, Jowett *et al.* 1991). Total benthic invertebrate production tends to be relatively low on extremely fine (*e.g.*, silt/clay, sand) and extremely coarse (*e.g.*, boulder, bedrock) substrates, while productivity is typically highest for substrate particles averaging 10 mm in diameter (*i.e.*, gravel, cobble) (Morin 1997). Post-impoundment, water flow is expected to be adequate (relatively short water residence time; low to high velocity aquatic habitat present) through the mainstem to produce and maintain a somewhat comparable density of drifting invertebrates, particularly in the transitional and riverine environments of the middle and upper reservoir.

Impoundment will flood several creek mouths (Portage, Two Goose, and Nap creeks) further upstream in the reservoir. Shallow riffle areas are known to be highly productive in terms of insect larvae (Scullion *et al.* 1982) and production declines at deeper water depth (Hynes 1970). In 2004 and 2005, drifting invertebrate density was typically highest near the mouths of small tributaries to the lower Nelson River (*i.e.*, downstream of the Limestone GS) in comparison to mainstem locations (Capar and Gill 2008, Gill and Chambers 2008). The amount of invertebrate production that would be lost from Portage, Two Goose, and Nap creeks after impoundment at Gull Rapids is difficult to assess without knowing the detailed gradient and substrate type of the potentially flooded riffles. Depending on the size of the substrate, stream beds at gradients greater than 2% may be unstable and support relatively few invertebrates (Cobb *et al.* 1992). Based on a coarse visual assessment of Figure 4.4-23, Figure 4.4-24, and Figure 4.4-25 in Section 4.4.2.2 of PE SV, it appears that there are relatively short sections of potentially flooded riffle habitat with gradients of greater than 2%; however, the majority of potentially affected riffle habitat looks to have gradients of less than 2%. Increasing water levels and decreasing velocities will reduce the production of insect larvae in low gradient (*i.e.*, less than 2%) riffles where productivity is expected to be relatively high; however, increasing water levels would provide more stable habitat in high gradient riffles.

A reduction in ice scour stress will increase the amount of aquatic habitat suitable for macroinvertebrates, possibly resulting in a small increase in the distribution and abundance of these organisms. However, available information suggests that disturbance of habitat induced by ice breakup and scour is temporary, with avoidance behaviour being suggested as one reason for the apparent resilience of some invertebrates (*e.g.*, larval insect nymphs) (Prowse and Culp 2003).

As was observed in Stephens Lake, shallow areas that experience low DO conditions may ultimately support a substantially higher mean abundance of benthic macroinvertebrates in the long-term post-impoundment if DO depletion continues to occur during the winter months. However, the community would continue to be dominated by chironomids with other major groups of invertebrates, such as mayflies, amphipods, aquatic earthworms, and fingernail clams, present, but at proportionately lower densities (Section 4.5.3.2). Chironomids and aquatic earthworms are also expected to be able tolerate the conditions of periodic exposure (desiccation, freezing) in the upper littoral zone as well as be able to rapidly take advantage of newly flooded terrestrial habitat in the short term (*i.e.*, first few years) following impoundment.

Local knowledge indicates that mayflies disappeared from Split Lake after CRD/LWR (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c). Whether this decline relates to hydroelectric development or other factors is not clear; Giberson *et al.* (1991, 1992) showed a dramatic decline in mayfly populations in Southern Indian Lake following hydroelectric development that was strongly correlated with air temperatures during the summer period, suggesting that weather, rather than hydroelectric development, was largely responsible for controlling the mayfly population abundance.

Elders from TCN have stated that they have noticed a decrease in the number of mayflies and have also stated that they feel that insects are emerging earlier, possibly due to climate change as they have also noticed other biological changes including waterfowl arriving earlier, moose breeding earlier, and bears breeding at different times (about a month earlier) (Split Lake Cree - Manitoba Hydro Joint Study Group 1996c).

4.5.4.2.3 Downstream of the Keeyask Generating Station

Operation-related pathways that were assessed for potential effects to aquatic macroinvertebrate distribution, abundance, and community composition included: alteration of flow patterns, and water velocities and depths, a reduction in the extent and severity of ice scour in the portion of the Nelson River to the inlet of Stephens Lake, the direct loss of aquatic habitat due to dewatering of Gull Rapids and the footprint of the GS structure, and a change in the density of drifting invertebrates entering from Gull Lake. Summaries of predicted responses of aquatic macroinvertebrates to changes resulting from the operation of the Project are presented in Figure 4-6. Where feasible, the effects of these pathways were considered using modelling exercises (quantification of potential effects), empirical information from Stephens Lake and other reservoirs in northern Manitoba, reservoirs in other northern temperate areas, and the scientific literature.

Macroinvertebrates are vulnerable to rapid diurnal changes in flow and regulated river reaches below generating stations, with erratic flow patterns, are typically characterized as having low total richness (*i.e.*, few taxa present) (Munn and Brusven 1991). Sudden increases in flow can cause considerable drift in response to increased sheer stress, thereby reducing benthic macroinvertebrate abundance (Layzer *et al.* 1989). The impact of high velocity water releases can also selectively influence the downstream macroinvertebrate community; small insect larvae and other invertebrates cannot tolerate high velocities and are often under-represented downstream of generating stations for this reason (De Jalon *et al.* 1994).

A reduction in ice scour stress will increase the amount of aquatic habitat suitable for macroinvertebrates, possibly resulting in a small increase in the distribution and abundance of these organisms. However, available information suggests that disturbance of habitat induced by ice breakup and scour is temporary, with avoidance behaviour being suggested as one reason for the apparent resilience of some invertebrates (*e.g.*, larval insect nymphs) (Prowse and Culp 2003).

The direct loss of aquatic habitat at Gull Rapids due to dewatering of the rapids and the footprint of the GS structure will likely result in a decrease in the production of drifting invertebrates from within Gull Rapids and the abundance of benthic macroinvertebrates typical of fast-flowing, hard substrate aquatic habitat. Although fast water habitat along the north shore and small river/rapids habitat on the south shore will be created, it is uncertain how the amount and type of new aquatic habitat will compare to that lost within Gull Rapids in terms of providing habitat for benthic macroinvertebrates favouring this type of environment and producing drifting invertebrates. Additionally, the GS itself will act as a physical barrier, thereby impeding or restricting the drift of aquatic invertebrates downstream to some extent and active upstream movements in localized areas. Analyzing the effect of a barrier on the movement of aquatic invertebrates is more complicated than doing so for relatively larger fish species. Marchant and Hehir (2002) reported a loss in the number of invertebrate taxa immediately downstream of 19 larger dams (greater than 15 m in height) in southeast Australia, which may have been contributed to by limited colonization through drift. The existence of a barrier effect on aquatic macroinvertebrates is likely influenced by the size and operational type of dam.

Post-impoundment, water flow is expected to be adequate (relatively short water residence time; low to high velocity aquatic habitat present) through the mainstem of the reservoir to produce and maintain a density of drifting invertebrates somewhat comparable to the existing environment, particularly in the transitional and riverine environments of the middle and upper reservoir. Production of drifting invertebrates from Gull Rapids contributes to the input of invertebrates to Stephens Lake; however, these rapids appear to produce overall fewer drifting invertebrates than does the Nelson River between

Birthday Rapids and the downstream extent of Gull Lake. Relatively lower production of drifting invertebrates from within Gull Rapids may be influenced by the greater proportion of high water velocity habitat (greater than 1.5 m/s) and substrate made up of predominantly cobble, boulder, and bedrock (Map 3-8 and Map 3-14).

The drifting macroinvertebrate community downstream of the Kettle GS was quantified in 2003 and 2004 to provide a proxy for assessing potential changes in production from specific areas (*i.e.*, Gull Rapids) associated with the proposed Keeyask hydroelectric development. Overall, the drift traps downstream of the Kettle GS (*i.e.*, downstream of Stephens Lake) were the least productive in terms of mean drifting invertebrate density within the study area. The relatively low mean drifting invertebrate density downstream of the Kettle GS may be the result of: drift traps being located approximately 1 km downstream of the GS structure, thereby potentially sampling the drifting invertebrates predominantly originating from this relatively short section of the river only rather than also from Stephens Lake; monthly average discharge (cms) of this portion of the Nelson River being lower in comparison to the Keeyask area; and/or the majority of drift traps being located in areas with relatively slower water velocities in comparison to those located in the Keeyask area. However, the lack of drifting invertebrate density information from immediately upstream of the Kettle GS within Stephens Lake makes it difficult to determine whether this low downstream density is contributed to by sampling location and/or a paucity of invertebrates originating from Stephens Lake and drifting through the Kettle GS.

Drifting invertebrate density in 2004 and 2005 was typically higher (approximately three to ten times) in traps located near rapids in the lower Nelson River (*i.e.*, downstream of the Limestone GS) in comparison to traps located in either the Limestone reservoir or tailrace area (Capar and Gill 2008, Gill and Chambers 2008). In 2004, drift traps located downstream of the Limestone GS at the downstream extent of the tailrace were marginally more productive in terms of mean drifting invertebrate density than those traps within the upstream reservoir; however, drift density was approximately half that observed downstream of the Kettle GS. Mayflies, followed by chironomids and caddisflies dominated the Limestone tailrace catch. Amphipods either dominated the catch in the upstream reservoir or were of similar drift density as mayflies; amphipods generally occur in greater numbers within slower-moving water of impounded areas in comparison to mainstem sites (NSC 2012).

4.5.4.2.4 Access Road Stream Crossings

Loss of aquatic macroinvertebrates due to the placement of the culvert and alteration due to the placement of riprap in the smaller streams will continue through the operating period. No incremental effects related to sediment inputs from erosion are expected due to the application of erosion control measures. No effects to aquatic macroinvertebrates in Looking Back Creek are expected.

4.5.4.2.5 Net Effects of Operation with Mitigation

The impoundment of the Nelson River at Gull Rapids will produce large changes in the aquatic macroinvertebrate community, both within the reservoir and the Nelson River immediately downstream of the GS. A large increase in the abundance of sediment-dwelling macroinvertebrates is expected in the reservoir in the long-term in response to the increased availability of aquatic habitat (creation of flooded areas and expansion of deep-water habitat as water levels increase). As aquatic vascular plants are not expected to begin to develop in the downstream portion of the reservoir until between 5 and 15 years after impoundment, plant-dwelling macroinvertebrates would be mostly absent from the reservoir during this time. Overall, there will be a reduction in the abundance of plant-dwelling macroinvertebrates in the reservoir, the extent of which will depend on the mode of operation of the GS.

The increase in benthic invertebrate abundance may be accompanied by a change in the community composition in the lower portions of the reservoir from that typical of riverine aquatic habitat to one more characteristic of slower-flowing water (*i.e.*, resembling portions of Stephens Lake). Flooding and peatland erosion/disintegration are expected to cause decreases in DO concentrations in localized areas (*i.e.*, in a small portion of the shallow, flooded bays of the reservoir characterized as having poor mixing and long water residence times) during the open water and ice-cover seasons. The effects are expected to be moderate-term in duration (*i.e.*, the first 10–15 years post-impoundment), but in highly isolated shallow areas where organic substrates persist and/or where floating peat islands are present, the duration of effects may be long-term (*i.e.*, longer than 25 years). Greater effects to DO in the reservoir will occur in winter, where a larger area will be affected, the magnitude of DO depletion will be greatest, and the duration of the effects would be longest. In winter, the area over which the most stringent PAL water quality objectives would be met in the reservoir is estimated as 62–69 km^2 (approximately 66–74% of the total reservoir area), depending on the mode of operation. Anoxic and hypoxic conditions are expected to develop in shallow areas over flooded terrestrial habitat with limited mixing with the mainstem during the ice-cover season. The low DO conditions are expected to limit invertebrate colonization to a few resilient groups (*e.g.*, chironomids) in the localized affected areas.

The reduction in fast water (high velocity) and hard substrate at rapids due to flooding, dewatering, and/or footprint of the GS, and conversion of tributary habitat to bays will result in a reduction in the abundance of macroinvertebrates favouring this type of aquatic habitat and likely contribute to a moderate, long-term decline in the production of drifting invertebrates (predominantly larval insects) from within Birthday and Gull rapids, and tributaries.

Access road stream crossings will result in the small loss of benthic invertebrates in the immediate footprint of the access road and culvert(s).

4.5.4.3 Residual Effects

4.5.4.3.1 Construction Period

The residual effects of construction on macroinvertebrates include losses in Gull Rapids where cofferdams are constructed and a potential reduction in drift downstream into Stephens Lake. This effect would be permanent.

4.5.4.3.2 Operation Period

The residual effects of operation on aquatic macroinvertebrates are:

- An overall increase in the total amount of benthic macroinvertebrates in the reservoir due to the doubling of aquatic habitat;
- A reduction in the amount of plant-dwelling macroinvertebrates; and

- A reduction in the amount of macroinvertebrates that inhabit rapids and a reduction in macroinvertebrate drift below the GS.

4.5.4.3.3 Summary of Residual Effects

The effects of the construction and operation of the Project on aquatic macroinvertebrates are expected to be moderate to large and long-term, and to occur over a small to medium geographic extent in the reservoir, at the GS site, and immediately downstream in Stephens Lake. Predicted changes involve both increases and decreases in macroinvertebrate abundance, depending on the specific area. Expected residual effects to the aquatic macroinvertebrate community in terms of distribution and/or abundance were assessed and are presented in Table 4-36A and Table 4-36B for the construction and operating periods, respectively.

The technical aquatic macroinvertebrate assessment is based on models, scientific literature and information collected from a proxy reservoir (*i.e.*, Stephens Lake,) and the overall certainty associated with the predictions is moderate to high. Overall, there is high certainty regarding the nature and direction of effects and the magnitude of effects predicted for the increase in availability of aquatic habitat for benthic macroinvertebrates (benthos) and reduction in aquatic habitat that produces drifting macroinvertebrates. Certainty regarding the magnitude of effects predicted for the colonization of flooded areas by both benthos and those macroinvertebrates that inhabit plant beds is moderate.

4.5.4.4 Environmental Monitoring and Follow-up

As described in Chapter 8 of the Response to EIS Guidelines, Environmental Monitoring Plans have been developed as part of the Environmental Protection Program for the Project. A comprehensive AEMP will be developed that specifically outlines monitoring to measure the effects of the Project on the aquatic environment, and discusses how results will be used as a basis for adaptive management. The AEMP will include monitoring of the aquatic macroinvertebrate community to verify the results of the aquatic macroinvertebrate assessment.

Aquatic macroinvertebrate community variables are not considered VECs from an environmental assessment perspective; however, as supporting variables for other AEMP components, benthic invertebrate community variables do provide important measurement endpoints indicating potential change within or outside the range of natural variability that may be attributed to the operation of the Project. Additionally, benthic invertebrates are commonly used as sentinels of environmental change because they are sedentary, respond relatively rapidly to environmental change, and are important components of the aquatic ecosystem.

The benthic invertebrate monitoring program would address environmental changes as a result of both the construction and operation phases of the Project. Construction monitoring would specifically address the biological effects of predicted increases in TSS as a result of instream work on the Nelson River, and would be designed to complement the water quality AEMP. Monitoring activities for the operation phase would focus on evaluating specific Project-related effects (SEM) identified in the EIS at selected representative sites to determine whether conclusions drawn in the EIS are valid (*e.g.*, a large increase in the abundance of aquatic macroinvertebrates is expected in the reservoir in the long term in response to the increased availability of aquatic habitat [creation of flooded areas and expansion of deep-water habitat as water levels increase]. The increase in benthic invertebrate abundance may be accompanied by a change in the community composition in the lower portions of the reservoir from that typical of riverine aquatic habitat to one more characteristic of slower-flowing water [*i.e.*, resembling portions of Stephens Lake]. In the relatively long term [*i.e.*, 10–15 years post-impoundment], DO depletion in bays within the shallow, flooded areas, and potentially longer [*i.e.*, greater than 30 years] in highly isolated areas where organic substrates persist, is expected to limit invertebrate colonization to a few resilient groups [*e.g.*, chironomids]). Aquatic macroinvertebrate community monitoring would be conducted annually during instream construction and for the first three years of operation; monitoring would then be conducted every three to five years for the first 20–30 years of operation, depending on results obtained.

Reports detailing the outcomes of monitoring programs will be prepared and submitted to MCWS and DFO, in compliance with the *Environment Act* and the *Fisheries Act*, respectively.

4.6 REFERENCES

4.6.1 Literature Cited

Baker A.D., and Hawkins, C.P. 1990. Patch-specific variation in drift density of Baetis. In Mayflies and stoneflies. Edited by I.C. Campbell. Kluwer Academic Publishers, Dordrecht, NL. 269-274 pp.

Baker, R., and Schneider, F. 1993. Development of the insect fauna in the Limestone Forebay following impoundment and comparison with undisturbed sites at the Long Spruce Forebay and the lower Nelson River mainstem. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 35 pp.

Beak, T.W. 1962. Technical report on three annual surveys of Burntwood River, Manitoba, taken prior to operation of the International Nickel Company plant at Thompson. T.W. Beak Consulting Biologist, Kingston, ON.

Bezte, C.L., and Kroeker, K. 2000. Water chemistry and phytoplankton data for the Rat/Burntwood and Footprint River Systems, Manitoba, 1999. Report #00-01. A Joint Environmental Studies report prepared for Nisichawayasihk Cree Nation-Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 74 pp.

Birtwell, I.K. 1999. The effects of sediment on fish and their habitat. Fisheries and Oceans Canada. Canadian Stock Assessment Secretariat Research Document 99/139.

Brittain J.E. and Eikeland, T.J. 1988. Invertebrate drift – a review. Hydrobiologia 166: 77-93 pp.

Bunn S.E., and Arthington, A.H. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30(4): 492-507 pp.

Burt, M. 2007. Aquatic macrophyte and associated epiphytic invertebrate data collected from the Long Spruce and Limestone forebays, Manitoba, 2004. Report #04-09. A draft report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 41 pp.

Burt, M., and Neufeld, L.J. 2007a. Zooplankton sampling results from the Long Spruce and Limestone Forebays, Manitoba, 2002 and 2004. Report #04-10. A draft report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 30 pp.

Burt, M.J., and Neufeld, L.J. 2007b. Long Spruce and Limestone Forebay benthic monitoring, fall 2003, with a comparison to historic data. Report #04-08. A draft report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 79 pp.

CCME (Canadian Council of Ministers of the Environment). 2004. Canadian water quality guidelines for the protection of aquatic life. Phosphorus: Canadian Guidance Framework for the Management of Freshwater Systems. In Canadian environmental quality guidelines. CCME, Winnipeg, MB.

Cann, R. 1991. Summary of benthos data from Rat, Threepoint, Cross, Sipiwesk, Split and Stephens lakes, 1987-1989. MS Rep. No. 91-05. Manitoba Department of Natural Resources, Winnipeg, MB. 65 pp.

Capar, L.N., and Gill, G.J. 2008. Lower Nelson River drifting invertebrate community investigations, 2004. Report # 04-07. A draft report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 78 pp.

Capar, L.N., Bernhardt, W.J., and MacDonald, J.E. 2006. Lower Churchill River Water Level Enhancement Weir Project Post-Project Monitoring: Assessment of invertebrate responses to operation of the Project Year VII, 2005. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 81 pp.

Chambers, P.A., Brown S., Culp J.M., Lowell, R.B., and Pietroniro, A. 2000. Dissolved oxygen decline in ice-covered rivers of northern Alberta and its effects on aquatic biota. Journal of Aquatic Ecosystem and Recovery 8: 27-38 pp.

Cobb, D.G., Galloway, T.D., and Flannagan, J.F. 1992. The effect on the Trichoptera of a stable riffle constructed in an unstable reach of Wilson Creek, Manitoba, Canada. *In* Proceedings of the Sixth International Symposium on Trichoptera, Lodz-Zakopane, Poland, 12-16 September, 1989. Adam Mickiewicz University Press, Poland. 81-88 pp.

Connolly, N.M., Crossland, M.R., and Pearson, R.G. 2004. Effect of low dissolved oxygen on survival, emergence, and drift of tropical stream macroinvertebrates. Journal of the North American Benthological Soceity 23: 251-270 pp.

Cooley, H.M., and Badiou, P. 2004a. Water chemistry and phytoplankton data for the Rat/Burntwood and Footprint River Systems, Manitoba, 2000. Report #04-01. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 178 pp.

Cooley, H.M., and Badiou, P. 2004b. Water chemistry, sediment chemistry, and phytoplankton data for the Rat/Burntwood and Footprint River Systems, Manitoba, 2001. Report #04-06. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 293 pp.

Cooley, P.M., and Dolce, L. 2008. Aquatic habitat utilization studies in Stephens Lake: macrophyte distribution and biomass, epiphytic invertebrates, and fish catch-per-unit-effort in flooded habitat. Report #06-08. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 62 pp.

Cooley, H.M., Badiou, P., and Shipley, E. 2003. Water chemistry and phytoplankton data for the Burntwood River System, Manitoba, 2002. Report #03-12. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 337 pp.

Crowe, J.M.E. 1973. Limnological investigations of Kettle Reservoir and the Nelson River above Kelsey. MS Rep. No. 73-06. Manitoba Department of Mines, Resources and Environment Management, Winnipeg, MB. 34 pp.

Dauer, D.M. 1993. Biological criteria, environmental health, and estuarine macrobenthic community structure. Marine Pollution Bulletin 26:249-257 pp.

De Jalon, D.G., Sanches, P., and Camargo, J.A. 1994. Downstream effects of a new hydropower impoundment on macrophyte, macroinvertebrate and fish communities. Regulated Rivers: Research and Management 9: 253-261 pp.

Dodds, W.K., Jones, J.R., and Welch, E.B. 1998. Suggested classification of stream trophic state: Distributions of temperate stream types by chlorophyll, total nitrogen, and phosphorus. Water Research 32(5): 1455-1462 pp.

Dunnigan, M., McEachern, P., Noton, L., and Kovats, Z.E. 2004. Effects of oxygen depletion on benthic macroinvertebrates in an ice-covered river. In Proceedings of the North American Benthological Society Annual Meeting, Vancouver, BC, 6-10 June, 2004.

Elliott, J.M. 1971. The distances traveled by drifting invertebrates in a lake district stream. Oecologia 6: 350-379 pp.

Fazakas C.R. 1999. Biological and environmental data from experimental gillnetting on Split Lake, Manitoba, August 1998. A report prepared for the Tataskweyak Environmental Monitoring Agency by North/South Consultants Inc. TEMA Data Report 99-03: ix + 101 pp.

Fazakas C.R., and Lawrence, M.J. 1998. Biological and environmental data from experimental gillnetting on Split Lake, Manitoba, August 1997. A report prepared for the Tataskweyak Environmental Monitoring Agency by North/South Consultants Inc. TEMA Data Report 98-03: ix + 105 pp.

Fazakas, C.R., and Zrum, L. 1999. Benthic invertebrate, sediment and water transparency data from under the ice at Split Lake, Manitoba, 1998. TEMA Data Report # 99-01. North/South Consultants Inc., Winnipeg, MB. 59 pp.

FLCN (Fox Lake Cree Nation). 2008 Draft. Preliminary sturgeon TK study.

Giberson, D.J., Rosenberg, D.M., and Wiens, A.P. 1991. Changes in abundance of burrowing mayflies in Southern Indian Lake - lessons for environmental monitoring. Ambio 20(3-4): 139-142 pp.

Giberson, D.J., Rosenberg, D.M., and Wiens, A.P. 1992. Long-term abundance patterns of *Hexagenia* (Ephemeroptera: Ephemeridae) in Southern Indian Lake, Manitoba: responses to weather and hydroelectric development. Canadian Technical Report of Fisheries and Aquatic Sciences 1837: 16 pp.

Gill, G.J., and Chambers, C. 2008. Invertebrate drift and plant biomass data in the lower Nelson River mainstem and associated tributaries, Manitoba, 2005. Report # 05-09. A draft report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 124 pp.

Green, D.J. 1990. Physical and chemical water quality data collected from the Rat-Burntwood and Nelson River systems, 1985-1989. MS Rep. No. 90-15. Manitoba Department of Natural Resources, Winnipeg, MB. 242 pp.

Hayeur, G. 2001. Summary of knowledge acquired in northern environments from 1970 to 2000. Hydro-Québec, Montréal, QC. 113 pp.

Hecky, R.E., and Guildford, S.J. 1984. Primary productivity of Southern Indian Lake before, during, and after impoundment and Churchill River Diversion. Canadian Journal of Fisheries and Aquatic Sciences 41: 591-604 pp.

Hecky, R.E., and Harper, R.J. 1974. Phytoplankton and primary productivity of the lower Churchill lakes, the Rat-Burntwood lakes, and the Nelson River lakes and reservoirs. The Lake Winnipeg, Churchill and Nelson Rivers Study Board technical report, Appendix 5, Volume 2, Section F. 39 pp.

Hecky, R.E., and Kilham, P. 1988. Nutrient limitation of phytoplankton in freshwater and marine environments: a review of recent evidence on the effects of enrichment. Limnology and Oceanography 33: 796-822.

Heinonen, P. 1980. Quantity and composition of phytoplankton in Finnish inland waters. Publications of the Water Research Institute, National Board of Waters, Finland 37: 1-91 pp.

Heinonen, P. 1982. On the annual variation of phytoplankton biomass in Finnish inland waters. Hydrobiologia 86: 29-31 pp.

Hellsten, S. 2000. Environmental factors and aquatic macrophytes in the littoral zone of regulated lakes – causes, consequences and possibilities to alleviate harmful effects. Academic Dissertation. Finnish Environment Institute, Hydrology and Water Management Division, Water Engineering and Ecotechnology Research Group, P.O. Box 413, FIN-90101 Oulu, Finland, and Department of Biology, University of Oulu, P.O. Box 3000, FIN-90401 Oulu, Finland.

Henriques, P.R. 1987. Aquatic biology and hydroelectric power development in New Zealand. Oxford University Press, Auckland, NZ. 280 pp.

Horne, B.D. 1996. Lower Nelson River Forebay monitoring program 1995, Year VII. North/South Consultants Inc., Winnipeg, MB. 43 pp.

Horne, B.D. 1997. Development of the invertebrate fauna in the Limestone Forebay 1996. North/South Consultants Inc., Winnipeg, MB. 36 pp.

Horne, A.J., and Goldman, C.R. 1994. Limnology. Second edition. McGraw-Hill Inc., New York, NY. 576 pp.

Hunt, P.C., and Jones, J.W. 1972. The effect of water level fluctuations on a littoral fauna. Journal of Fish Biology 4: 385-394 pp.

Hynes, H.B.N. 1970. The ecology of running waters. University of Toronto Press, Toronto, ON. 555 pp.

Invasive Species Council of Manitoba [online]. Available from www.invasivespeciesmanitoba.com [accessed March 2012].

Janusz, L. 1990a. Summary of phytoplankton data for the Nelson River and Rat-Burntwood lakes, 1989. MS Rep. No. 90-12. Manitoba Department of Natural Resources, Winnipeg, MB. 79 pp.

Janusz, L. 1990b. Summary of zooplankton data for the Nelson River and Rat-Burntwood lakes, 1986-89. MS Rep. No. 90-11. Manitoba Department of Natural Resources, Winnipeg, MB. 150 pp.

Janusz, L. 1990c. Summary of net plankton data and secchi transparency data for the Nelson River and Rat-Burntwood lakes, 1987-89. MS Rep. No. 90-09. Manitoba Department of Natural Resources, Winnipeg, MB. 64 pp.

Jowett, I.G., Richardson, J., Biggs, B.J.F., Hickey, C.W., and Quinn, J.M. 1991. Microhabitat preferences of benthic invertebrates and the development of generalised *Deleatidium* spp. habitat suitability curves, applied to four New Zealand rivers. New Zealand Journal of Marine and Freshwater Research 25:187-199 pp.

Kalff, J., and Knoechel, R. 1978. Phytoplankton and their dynamics in oligotrophic and eutrophic lakes. Annual Review of Ecology and Systematics 9: 475-495 pp.

Keeyask Hydropower Partnership Ltd. 2009. Keeyask Infrastructure Project environmental assessment report. July 2009.

Koskenniemi, E. 1994. Colonization, succession and environmental conditions of the macrozoobenthos in a regulated, polyhumic reservoir, Western Finland. Internationale Revue der gessamten Hydrobiologie 79: 521-555 pp.

Kroeker, K. 1999. Split Lake depth map. TEMA Data Report #99-02. A report prepared for the Tataskweyak Environmental Monitoring Agency by North/South Consultants Inc., Winnipeg, MB. 4 pp.

Lawrence, M.J., and Fazakas, C.R. 1997. Benthic invertebrate, sediment, and water transparency data from under the ice at Split Lake, Manitoba, January 1997. TEMA Data Report #97-01. A report prepared for the Tataskweyak Environmental Monitoring Agency by North/South Consultants Inc., Winnipeg, MB. 20 pp.

Lawrence M. J., and Cooley, M. 1999. Split Lake aquatic monitoring program. A report prepared for the Tataskweyak Environmental Monitoring Agency by North/South Consultants Inc., Winnipeg, MB. 52 pp.

Lawrence, M.J., Fazakas, C.R, Zrum, L., Bezte, C., and Bernhardt, W.J. 1999. The Split Lake aquatic ecosystem: a synthesis of Split Lake biological and environmental data, January 1997 to October 1998. A report prepared for the Tataskweyak Environmental Monitoring Agency by North/South Consultants Inc., Winnipeg, MB. 87 pp.

Layzer, J.B., Nehus, T.J., Pennington, W., Gore, J.A., and Nestler, J.M. 1989. Seasonal variation in the composition of drift below a peaking hydroelectric project. Regulated Rivers: Research and Management 3:305-317 pp.

Lenat, D.R., Penrose, D.L., and Eagleson, K.W. 1979. Biological evaluation of non-point source pollutants in North Carolina streams and rivers. Biological Series 102. North Carolina Department of Natural Resources and Community Development, Raleigh, NC.

Livingston, L. 1987. A checklist of phytoplankton genera for Nelson River and Rat-Burntwood lakes, 1980, 1981 and 1986. MS Rep. No. 87-14. Manitoba Department of Natural Resources, Winnipeg, MB. 20 pp.

Livingston, L. 1988. Summary of phytoplankton data for the Nelson River and Rat-Burntwood lakes, 1987. MS Rep. No. 88-04. Manitoba Department of Natural Resources, Winnipeg, MB. 163 pp.

Livingston, L. 1989. Summary of phytoplankton data for the Nelson River and Rat-Burntwood lakes, 1988. MS Rep. No. 89-07. Manitoba Department of Natural Resources, Winnipeg, MB. 25 pp.

Lowell, R.B., and Culp, J.M. 1999. Cumulative effects of multiple effluent and low dissolved oxygen stressors on mayflies at cold temperatures. Canadian Journal of Fisheries and Aquatic Sciences 56: 1624-1630 pp.

Manitoba Conservation Data Centre. 2012a. Ecoregion Search: Hayes River Upland. http://www.gov.mb.ca/conservation/cdc/ecoreg/hayesriver.html [accessed May 17, 2012].

Manitoba Conservation Data Centre. 2012b. Ecoregion Search: Churchill River Upland. http://www.gov.mb.ca/conservation/cdc/ecoreg/churchill.html [accessed May 17, 2012].

Manitoba Hydro and NCN (Nisichawayashik Cree Nation). 2003. Wuskwatim Generation Project environmental impact statement. Volumes 1-10.

Manitoba Water Stewardship. 2002. Water Quality Management Section, Manitoba Water Stewardship. 123 Main Street, Suite 160. Winnipeg, Manitoba. R3C 1A5.

Marchant, R. and Hehir, G. 2002. The use of AUSRIVAS predictive models to assess the response of lotic macroinvertebrates to dams in south-east Australia. Freshwater Biology 47: 1033-1050 pp.

McCartney, M.P., Sullivan, C., and Acreman, M.C. 2000. Ecosystem impacts of large dams. Thematic review II.1 prepared as an input to the World Commission on Dams. Cape Town, www.dams.org. 75 pp.

Merritt, R.W., and Cummins, K.W. 1996. An introduction to the aquatic insects of North America. Third edition. Kendall/Hunt Publishing Company, Dubuque, IW. 862 pp.

Minshall, G.W. 1984. Aquatic insect-substratum relationships. In The ecology of aquatic insects. Edited by Resh, V.H. and D.M. Rosenberg. Praeger Publishers, New York, NY. 358-400 pp.

Morin, A. 1997. Empirical models predicting population abundance and productivity in lotic systems. Journal of the North American Benthological Society 16: 319-337 pp.

Munn, M.D., and Brusven, M.A. 1991. Benthic invertebrate communities in nonregulated and regulated waters of the Clearwater River, Idaho, USA. Regulated Rivers: Research and Management 6: 1-11 pp.

Murphy K.J., Rorslett, B., and Springuel, I. 1990. Strategy analysis of submerged lake macrophyte communities: an international example. Aquatic Botany 36: 303-323 pp.

Nalcor Energy. 2009. Lower Churchill Hydroelectric Generation Project Environmental Impact Statement. Volume II, Part A, Bioiphysical Assessment. February 2009. 368 pp.

Nilsson C., and Keddy, P.A. 1988. Predictability of change in shoreline vegetation in a hydroelectric reservoir, northern Sweden. Canadian Journal of Fisheries and Aquatic Sciences 45:1896-1904 pp.

NSC (North/South Consultants Inc.). 2012. Limestone Generating Station: Aquatic Environment Monitoring Programs. A Synthesis of Results from 1985 to 2003. A report prepared for Manitoba Hydro. 192 pp.

Northcote, T.G., and Atagi, D.Y. 1997. Ecological interactions in the flooded littoral zone of reservoirs: the importance and role of submerged vegetation with special reference to fish, fish habitat, and fisheries in the Nechako Reservoir of British Columbia, Canada. Skeena Fisheries Report SK-111. Ministry of Environment, Lands, and Parks, Skeena Region, Fisheries Branch, Skeena, BC. 64 pp.

Nürnberg, G.K. 1996. Trophic state of clear and colored, soft- and hardwater lakes with special consideration of nutrients, anoxia, phytoplankton and fish. Lake and Reservoir. Management 8: 17-30 pp.

OECD (Organization for Economic Cooperation and Development). 1982. Eutrophication of waters. Monitoring, assessment and control. Final Report. OECD cooperative programme on monitoring of inland waters (eutrophication control). Environment directorate, OECD, Paris. 154 pp.

Paterson, M.J., Findlay, D., Beaty, K., Findlay, W., Schindler, E.U., Stainton, M., and McCullough, G. 1997. Changes in the planktonic food web of a new experimental reservoir. Canadian Journal of Fisheries and Aquatic Sciences 54: 1088-1102 pp.

Peterson, B.V. 1996. Simuliidae. In An introduction to the aquatic insects of North America. Edited by R.W. Merritt and K.W. Cummins. Third edition. Kendell-Hunt, Dubuque, IW. 591-634 pp.

Pienitz, R., and Vincent, W.F. 2000. Effects of climate change relative to ozone depletion on UV exposure in subarctic lakes. Nature 404: 484-487 pp.

Prowse, T.D., and Culp, J.M. 2003. Ice breakup: a neglected factor in river ecology. Canadian Journal of Civil Engineering 30: 128-144 pp.

Ramsey, D.J., Livingston, L., Hagenson, I., and Green, D.J. 1989. Evolution of limnological conditions in lakes of the Nelson and Rat-Burntwood river systems after Churchill River diversion and Lake Winnipeg regulation: an overview. MS Rep. No. 89-15. Manitoba Department of Natural Resources, Winnipeg, MB. 93 pp.

Rorslett B. 1988. An integrated approach to hydropower impact assessment. I. Environmental features of some Norwegian hydro-electric lakes. Hydrobiologia 164: 39-66 pp.

Schlick, R.O. 1968. A survey of Split Lake in 1966. MS Rep. No. 68-8. Manitoba Department of Mines and Natural Resources, Winnipeg, MB. 21 pp.

Schneider, F., and Baker, R. 1993. Summary of lower trophic level studies in the lower Nelson River, 1988 to 1992. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 44 pp.

Schneider -Vieira, F. 1994. Development of the invertebrate fauna in the Limestone Forebay 1993. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 34 pp.

Schneider -Vieira, F. 1996. Development of the invertebrate fauna in the Limestone Forebay 1994. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 26 pp.

Scullion, J., Parish, C.A., Morgan, N., and Edwards, R.W. 1982. Comparison of benthic macroinvertebrate fauna and substratum composition in riffles and pools in the impounded River Elan and the unregulated River Wye, mid-Wales. Journal of Freshwwater Biology 12: 579-595 pp.

Shearer, K.A., Hayes, J.W., and Stark, J.D. 2002. Temporal and spatial quantification of aquatic invertebrate drift in the Maruia River, South Island, New Zealand. New Zealand Journal of Marine and Freshwater Research 36: 529-536 pp.

Shearer, K.A., Stark, J.D., Hayes, J.W., and Young, R.G. 2003. Relationships between drifting and benthic invertebrates in three New Zealand rivers: implications for drift-feeding fish. New Zealand Journal of Marine and Freshwater Research 37: 809-820 pp.

Split Lake Cree - Manitoba Hydro Joint Study Group. 1996a. Analysis of change: Split Lake Cree Post Project Environmental Review. Split Lake Cree - Manitoba Hydro Joint Study Group; vol. 1 of 5.

Split Lake Cree - Manitoba Hydro Joint Study Group. 1996b. History and first order effects: Manitoba Hydro projects - Split Lake Cree Post Project Environmental Review. Split Lake Cree - Manitoba Hydro Joint Study Group; vol. 2 of 5.

Split Lake Cree - Manitoba Hydro Joint Study Group. 1996c. Environmental matrices: Summary of Manitoba Hydro impacts - Split Lake Cree Post Project Environmental Review. Support from William Kennedy Consultants Ltd. & InterGroup Consultants Ltd. Split Lake Cree - Manitoba Hydro Joint Study Group; vol. 3 of 5.

Stockner, J.G., Langston, A.R., and Wilson, G.A. 2001. The limnology of the Williston Reservoir. Peace/Williston Fish and Wildlife Compensation Program, Report No. 242. 51 pp.

Strange, N.E. 1990. Water quality data for sites affected by Churchill River Diversion and Lake Winnipeg Regulation, northern Manitoba, 1986-89. FEMP unpublished report. 98 pp.

Svendsen, C.R., Quinn, T., and Kolbe, D. 2004. Review of macroinvertebrate drift in lotic ecosystems. Final Report Manuscript, October 2004. A report prepared for Wildlife Research Program, Environmental and Safety Division, Seattle City Light. 92 pp.

TetrES Consultants Inc. and NSC. 1998. Kettle Generating Station: change in summer operations: initial environmental evaluation. TetrES Consultants Inc., Winnipeg, MB. 63 pp.

Wetzel, R.G. 1983. Limnology. Second edition. Saunders College Publishing, New York, NY. 769 pp.

Wetzel, R.G. 2001. Limnology. Lake and river ecosytems. Third Edition. Academic Press, New York, NY. 1006 pp.

Winter, A., Ciborowski, J.J.H., and Reynoldson, T.B. 1996. Effects of chronic hypoxia and reduced temperature on survival and growth of burrowing mayflies, *Hexagenia limbata* (Ephemeroptera: Ephemeridae). Canadian Journal of Fisheries and Aquatic Sciences 53: 1565-1571 pp.

Zrum, L., and Wyn, B. 2009. Benthic invertebrate baseline monitoring.: synthesis report 2004 and 2007. A report prepared for Wuskwatim Power Limited Partnership by North/South Consultants Inc., Winnipeg, MB. 67 pp.

Zrum, L., and Juliano, K.M. 2004. Benthic invertebrate data for the Rat/Burntwood River system, Manitoba, 1998 - 2001. Report #04-03. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 248 pp.

Zrum, L., and Neufeld, L.J. 2003a. Zooplankton, benthic invertebrate and sediment data for Notigi Lake, Manitoba, 1999-2001. Report #03-20. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 72 pp.

Zrum, L., and Neufeld, L.J. 2003b. Zooplankton, benthic invertebrate and sediment data for Threepoint Lake, Manitoba, 1998-2001. Report #03-22. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 90 pp.

Zrum, L., and Neufeld, L.J. 2003c. Zooplankton, benthic invertebrate and sediment data for Birch Tree Lake, Manitoba, 2000-2001. Report #03-25. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 60 pp.

Zrum, L., and Neufeld, L.J. 2003d. Zooplankton, benthic invertebrate and sediment data for Leftrook Lake, Manitoba, 1999-2001. Report #03-21. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 71 pp.

Zrum, L., Burt, M.J., and Neufeld, L.J. 2003. Zooplankton, benthic invertebrate and sediment data for Wapisu Lake, Manitoba, 1999-2001. Report #03-24. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 73 pp.

Zrum, L., and Kennedy, J. 2000. Development of the invertebrate fauna in the Limestone Forebay, 1999. A report prepared for Manitoba Hydro by North/South Consultants Inc., Winnipeg, MB. 29 pp.

TABLES, FIGURES, AND MAPS

Table 4-1: Summary of phytoplankton biomass (mg/m^3) by area and year for the Aquatic Environment Study Area: 1999–2002 open water seasons

Site	Year	Mean (±SE)[1]	Range	n[2]
Split Lake Area				
Split and Clark Lakes				
All sites	2001	499 (± 130)	57–2506	24
All sites	2002	852 (± 149)	199–3454	28
All sites	2001–2002	689 (± 102)	57–3457	52
Assean Lake				
All sites	2001	111 (± 38)	27–311	8
All sites	2002	355 (± 50)	130–572	8
All sites	2001–2002	233 (± 44)	27–572	16
Keeyask Area				
All sites	1999[3]	970 (± 98)	822–1154	3
All sites	2001	235 (± 27)	128–423	12
All sites	2002	722 (± 212)	212–1482	6
All sites	2001–2002	479 (± 89)	128–1482	21
Stephens Lake Area				
All sites	2001	256 (± 38)	157–448	8
All sites	2002	851 (± 250)	119–1910	8
All sites	2001–2002	553 (± 144)	119–1910	16

1. SE = standard error.
2. Number of samples collected per year.
3. Sites were only visited once (October) in 1999.

Table 4-2: Summary of chlorophyll *a* concentration (µg/L) by reach and year and trophic status based on chlorophyll *a* concentration for the Aquatic Environment Study Area: 1999–2004 open water seasons

Site	Year	Mean (± SE)[1]	Range	n[2]	Trophic Status	Application
Split Lake Area						
Burntwood River at Split Lake						
SPL1	2001–2004	4 (± 0.3)	<1–5	16	oligotrophic	stream[3]
Nelson River between Kelsey GS and Split Lake						
All sites	2001	9 (± 2.0)	4–13	4	oligotrophic	stream
All sites	2002	6 (± 0.6)	5–8	4		
All sites	2003	4 (± 1.1)	<1–6	4		
All sites	2004	5 (± 0.6)	2–9	16		
All sites	2001–2004	6 (± 0.5)	<1–13	28		
Split and Clark Lakes						
All sites	2001	7 (± 0.7)	2–15	28	mesotrophic	lake[4]
All sites	2002	5 (± 0.3)	1–8	32		
All sites	2003	5 (± 0.5)	<1–14	32		
All sites	2004	5 (± 0.4)	2–8	15		
All sites	2001–2004	6 (± 0.3)	<1–15	107		
Aiken River						
AK1	2002–2003	5 (± 1.2)	<1–10	8	oligotrophic	stream
Assean Lake						
All sites	2001	3 (± 0.5)	2–6	8	oligotrophic-mesotrophic	lake
All sites	2002	3 (± 0.4)	1–4	8		
All sites	2003	3 (± 0.3)	2–4	8		
All sites	2001–2003	3 (± 0.2)	1–6	24		
Keeyask Area						
Gull Lake						
All sites	1999[5]	2 (± 0)	2–2	2	mesotrophic	lake
All sites	2001	7 (± 1.0)	4–10	8		
All sites	2002	5 (± 0.4)	4–7	8		
All sites	2003	5 (± 0.6)	3–8	8		
All sites	1999–2003	6 (± 0.5)	2–10	26		
Nelson River						
All sites	1999[5]	12	-	1	oligotrophic	stream
All sites	2001	7 (± 1.1)	2–12	8		
All sites	2002	6 (± 0.4)	4–8	8		

Table 4-2: Summary of chlorophyll *a* concentration (µg/L) by reach and year and trophic status based on chlorophyll *a* concentration for the Aquatic Environment Study Area: 1999–2004 open water seasons

Site	Year	Mean (± SE)[1]	Range	n[2]	Trophic Status	Application
Nelson River (Continued)						
All sites	2003	4 (± 0.4)	2–8	16	Oligotrophic	stream
All sites	2004	6 (± 0.4)	3–8	16		
All sites	1999–2004	5 (± 0.3)	1–12	49		
Two Goose Creek						
TRIB1	2003–2004	1 (± 0.4)	<1–4	8	oligotrophic	stream
Portage Creek						
TRIB2	2003–2004	2 (± 0.5)	1–5	8	oligotrophic	stream
Rabbit Creek						
TRIB3	2003–2004	2 (± 0.4)	<1–4	8	oligotrophic	stream
Stephens Lake Area						
All sites	2001	8 (± 0.9)	4–12	8	mesotrophic	lake
All sites	2002	6 (± 1.1)	2–16	12		
All sites	2003	5 (± 0.5)	2–10	20		
All sites	2004	5 (± 0.4)	1–8	24		
All sites	2001–2004	5 (± 0.3)	1–16	64		

1. SE = standard error.
2. Number of samples collected per year.
3. Oligotrophic (<10 ug/L), mesotrophic (10-30), eutrophic (>30) (Dodds et al. 1998).
4. Oligotrophic (<2.5 ug/L). mesotrophic (2.5-8), eutrophic (8-25), hyper-eutrophic (>25) (OECD 1982)
   ultra-oligotrophic (0.01-0.5), oligotrophic (0.3-3). mesotrophic (2-15), eutrophic (10-500) (Wetzel 1983)
   oligotrophic (<3.5 ug/L). mesotrophic (3.5-9), eutrophic (9.1-25), hyper-eutrophic (>25) (Nurnberg 1996).
5. Sites were only visited once (October) in 1999.

Table 4-3: Summary of chlorophyll *a* (µg/L) by reach and year and trophic status based on chlorophyll *a* concentration for the Aquatic Environment Study Area: 2001–2004 under ice cover (March/April)

Site	Year	Mean (± SE)[1]	Range	n[2]	Trophic Status	Application
Split Lake Area						
Burntwood River at Split Lake						
SPL1	2001–2004	<1 (± 0.2)	<1–1	3	oligotrophic	stream[3]
Split Lake						
All sites	2001	<1 (± 0.1)	<1–1	6	oligotrophic	lake[4]
All sites	2002	1 (± 0.2)	<1–2	6		
All sites	2004	1.3 (± 0.8)	<1–2	2		
All sites	2001–2004	<1	<1–2	14		
Assean Lake						
All sites	2001	<1 (± 0)	<1–<1	2	oligotrophic	lake
All sites	2002	1.5 (± 0.5)	1–2	2		
All sites	2001–2002	1 (± 0.4)	<1–2	4		
Keeyask Area						
Gull Lake						
All sites	2002	1	-	1	oligotrophic	lake
All sites	2003	1.3 (± 0.8)	<1–2	2		
All sites	2004	2	-	1		
All sites	2002–2004	1 (± 0.4)	<1–2	4		
Nelson River						
All sites	2001	<1	-	1	oligotrophic	stream
All sites	2003	9.3 (± 8.8)	-	2		
All sites	2004	2	-	1		
All sites	2001–2004	5.3 (± 4.3)	<1–18	4		
Stephens Lake Area						
All sites	2001	2	-	1	oligotrophic-mesotrophic	lake
All sites	2002	1 (± 0)	-	2		
All sites	2003	2 (± 2.0)	1–6	3		
All sites	2001–2004	2 (± 0.7)	1–6	7		

1. SE = standard error.
2. Number of samples collected per year.
3. Oligotrophic (<10 ug/L), mesotrophic (10-30), eutrophic (>30) (Dodds et al. 1998).
4. Oligotrophic (<2.5 ug/L). mesotrophic (2.5-8), eutrophic (8-25), hyper-eutrophic (>25) (OECD 1982) ultra-oligotrophic (0.01-0.5), oligotrophic (0.3-3). mesotrophic (2-15), eutrophic (10-500) (Wetzel 1983) oligotrophic (<3.5 ug/L). mesotrophic (3.5-9), eutrophic (9.1-25), hyper-eutrophic (>25) (Nurnberg 1996).

Table 4-4: Summary of chlorophyll *a* concentrations (µg/L) and trophic status based on chlorophyll *a* concentration at potential stream crossing sites for the Aquatic Environment Study Area: 2003–2005 open water seasons

Site	Year	Mean (± SE)[1]	Range	n[2]	Trophic Status	Application
North-Side						
SC1	2003–2005	4 (± 1.7)	<1–16	9	oligotrophic	stream[3]
SC2	2003–2005	4 (± 1.3)	<1–12	9	oligotrophic	stream
'SC3' (near Pond 13)	2003–2004	2 (± 0.4)	<1–4	8	oligotrophic	stream
South-Side						
SC3 (previously SC4)	2003–2004	2 (± 0.6)	<1–5	8	oligotrophic	stream
'SC1 (May)'	2005	3	-	1	oligotrophic	stream
'SC2 (May)'	2005	1	-	1	oligotrophic	stream
'SC3 (May)'	2005	2	-	1	oligotrophic	stream
SC5 (previously SC6-May)	2005	2	-	1	oligotrophic	stream

1. SE = standard error.
2. Number of samples collected per year.
2. Oligotrophic (<10 ug/L), mesotrophic (10-30), eutrophic (>30) (Dodds et al. 1998).

Table 4-5A: Residual effects on the phytoplankton community: construction period

Environmental Effect	Mitigation	Residual Effect(s)
Downstream of the Outlet of Clark Lake Phytoplankton biomass would be affected by changes in water quality (increases in concentration of TSS, nutrients, and metals)	A number of measures will be implemented to minimize effects of construction activities on water quality	Adverse, small magnitude, small to medium extent, and short-term duration decrease in phytoplankton biomass
South Access Road Stream Crossings No effect	N/A[1]	None

1. N/A = not applicable.

Table 4-5B: Residual effects on the phytoplankton community: operation period

Environmental Effect	Mitigation	Residual Effect(s)
Upstream of Outlet of Clark Lake No effect	Project design to avoid water level effects to Split Lake	None
Outlet of Clark Lake to Generating Station Phytoplankton biomass would be affected by changes in surface water quality (decrease in TSS and nutrients along mainstem; increase in TSS, nutrients, organic carbon, colour in off-current areas) and changes in water residence time (increase in water level and volume, reduction in water velocity)	None	Adverse (due to bloom potential), small to moderate magnitude, small extent, and long-term duration increase in phytoplankton biomass in reservoir bays with long residence times
Downstream of Generating Station Phytoplankton biomass would be affected by change in inflowing water from the reservoir (decrease in TSS) and changes in upstream phytoplankton and zooplankton	None	Positive (due to existing low biomass), small magnitude, medium extent, and long-term
North and South Access Road Stream Crossings No effect	N/A[1]	None

1. N/A = not applicable.

Table 4-6: Aquatic macrophyte taxa observed in the Aquatic Environment Study Area, 1997–2006

Scientific Name	Common Name(s)	Species Code
Vascular Macrophytes		
Callitriche palustris	vernal water-starwort common water-starwort	CALL
Carex spp.	sedge	CAREX
Ceratophyllum demersum	hornwort coon's tail	CER DEM
Cicuta spp.	water hemlock	CICUTA
Cyperaceae	sedges	CYP
Elodea canadensis	Canada waterweed Canada pondweed	ELO CAN
Eleocharis palustris	common spikerush marsh spikerush creeping spikerush	ELE PAL
Equisetum fluviatile	horsetail	EQU FLU
Galium spp.	bedstraw	GAL
Hippuris vulgaris	common mare's-tail	HIP VUL
Juncaceae	rushes	JUN
Lemna trisulca	star duckweed	LEM TRI
Limosella aquatica	water mudwort	LIM AQU
Myriophyllum sibiricum	shortspike watermilfoil northern watermilfoil common watermilfoil	MYR SIB
Nasturtium spp.	yellowcress watercress	NAST
Nuphar spp.	pond lily	NUPH
Poaceae	true grasses	POA
Polygonum amphibium	water knotweed water smartweed	POL AMP
Polygonum persicaria	spotted ladysthumb lady's thumb ladysthumb smartweed	POL PER
Potamogeton friesii	Fries' pondweed	POT FRI
Potamogeton gramineus	variableleaf pondweed	POT GRA
Potamogeton praelongus	whitestem pondweed	POT PRA

Table 4-6: Aquatic macrophyte taxa observed in the Aquatic Environment Study Area, 1997–2006

Scientific Name	Common Name(s)	Species Code
Potamogeton richardsonii	Richardson's pondweed clasping-leaved pondweed	POT RIC
Potamogeton zosteriformis	flatstem pondweed flat-stemmed pondweed eel-grass pondweed	POT ZOS
Ranunculus aquatilis	white water-crowfoot common water-crowfoot	RAN AQU
Sagittaria cuneata	arumleaf arrowhead wapato	SAG CUN
Scirpus spp.	bulrush	SCIRP
Sparganium spp.	bur-reed	SPARG
Stukenia filiformis	fineleaf pondweed	STU FIL
Stukenia pectinata	sago pondweed broadleaf pondweed	STU PEC
Stukenia vaginata	big-sheath pondweed sheathed pondweed	STU VAG
Typha spp.	cattail	TYPHA
Utricularia vulgaris	common bladderwort greater bladderwort	UTR VUL
Zannichellia palustris	horned pondweed	ZAN PAL
Non-Vascular Macrophytes		
Aquatic Moss		MOSS
Chara spp.	muskgrass stonewort	CHARA
Cyanophycota	blue-green algae cyanophytes	CYAN
Filamentous Green Algae	FGA	FGA
Sphagnum spp.	aquatic moss	SPHAG

Table 4-7: Areas (in hectares [ha]) and percentages of aquatic macrophyte coverage per reach at selected locations in Clark Lake and the Keeyask area, 2001

Reach	Location	Area (ha) Covered by Macrophytes	Percentage of Total Macrophytes	Percentage of Geographic Zone	Total Area (ha) of Geographic Zone [1]	Species Observed [2]
1	Clark Lake	5.8	1.5	0.5	1154.1	NR
2A	Nelson River - D/S of Clark Lake	1.3	0.3	0.6	200.1	NR
2B	Nelson River - U/S of Fork Creek	0.0	0.0	0.0	198.2	NP
3	Nelson River - U/S of Birthday Rapids	0.9	0.3	0.4	268.5	NR
4	Nelson River - D/S of Birthday Rapids	1.0	0.3	0.3	307.0	CAREX; NUPH; POT
5	Nelson River - Poplar Bay to Kahpowinic Bay	2.6	0.7	0.3	750.4	ELEPAL; POLPER; POT; STUK
5	Pahwaybanic Bay	29.1	7.7	37.9	76.7	Refer to Table 4-12
Total 5		31.7	8.4	3.8	827.1	-
6	Nelson River - Kahpowinic Bay to John Garson Bay (Gull Lake)	128.1	34.1	6.3	2038.8	ELEPAL; NUPH; POLPER; POT; POTGRA; STUVAG
6	Kahpowinic Bay	13.6	3.6	22.4	61.0	Refer to Table 4-12
6	Small Bay East of Rabbit Creek	11.7	3.1	45.0	25.9	STUVAG
6	John Garson Bay	59.3	15.8	54.5	108.8	Refer to Table 4-12
Total 6		212.7	56.6	9.5	2234.6	-
7	Nelson River - Gull Lake	18.2	4.8	2.6	709.3	ELEPAL; LEMTRI; POLPER; POT; POTRIC; STUVAG
7	Tub Bay	6.9	1.8	34.0	20.4	Refer to Table 4-12
7-8	John Kitch Bay	4.2	1.1	3.0	139.3	NR
8	Nelson River - Gull Lake	62.6	16.6	11.0	568.3	POTRIC; STUVAG
8	North of Caribou Island (Gull Lake)	29.6	7.9	43.8	67.6	Refer to Table 4-12

Table 4-7: Areas (in hectares [ha]) and percentages of aquatic macrophyte coverage per reach at selected locations in Clark Lake and the Keeyask area, 2001

Reach	Location	Area (ha) Covered by Macrophytes	Percentage of Total Macrophytes	Percentage of Geographic Zone	Total Area (ha) of Geographic Zone [1]	Species Observed [2]
Total 7, 8		121.6	32.3	8.1	1504.8	-
9	Nelson River - Gull Rapids	0.1	0.0	0.0	486.4	POT
10	Mouth of Pond 13	NS	NS	NS	2.1	NS
11	Nelson River - D/S of Gull Rapids	0.9	0.2	0.2	564.1	POT
Total Area		**376.1**	**100.0**	**4.9**	**7747.0**	**-**

1. Area is based on the 95^{th} flow percentile shoreline.
2. Species observed during surveys; codes as per Table 4-6; species listed in alphabetical order

NS - Aquatic macrophytes not surveyed in this area during study year
NP - Aquatic macrophytes not present in this area during study year
NR - Species information not recorded in this area during study year.

Table 4-8: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2003

Reach	1		5		6		6		6		7		7-8		8	
Location	Clark Lake		Pahwaybanic Bay		Kahpowinic Bay		Small Bay East of Rabbit Creek		John Garson Bay		Tub Bay		John Kitch Bay		North of Caribou Island	
Species	%	RD[1]	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD
Vascular Macrophytes																
Callitriche palustris	31.5	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Carex spp.	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Ceratophyllum demersum	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Cicuta sp.	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Eleocharis palustris	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Equisetum fluviatile	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Hippuris vulgaris	0.0	0	0.0	0	0.0	0	0.0	0	0.0	P	0.0	0	0.0	0	0.0	0
Lemna trisulca	0.0	0	3.7	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Myriophyllum sibiricum	0.0	0	0.0	0	0.0	0	0.0	0	0.0	P	0.8	1	0.0	P	0.0	1
Poaceae	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Polygonum amphibium	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton friesii	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton gramineus	0.0	0	0.0	0	24.5	1	15.8	1	4.9	1	12.8	1	0.0	P	2.7	1
Potamogeton praelongus	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton richardsonii	55.3	1	7.4	1	0.0	P	4.7	1	90.1	2	5.8	1	2.1	1	4.7	1
Potamogeton zosteriformis	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton spp.	4.6	1	0.0	0	3.3	1	0.0	0	0.0	0	7.5	1	6.6	1	0.8	1
Sagittaria cuneata	0.0	0	17.9	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Scirpus sp.	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0

Table 4-8: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2003

Reach	1		5		6		6		6		7		7-8		8	
Location	Clark Lake		Pahwaybanic Bay		Kahpowinic Bay		Small Bay East of Rabbit Creek		John Garson Bay		Tub Bay		John Kitch Bay		North of Caribou Island	
Species	**%**	**RD[1]**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**
Sparganium sp.	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Stuckenia pectinatus	0.7	1	0.0	0	60.4	1	0.0	0	0.0	0	1.1	1	2.2	1	6.5	1
Stuckenia vaginatus	7.8	1	70.4	1	0.0	0	79.3	3	5.0	1	59.7	1	74.1	1	55.2	1
Stuckenia sp.	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Utricularis vulgaris	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Non-Vascular Macrophytes																
aquatic moss	0.0	0	0.0	0	9.4	1	0.1	1	0.0	0	0.0	0	14.9	1	30.1	1
Chara spp.	0.0	0	0.6	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Cyanophycota	0.0	0	0.0	0	2.4	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
filamentous algae	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	11.5	1	0.0	0	0.0	0
unidentified	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.8	1	0.0	0	0.0	0
Overall Relative Density	-	2	-	2	-	2	-	3	-	3	-	2	-	2	-	2
Number of Vascular Taxa	5	-	4	-	4	-	3	-	5	-	6	-	6	-	6	-
Number of Non-Vascular Taxa	0	-	1	-	2	-	1	-	0	-	1	-	1	-	1	-

P - presence of species noted during walking survey.
1. RD = relative density; dry weight of species (g/m^2) = 0, code = 0, definition = absent
g/m^2 = 0-10, code = 1, definition = sparse
g/m^2 = 10-30, code = 2, definition = low density
g/m^2 = 30-60, code = 3, definition = medium density
g/m^2 >60, code = 4, definition = high density.

Table 4-9: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2004

Reach	1		5		6		6		6		7		7-8		8	
Location	Clark Lake		Pahwaybanic Bay		Kahpowinic Bay		Small Bay East of Rabbit Creek		John Garson Bay		Tub Bay		John Kitch Bay		North of Caribou Island	
Species	%	RD[1]	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD
Vascular Macrophytes																
Callitriche palustris	0.0	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Carex spp.	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Ceratophyllum demersum	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Cicuta sp.	0.2	1	0.0	0	1.3	1	0.0	0	3.3	1	0.0	0	0.0	0	0.0	0
Eleocharis palustris	60.9	2	35.2	2	52.0	2	0.4	1	91.2	2	4.0	1	0.1	1	0.0	0
Equisetum fluviatile	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Hippuris vulgaris	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Lemna trisulca	0.0	1	0.4	1	2.2	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Myriophyllum sibiricum	0.2	1	27.3	2	3.3	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Poaceae	0.4	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Polygonum amphibium	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton friesii	0.0	0	0.0	0	34.3	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton gramineus	0.0	0	0.1	1	0.7	1	42.3	1	0.0	0	0.2	1	0.0	0	0.0	0
Potamogeton praelongus	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton richardsonii	36.1	1	22.9	1	3.2	1	25.0	1	0.0	0	71.0	2	22.0	1	17.5	1
Potamogeton zosteriformis	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Potamogeton spp.	0.0	1	0.3	1	0.0	0	0.0	0	0.3	1	0.0	0	0.0	0	0.0	0
Sagittaria cuneata	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Scirpus sp.	0.0	0	3.4	1	2.6	1	0.0	0	2.9	1	0.0	0	0.0	0	0.0	0
Sparganium sp.	1.9	1	2.3	1	0.0	0	2.4	1	2.2	1	0.9	1	0.0	0	0.0	0

Table 4-9: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2004

Reach	1		5		6		6		6		7		7-8		8	
Location	**Clark Lake**		**Pahwaybanic Bay**		**Kahpowinic Bay**		**Small Bay East of Rabbit Creek**		**John Garson Bay**		**Tub Bay**		**John Kitch Bay**		**North of Caribou Island**	
Species	**%**	**RD[1]**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**
Stuckenia pectinatus	0.2	1	3.3	1	0.3	1	23.1	1	0.0	0	7.9	1	0.0	0	5.7	1
Stuckenia vaginatus	0.0	0	0.0	0	0.0	0	3.0	1	0.0	0	15.2	1	53.6	1	47.0	2
Stuckenia sp.	0.0	0	0.0	0	0.0	0	3.8	1	0.1	1	0.0	0	0.0	1	2.8	1
Utricularis vulgaris	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Non-Vascular Macrophytes																
aquatic moss	0.0	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Chara spp.	0.0	0	4.2	1	0.1	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0
Cyanophycota	0.0	1	0.0	1	0.0	1	0.0	1	0.0	0	0.0	0	0.2	1	0.0	0
filamentous algae	0.0	0	0.6	1	0.0	0	0.0	0	0.0	0	0.7	1	0.9	1	26.9	1
unidentified	0.0	1	0.0	0	0.0	0	0.0	0	0.0	0	0.0	0	23.1	1	0.0	0
Overall Relative Density	-	2	-	3	-	2	-	1	-	3	-	2	-	1	-	2
Number of Vascular Taxa	10	-	9	-	9	-	7	-	6	-	6	-	4	-	4	-
Number of Non-Vascular Taxa	2	-	2	-	2	-	1	-	0	-	1	-	2	-	1	-

1. RD = relative density; dry weight of species (g/m^2) = 0, code = 0, definition = absent
g/m^2 = 0-10, code = 1, definition = sparse
g/m^2 = 10-30, code = 2, definition = low density
g/m^2 = 30-60, code = 3, definition = medium density
g/m^2 >60, code = 4, definition = high density.

Table 4-10: Areas (in hectares [ha]) and percentages of aquatic macrophyte coverage per reach at selected locations in Clark Lake and the Keeyask area, 2006

Reach	Location	Area (ha) Covered by Macrophytes	Percentage of Total Macrophytes	Percentage of Geographic Zone	Total Area (ha) of Geographic Zone [1]	Species Observed [2]
1	Clark Lake	NS	NS	NS	1154.1	NS
2A	Nelson River - D/S of Clark Lake	NS	NS	NS	200.1	NS
2B	Nelson River - U/S of Fork Creek	NS	NS	NS	198.2	NS
3	Nelson River - U/S of Birthday Rapids	NS	NS	NS	268.5	NS
4	Nelson River - D/S of Birthday Rapids	1.1	0.7	0.3	307.0	ELE; POTGRA; POTRIC
5	Nelson River - Poplar Bay to Kahpowinic Bay	8.2	5.1	1.1	750.4	MYRSIB; POTGRA; POTRIC; STU
5	Pahwaybanic Bay	9.6	6.0	12.5	76.7	ELE; MYRSIB; POLAMP; POTGRA; STU
Total 5		17.8	11.1	2.2	827.1	-
6	Nelson River - Kahpowinic Bay to John Garson Bay (Gull Lake)	27.8	17.3	1.4	2038.8	ELE; MYRSIB; POLAMP; POTGRA; POTRIC
6	Kahpowinic Bay	42.3	26.3	69.3	61.0	ELE; POLAMP; POTGRA; POTRIC; SAG; STU
6	Small Bay East of Rabbit Creek	5.8	3.6	22.2	25.9	ELE; POTGRA
6	John Garson Bay	22.5	14.0	20.7	108.8	ELE; POLAMP; POTGRA; POTRIC; STU
Total 6		98.4	61.3	4.4	2234.6	-
7	Nelson River - Gull Lake	6.6	4.1	0.9	709.3	ELE; MYRSIB; POLAMP; POTGRA; POTRIC

Table 4-10: Areas (in hectares [ha]) and percentages of aquatic macrophyte coverage per reach at selected locations in Clark Lake and the Keeyask area, 2006

Reach	Location	Area (ha) Covered by Macrophytes	Percentage of Total Macrophytes	Percentage of Geographic Zone	Total Area (ha) of Geographic Zone [1]	Species Observed [2]
7	Tub Bay	0.0	0.0	0.0	20.4	-
7-8	John Kitch Bay	11.6	7.2	8.3	139.3	ELE; POTGRA; POTRIC; STU
8	Nelson River - Gull Lake	23.3	14.5	4.1	568.3	ELE; MYRSIB; POLAMP; POTGRA; POTRIC; STU
8	North of Caribou Island (Gull Lake)	1.0	0.6	1.5	67.6	ELE; POTGRA; FGA
Total 7, 8		42.5	26.5	2.8	1504.8	-
9	Nelson River - Gull Rapids	0.4	0.2	0.1	486.4	MYRSIB; STU
10	Mouth of Pond 13	0.04	0.02	0.01	2.1	MYRSIB; POLAMP; RANAQU; SPARG; STU
11	Nelson River - D/S of Gull Rapids	0.4	0.2	0.1	564.1	MYRSIB; RANAQU; STU
Total Area		**160.6**	**100.0**	**2.1**	**7747.0**	**-**

1. Area is based on the 95th flow percentile shoreline.
2. Species observed during surveys; codes as per Table 4-6; species listed in alphabetical order.
NS - Aquatic macrophytes not surveyed in this area during study year.

Table 4-11: Areas (in hectares [ha]) and percentages of aquatic macrophyte coverage per reach at selected locations in Clark Lake and the Keeyask area, 2003

Reach	Location	Area (ha) Covered by Macrophytes	Percentage of Total Macrophytes	Percentage of Geographic Zone	Total Area (ha) of Geographic Zone [1]	Species Observed [2]
1	Clark Lake	NS	NS	NS	1154.1	Refer to Table 4-8
2A	Nelson River - D/S of Clark Lake	NS	NS	NS	200.1	NS
2B	Nelson River - U/S of Fork Creek	NS	NS	NS	198.2	NS
3	Nelson River - U/S of Birthday Rapids	NS	NS	NS	268.5	NS
4	Nelson River - D/S of Birthday Rapids	NS	NS	NS	307.0	NS
5	Nelson River - Poplar Bay to Kahpowinic Bay	3.1	1.1	0.4	750.4	MYRSIB; POTRIC; STUPEC
5	Pahwaybanic Bay	NS	NS	NS	76.7	Refer to Table 4-8
Total 5		3.1	1.1	0.4	827.1	-
6	Nelson River - Kahpowinic Bay to John Garson Bay (Gull Lake)	34.4	12.2	1.7	2038.8	HIPVUL; MYRSIB; POTRIC; POT; STUPEC
6	Kahpowinic Bay	1.7	0.6	2.8	61.0	Refer to Table 4-8
6	Small Bay East of Rabbit Creek	2.7	1.0	10.5	25.9	Refer to Table 4-8
6	John Garson Bay	55.9	19.8	51.3	108.8	Refer to Table 4-8
Total 6		94.6	33.5	4.2	2234.6	-
7	Nelson River - Gull Lake	0.4	0.2	0.1	709.3	POTRIC
7	Tub Bay	12.8	4.5	62.5	20.4	Refer to Table 4-8
7-8	John Kitch Bay	100.6	35.6	72.2	139.3	Refer to Table 4-8
8	Nelson River - Gull Lake	27.0	9.6	4.8	568.3	HIPVUL; MYRSIB; POTRIC; POT
8	North of Caribou Island (Gull Lake)	44.2	15.6	65.4	67.6	Refer to Table 4-8
Total 7, 8		185.0	65.4	12.3	1504.8	-

Table 4-11: Areas (in hectares [ha]) and percentages of aquatic macrophyte coverage per reach at selected locations in Clark Lake and the Keeyask area, 2003

Reach	Location	Area (ha) Covered by Macrophytes	Percentage of Total Macrophytes	Percentage of Geographic Zone	Total Area (ha) of Geographic Zone [1]	Species Observed [2]
9	Nelson River - Gull Rapids	NS	NS	NS	486.4	NS
10	Mouth of Pond 13	NS	NS	NS	2.1	NS
11	Nelson River - D/S of Gull Rapids	NS	NS	NS	564.1	NS
Total Area		**282.7**	**100.0**	**3.6**	**7747.0**	**-**

1. Area is based on the 95th flow percentile shoreline
2. Species observed during surveys; codes as per Table 4-6; species listed in alphabetical order

NS - Aquatic macrophytes not surveyed in this area during study year

Table 4-12: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2001

Reach	1		5		6		6		6		7		7-8		8	
Location	Clark Lake		Pahwaybanic Bay		Kahpowinic Bay		Small Bay East of Rabbit Creek		John Garson Bay		Tub Bay		John Kitch Bay		North of Caribou Island	
Species	%	RD[1]	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD
Vascular Macrophytes																
Callitriche palustris	NS	NS	0.0	0	0.0	0	NS	NS	0.0	0	0.0	0	NS	NS	0.0	0
Carex spp.			0.0	0	0.0	0			0.0	0	0.5	1			0.0	0
Ceratophyllum demersum			0.0	0	0.0	1			0.0	0	0.0	0			0.0	0
Cicuta sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Eleocharis palustris			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Equisetum fluviatile			0.0	0	10.8	1			0.0	0	0.0	0			0.0	0
Hippuris vulgaris			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Lemna trisulca			24.5	1	38.5	2			64.6	3	0.0	0			0.0	0
Myriophyllum sibiricum			40.3	2	34.2	2			0.2	1	0.9	1			0.0	0
Poaceae			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Polygonum amphibium			0.0	0	0.0	0			2.6	1	48.2	1			0.0	0
Potamogeton friesii			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Potamogeton gramineus			0.0	0	0.0	0			1.7	1	0.0	0			0.0	0
Potamogeton praelongus			0.0	0	0.0	0			0.0	0	0.0	0			16.2	1
Potamogeton richardsonii			0.9	1	1.3	1			0.9	1	2.8	1			6.5	1
Potamogeton zosteriformis			0.0	0	0.0	0			0.0	0	2.9	1			19.4	1
Potamogeton spp.			5.1	1	9.1	1			23.5	2	31.3	1			37.8	1
Sagittaria cuneata			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Scirpus sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Sparganium sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0

Table 4-12: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2001

Reach	1		5		6		6		6		7		7-8		8	
Location	Clark Lake		Pahwaybanic Bay		Kahpowinic Bay		Small Bay East of Rabbit Creek		John Garson Bay		Tub Bay		John Kitch Bay		North of Caribou Island	
Species	**%**	**RD[1]**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**
Stuckenia pectinatus			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Stuckenia vaginatus			21.2	1	0.0	0			0.0	0	0.0	0			0.0	0
Stuckenia sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Utricularis vulgaris			0.2	1	0.0	0			0.1	1	0.0	0			0.0	0
Non-Vascular Macrophytes																
aquatic moss			2.1	1	0.0	0			4.0	1	0.1	1			0.0	0
Chara spp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Cyanophycota			4.1	1	4.7	1			0.1	1	11.3	1			3.3	1
filamentous algae			0.0	0	1.1	1			0.0	0	0.0	0			15.6	1
unidentified			1.5	1	0.4	1			2.3	1	0.7	1			1.1	1
Overall Relative Density			-	3	-	3			-	3	-	2			-	2
Number of Vascular Taxa			6	-	5	-			7	-	6	-			4	-
Number of Non-Vascular Taxa			2	-	2	-			2	-	2	-			2	-

NS - Aquatic macrophytes not surveyed in this area during study year.
1. RD = relative density; dry weight of species (g/m^2) = 0, code = 0, definition = absent
g/m^2 = 0-10, code = 1, definition = sparse
g/m^2 = 10-30, code = 2, definition = low density
g/m^2 = 30-60, code = 3, definition = medium density
g/m^2 >60, code = 4, definition = high density.

Table 4-13: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2002

Reach	1		5		6		6		6		7		7-8		8	
Location	Clark Lake		Pahwaybanic Bay		Kahpowinic Bay		Small Bay East of Rabbit Creek		John Garson Bay		Tub Bay		John Kitch Bay		North of Caribou Island	
Species	%	RD[1]	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD	%	RD
Vascular Macrophytes																
Callitriche palustris	NS	NS	0.0	0	0.0	0	NS	NS	0.0	0	0.0	0	NS	NS	0.0	0
Carex spp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Ceratophyllum demersum			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Cicuta sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Eleocharis palustris			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Equisetum fluviatile			0.0	0	0.4	1			0.0	0	0.0	0			0.0	0
Hippuris vulgaris			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Lemna trisulca			18.9	2	53.5	2			0.1	1	2.2	1			0.0	0
Myriophyllum sibiricum			52.9	3	3.3	1			0.0	0	0.0	0			0.0	0
Poaceae			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Polygonum amphibium			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Potamogeton friesii			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Potamogeton gramineus			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Potamogeton praelongus			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Potamogeton richardsonii			1.0	1	32.1	2			77.5	3	49.2	2			9.4	1
Potamogeton zosteriformis			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Potamogeton spp.			0.0	0	0.1	1			0.0	0	31.4	2			90.6	4
Sagittaria cuneata			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Scirpus sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Sparganium sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0

Table 4-13: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Clark Lake and the Keeyask area, 2002

Reach	**1**		**5**		**6**		**6**		**6**		**7**		**7-8**		**8**	
Location	**Clark Lake**		**Pahwaybanic Bay**		**Kahpowinic Bay**		**Small Bay East of Rabbit Creek**		**John Garson Bay**		**Tub Bay**		**John Kitch Bay**		**North of Caribou Island**	
Species	**%**	**RD[1]**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**	**%**	**RD**
Stuckenia pectinatus			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Stuckenia vaginatus			24.6	2	6.7	1			21.7	2	17.1	1			0.0	0
Stuckenia sp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Utricularis vulgaris			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Non-Vascular Macrophytes																
aquatic moss			0.0	0	0.2	1			0.6	1	0.0	0			0.0	0
Chara spp.			0.0	0	0.0	0			0.0	0	0.0	0			0.0	0
Cyanophycota			0.6	1	1.6	1			0.0	0	0.1	1			0.0	0
filamentous algae			0.0	0	1.4	1			0.0	0	0.0	0			0.0	0
unidentified			2.0	1	0.6	1			0.0	0	0.0	0			0.0	0
Overall Relative Density			-	4	-	3			-	4	-	3			-	4
Number of Vascular Taxa			4	-	6	-			3	-	4	-			2	
Number of Non-Vascular Taxa			1	-	3	-			1	-	1	-			0	

NS - Aquatic macrophytes not surveyed in this area during study year.
1. RD = relative density; dry weight of species (g/m^2) = 0, code = 0, definition = absent
   g/m^2 = 0-10, code = 1, definition = sparse
   g/m^2 = 10-30, code = 2, definition = low density
   g/m^2 = 30-60, code = 3, definition = medium density
   g/m^2 >60, code = 4, definition = high density.

Table 4-14: Mean drifting plant density and community composition information for large drift traps set in the Aquatic Environment Study Area in comparable sampling periods during the open-water season, 2003 and 2004

Location of Drift Traps	Upstream of Birthday Rapids	Downstream of Birthday Rapids	Upstream of Gull Rapids (at the downstream end of Gull Lake)	Downstream of Gull Rapids (near the base of Gull Rapids)	Downstream of the Kettle GS
n[1]	**20**	**20**	**20**	**16**	**16**
Mean Drift Density (mg dried weight/100 m^3)					
Non-Vascular	7.554	45.737	9.198	13.433	3.457
Vascular	2.761	10.520	24.372	12.715	1.140
Total Plants	**10.315**	**56.239**	**34.541**	**26.148**	**4.597**
Percent Composition (%)					
Non-Vascular	73.231	81.326	26.630	51.372	75.191
Vascular	26.769	18.705	70.560	48.628	24.809

1. Number of samples collected per area for 2003 and 2004 sampling periods combined.

Table 4-15: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Stephens Lake, 2005–2006

Study Year	2005		2006	
Species	**%**	**Relative Density[1]**	**%**	**Relative Density**
Vascular Macrophytes				
Callitriche palustris	0.0	0	0.0	0
Carex spp.	0.0	0	0.0	0
Ceratophyllum demersum	0.0	0	0.0	0
Cicuta sp.	0.0	0	0.0	0
Eleocharis palustris	0.0	0	0.0	0
Equisetum fluviatile	0.0	0	0.0	0
Hippuris vulgaris	0.0	0	0.0	0
Lemna trisulca	0.0	1	0.0	0
Myriophyllum sibiricum	19.4	1	24.9	2
Poaceae	0.0	P	0.0	P
Polygonum amphibium	0.0	P	12.2	1
Potamogeton friesii	0.0	0	0.0	0
Potamogeton gramineus	4.3	1	0.0	P
Potamogeton praelongus	0.0	0	0.0	0
Potamogeton richardsonii	53.4	2	54.4	3
Potamogeton zosteriformis	0.0	0	0.0	0
Potamogeton spp.	0.0	0	0.0	0
Sagittaria cuneata	0.0	0	0.0	0
Scirpus sp.	0.0	0	0.0	0
Sparganium sp.	0.0	0	0.0	0
Stuckenia pectinatus	14.6	1	0.0	P
Stuckenia vaginatus	0.0	P	8.6	1
Stuckenia sp.	0.0	0	0.0	0
Utricularis vulgaris	0.2	1	0.0	P

Table 4-15: Composition (%) and estimate of relative density of aquatic macrophyte species per selected locations in Stephens Lake, 2005–2006

Study Year	**2005**		**2006**	
Species	**%**	**Relative Density[1]**	**%**	**Relative Density**
Non-Vascular Macrophytes				
aquatic moss	2.0	1	0.0	0
Chara spp.	0.0	0	0.0	0
Cyanophycota	0.0	1	0.0	0
filamentous algae	0.0	0	0.0	0
unidentified	6.1	1	0.0	0
Overall Relative Density	-	3	-	4
Number of Vascular Taxa	9	-	8	-
Number of Non-Vascular Taxa	3	-	0	-

P - presence of species noted during walking survey.
1. dry weight of species (g/m^2) = 0, code = 0, definition = absent
   g/m^2 = 0-10, code = 1, definition = sparse
   g/m^2 = 10-30, code = 2, definition = low density
   g/m^2 = 30-60, code = 3, definition = medium density
   g/m^2 >60, code = 4, definition = high density.

Table 4-16: Occupied aquatic vascular plant habitat in the existing environment (EE) and at post-Project (PP) time steps under different generating station operating scenarios

	EE[1]	Year 1 PP			Year 5 PP			Year 15 PP			Year 30 PP		
		Peaking Mode[2]		Base Loaded[3]	Peaking Mode		Base Loaded	Peaking Mode		Base Loaded	Peaking Mode		Base Loaded
		MOL[4]	FSL[5]		MOL	FSL		MOL	FSL		MOL	FSL	
Total Occupied Area (ha)	207.3	0.1	2.4	2.4	0.1	2.4	2.4	10.0	37.6	37.6	48.9	187.8	187.8
Loss(-)/Gain(+) of Occupied Area (ha)	0.0	-207.2	-204.9	-204.9	-207.2	-204.9	-204.9	-197.3	-169.7	-169.7	-158.4	-19.5	-19.5
Percent of Occupied Area Relative to EE (%)	100.0	0.1	1.1	1.1	0.1	1.1	1.1	4.8	18.1	18.1	23.6	90.6	90.6

1. At the 95th percentile flow.
2. Assumes weekly cycling.
3. Assumes no cycling (*i.e.*, Full Supply Level [FSL] with no intermittently exposed zone [IEZ]).
4. Minimum Operating Level – no IEZ.
5. Includes IEZ.

Table 4-17A: Residual effects on the aquatic macrophyte community: construction period

Environmental Effect	Mitigation	Residual Effect(s)
Downstream of the Outlet of Clark Lake Aquatic plant abundance would be affected by changes in water quality (increases in concentrations of TSS, nutrients, and metals) and changes in physical attributes of aquatic habitat (downstream sedimentation, loss of habitat in footprint of supporting infrastructure)	A number of measures will be implemented to minimize effects of construction on water quality	Adverse, moderate magnitude, small extent, and long-term duration decrease in production of drifting non-vascular plant (predominantly filamentous algae) biomass from Gull Rapids
South Access Road Stream Crossings Aquatic plant cover would be affected by infilling of aquatic habitat at crossings	A number of measures will be implemented to minimize effects of construction on water quality and aquatic habitat	Adverse, large magnitude, small extent, and long-term loss of aquatic plants at culvert locations

Table 4-17B: Residual effects on the aquatic macrophyte community: operation period

Environmental Effect	Mitigation	Residual Effect(s)
Upstream of Outlet of Clark Lake No effect	Project design to avoid water level effects to Split Lake	None
Outlet of Clark Lake to Generating Station Aquatic plant distribution and abundance would be affected by: flooding (loss of existing habitats, creation of new habitats); conversion of existing hard substrates to silt/clay due to sedimentation in Gull Lake; increase in the frequency of water level fluctuations (reduction in overall magnitude); and, reduction in the extent and severity of ice scour	None	Adverse, large (short-term duration) to small magnitude (long-term duration), and medium extent reduction in occupied aquatic macrophyte habitat Adverse, large (small extent) to small (medium extent) magnitude, and long-term duration decrease in production of drifting non-vascular (predominantly filamentous algae) and vascular plant biomass
Downstream of Generating Station Aquatic plant distribution and abundance would be affected by: reduction in the extent and severity of ice scour in certain areas near the inflow of the river to Stephens Lake; direct loss of aquatic habitat due to dewatering of Gull Rapids and footprint of GS; and no effect in Stephens Lake proper	None	Adverse, moderate magnitude, small extent, and long-term duration decrease in production of drifting non-vascular plant (predominantly filamentous algae) biomass
North and South Access Road Stream Crossings Aquatic plant cover would be affected by infilling of aquatic habitat at crossings and inputs of sediment from erosion	Clear span bridge on Looking Back Creek; placement of culverts as per Manitoba Stream Crossing Guidelines; effective erosion control measures	Adverse, large magnitude, small extent, and long-term loss of aquatic plants at culvert locations

Table 4-18: Zooplankton summary statistics for Aquatic Environment Study Area lakes in 2001 and 2002 for all sampling periods

Study Year	2001				2002			
Lake Sampled	Number of Samples	Overall Mean Density (individuals/m^3)	SE[1]	Range	Number of Samples	Overall Mean Density (individuals/m^3)	SE	Range
Assean (standing water)	8	47,516	8,234	17,214–79,756	8	42,254	13,013	12,204–101,017
Split (standing and flowing)	23	2,929	770	236–14,681	23	6380	1,728	140–26,017
Split (standing only)	5	7,619	2,382	1,991–14,681	8	14,664	3,254	2,126–26,017
Split (flowing only)	18	1,626	388	236–5,576	15	1,962	605	140–8,756
Clark (flowing)	4	2,672	1,622	706–7,520	3	2,845	1,776	590–6,349
Gull (flowing)	8	779	224	64–1,618	8	705	231	39–1,817
Stephens (flowing)	8	264	65	94–579	8	761	198	22–1,619

1. SE = standard error.

Table 4-19: The total number of Cladocera and Copepoda taxa found in the Aquatic Environment Study Area, 2001 and 2002

Waterbody	2001	2002
Spilt Lake Area		
Split Lake	30	27
Clark Lake	23	21
Assean Lake	27	23
Keeyask Area		
Gull Lake	21	21
Stephens Lake Area		
Stephens Lake	24	22

Table 4-20A: Residual effects on the zooplankton community: construction period

Environmental Effect	Mitigation	Residual Effect(s)
Downstream of the Outlet of Clark Lake Zooplankton abundance would be affected by changes in water quality (increases in concentrations of TSS and metals; decreases in DO concentrations) and changes in phytoplankton (decrease in biomass)	A number of measures will be implemented to minimize effects of construction activities on water quality	Given the nature and duration of expected changes to water quality and phytoplankton, no effects are expected

Table 4-20A: Residual effects on the zooplankton community: operation period

Environmental Effect	Mitigation	Residual Effect(s)
Upstream of Outlet of Clark Lake No effect	Project design to avoid water level effects to Split Lake	None
Outlet of Clark Lake to Generating Station Zooplankton abundance and species composition would be affected by: changes in surface water quality (e.g., decrease in dissolved oxygen concentrations); changes in water residence time (increase in water level and volume, reduction in water velocity); and changes in phytoplankton	None	Positive (food source for fish), small to moderate magnitude, small extent, and long-term duration increase in zooplankton abundance in reservoir bays with long residence times; shift in community composition to larger daphnids (type of cladoceran)
Downstream of Generating Station Zooplankton abundance would be affected by change in inflowing water from the reservoir (decrease in TSS) and changes in upstream phytoplankton and zooplankton	None	No change is expected in Stephens Lake as the water residence time is too short for zooplankton to increase in response to any changes in phytoplankton biomass

Table 4-21: Number of taxa observed for the sediment-dwelling, plant-dwelling, and drifting macroinvertebrate communities of the Aquatic Environment Study Area, 1999–2004

Waterbody	Sediment-Dwelling Community		Plant-Dwelling Community		Drifting Community		All Communities	
	Years[1]	n[2]	Years	n	Years	n	Years	n
Split Lake Area								
Assean River	-	-	-	-	2001–2002	30	-	-
Assean Lake	2001–2002, 2004	40 (19) [4]	-	-	-	-	-	-
Assean Total	**2001–2002, 2004**	**40 (19)**			**2001–2002**	**30**	**2001, 2002, 2004**	**55 (34)**
Clark Lake	2001–2002, 2004	36 (13)	2003–2004	25	-	-	-	-
Split Lake/York Landing Arm	2001–2002	21	-	-	-	-	-	-
Split Lake Area Total [3]	**2001–2002, 2004**	**44 (21)**	**2003–2004**	**25**	**-**	**-**	**2001–2004**	**49 (21)**
Keeyask Area								
Tributaries [5]	-	-	-	-	2001–2002	27	-	-
Upstream of Birthday Rapids	-	-	-	-	2003–2004	51	-	-
Downstream of Birthday Rapids	-	-	-	-	2001–2004	65 (25)	-	-
Upstream of Gull Rapids	-	-	-	-	2003–2004	50	-	-
Downstream of Gull Rapids	-	-	-	-	2001–2004	69 (30)	-	-
Mainstem Total	1999, 2001–2002, 2004	43 (22)	2001–2004	56 (19)	2001–2004	83 (32)	-	-
Keeyask Area Total	**1999, 2001–2002, 2004**	**43 (22)**	**2001–2004**	**56 (19)**	**2001–2004**	**85 (36)**	**1999, 2001–2004**	**93 (39)**

Table 4-21: Number of taxa observed for the sediment-dwelling, plant-dwelling, and drifting macroinvertebrate communities of the Aquatic Environment Study Area, 1999–2004

Waterbody	Sediment-Dwelling Community		Plant-Dwelling Community		Drifting Community		All Communities	
	Years[1]	n[2]	Years	n	Years	n	Years	n
Stephens Lake Area								
Stephens Lake	2001–2002, 2004	32 (16)	-	34 [6]	-	-	-	-
Downstream of Kettle GS	-	-	-	-	2003–2004	40	-	-
Stephens Lake Area Total	**2001–2002, 2004**	**32 (16)**	**-**	**34**	**2003–2004**	**40**	**2001–2004**	**54 (16)**
Overall Total	**1999, 2001–2002, 2004**	**58 (25)**	**2001–2004**	**56 (19)**	**2001–2004**	**85 (37)**	**1999, 2001–2004**	**95 (40)**

1. Data included from Lower Nelson River Information System (LNRIS) database (1999-2004 finalized data imported).
   data that are part of EIS, but not included in the LNRIS database:
   Split Lake/York Landing Arm, Sediment-Dwelling, 1997, 1998, and 2000 (TEMA and York Factory First Nation programs not included in LNRIS database)
   Stephens Lake, Sediment-Dwelling, 2006 (to be imported to LNRIS database when technical report finalized)
   Stephens Lake, Plant-Dwelling, 2005, 2006 (to be imported to LNRIS database when technical report finalized)
2. Number of taxa reported at Family level; if group identified to higher level, then it was assumed that only one Family was represented and this likely resulted in a conservative estimate of number of taxa.
3. Split Lake Area Total includes Clark Lake and Split Lake/York Landing Arm only.
4. Number in parentheses includes data for 1999, 2001 and 2002 only; macroinvertebrates identified to lower taxonomic level in 2003 and 2004 (see Appendix 4A) and this resulted in a step-trend increase to number of taxa.
5. Nap Creek, Portage Creek, and Two Goose Creek.
6. 2005 and 2006 data for Stephens Lake not imported to LNRIS database and not included in overall or communities totals.

Table 4-22: Comparison of overall mean number of sediment-dwelling macroinvertebrates (individuals/m^2) for selected northern Manitoba waterbodies

Waterbody	Churchill River[1]		Rat River[2]	Burntwood River[2]	Notigi Lake[3]	Wapisu Lake[4]	Threepoint Lake[5]	Wuskwatim Lake[6]	Leftrook Lake[7]	Lower Nelson River[8]
Study Years	**1995–1996 (pre-weir)**	**1999–2005 (post-weir)**	**2004**	**2004**	**1999–2000**	**1999–2000**	**1998–2000**	**1998–2001**	**1999, 2001**	**2003**
Oligochaeta	1675	235	208	157	823	511	403	143	692	654
Amphipoda	733	227	336	136	476	619	618	321	1,010	0
Ephemeroptera	567	129	135	141	116	367	298	286	191	1
Plecoptera	0	1	2	3	0	1	1	0	0	1
Trichoptera	42	36	59	147	131	34	62	58	19	3
EPT[9]	608	165	196	292	247	402	361	344	210	5
Chironomidae	3,357	1,815	733	641	473	632	633	679	3,444	1,788
Ceratopogonidae	-	26	175	71	124	117	56	62	66	13
Gastropoda	485	249	132	79	67	64	33	118	357	21
Pisidiidae	1,912	784	373	382	51	137	300	288	2,364	71
Total Invertebrates	**9,529**	**3,653**	**2,275**	**1,912**	**2,180**	**2,572**	**2,470**	**2,122**	**8439**	**2,557**

1. After Capar *et al.* (2006).
2. After Zrum and Wyn (2009).
3. After Zrum and Neufeld (2003a).
4. After Zrum *et al.* (2003).
5. After Zrum and Neufeld (2003b).
6. After Manitoba Hydro and NCN (2003) and Zrum and Juliano (2004).
7. After Zrum and Neufeld (2003d).
8. After Capar and Gill (2008).
9. Sum of Ephemeroptera, Plecoptera, and Trichoptera.

Table 4-23: Summary of benthic macroinvertebrate community information for aquatic habitat types sampled in the Split Lake area (Split Lake, the York Landing Arm of Split Lake, and Clark Lake), 1997–2004

Habitat Type[1]	Years	n[2]	Total Abundance (individuals/m^2)				Mean Percent Composition of Major Groups[3]											
			Mean	±SE[4]	Min	Max	Oli	Amp	Hem	Eph	Ple	Tri	EPT	Chi	Cer	Gas	Pis	Oth
S-IEZ ST S M RV	1998, 2000–2002, 2004	13	5,025	891	707	10,262	17.5	11.0	0.0	10.4	0.0	5.0	15.5	40.6	1.5	9.0	3.9	1.1
S ST S M RV	1997–1998, 2001–2002	14	2,316	450	272	6,489	2.6	5.0	0.2	21.4	0.3	2.1	23.9	29.9	1.6	16.7	19.0	1.1
S-IEZ L S M RV	2001–2002	3	2,192	285	1,815	2,750	1.8	0.5	0.0	17.9	0.0	2.8	20.7	15.6	1.3	10.0	49.4	0.7
S L S M RV	1998, 2001	3	2,575	343	1,924	3,087	1.2	1.0	0.0	26.6	0.3	0.4	27.3	7.9	0.3	18.6	43.4	0.3
S-IEZ ST S M NP	1998, 2000	6	3,419	1,025	798	7,964	22.3	19.4	0.1	6.8	0.3	1.8	8.9	17.2	0.8	21.3	8.8	1.2
S ST S M NP	1997–1998, 2000–2002, 2004	41	2,437	317	250	10,174	4.0	12.5	0.1	18.1	0.0	0.7	18.8	18.0	1.5	18.3	25.6	1.2
S L S M NP	1997	2	3,443	-	1,521	5,366	4.1	9.5	0.0	12.2	7.1	4.6	23.9	15.6	0.2	41.2	5.4	0.1
D ST S M NP	1997–1998, 2000–2002	80	3,692	490	43	33,151	3.1	14.5	0.0	16.9	0.1	0.9	17.9	9.3	0.3	28.1	22.5	4.3
D L S M NP	1998, 2001–2002	15	3,516	596	272	8,554	1.1	13.2	0.0	13.0	0.0	4.6	17.7	10.8	0.2	8.7	47.6	0.7

1. S shallow, S-IEZ shallow intermittently exposed, D deep, ST standing water, L low water velocity, S soft substrate, M mineral-based substrate, RV rooted vascular plants, NP no plants.
2. Number of replicates collected per habitat type.
3. Oli Oligochaeta; Amp Amphipoda; Hem Hemiptera; Eph Ephemeroptera; Ple Plecoptera; Tri Trichoptera; EPT sum of Eph, Ple, and Tri; Chi Chironomidae; Cer Ceratopogonidae; Gas Gastropoda; Pis Pisidiidae; Oth Other Groups.
4. SE = standard error.

Table 4-24: Summary of benthic macroinvertebrate community information for aquatic habitat types sampled in Assean Lake, 2001–2002, and 2004

Habitat Type[1]	Years	n[2]	Total Abundance (individuals/m^2)				Mean Percent Composition of Major Groups[3]											
			Mean	±SE[4]	Min	Max	Oli	Amp	Hem	Eph	Plec	Tri	EPT	Chir	Cer	Gas	Pisi	Oth
S ST S M RV	2001–2002, 2004	18	4,217	899	435	17,239	3.4	8.5	0.1	3.7	0.0	1.4	5.2	51.9	3.3	9.7	13.8	4.0
S ST S M NP	2001–2002	11	1,851	449	489	4,283	6.4	3.4	0.1	11.7	0.0	0.9	12.7	46.2	5.3	6.8	17.0	2.1
D ST S M NP	2001–2002	27	1,012	148	283	3,942	4.9	0.1	0.1	14.7	0.0	1.7	16.4	38.5	3.1	7.6	28.3	1.0

1. S shallow, S-IEZ shallow intermittently exposed, D deep, ST standing water, L low water velocity, S soft substrate, M mineral-based substrate, RV rooted vascular plants, NP no plants.
2. Number of replicates collected per habitat type.
3. Oli Oligochaeta; Amp Amphipoda; Hem Hemiptera; Eph Ephemeroptera; Ple Plecoptera; Tri Trichoptera; EPT sum of Eph, Ple, and Tri; Chi Chironomidae; Cer Ceratopogonidae; Gas Gastropoda; Pis Pisidiidae; Oth Other Groups.
4. SE = standard error.

Table 4-25: Summary of plant-dwelling macroinvertebrate community information for aquatic habitat types sampled in the Split Lake area (Clark Lake), 2003–2004

Habitat Type[1]	Years	n[2]	Total Abundance (individuals/m^2)				Mean Percent Composition of Major Groups[3]											
			Mean	±SE[4]	Min	Max	Oli	Amp	Hem	Eph	Plec	Tri	EPT	Chi	Cer	Gas	Pis	Oth
S ST S M RV	2003–2004	9	181	48	17	492	28.9	3.9	2.2	0.3	0.0	0.4	0.7	38.4	0.1	24.8	0.1	1.0

1. S shallow, S-IEZ shallow intermittently exposed, D deep, ST standing water, L low water velocity, S soft substrate, M mineral-based substrate, RV rooted vascular plants, NP no plants.
2. Number of replicates collected per habitat type.
3. Oli Oligochaeta; Amp Amphipoda; Hem Hemiptera; Eph Ephemeroptera; Ple Plecoptera; Tri Trichoptera; EPT sum of Eph, Ple, and Tri; Chi Chironomidae; Cer Ceratopogonidae; Gas Gastropoda; Pis Pisidiidae; Oth Other Groups.
4. SE = standard error.

Table 4-26: Summary of aquatic macrophyte community information for aquatic habitat types sampled in the Split Lake area (Clark Lake), 2003–2004

Habitat Type[1]	Years	n[2]	Total Abundance (g dry weight/m^2)				Mean Percent Composition of Major Species[3]										
			Mean	±SE[4]	Min	Max	Calli	Eleo	Equis	Lemna	Myrio	Poly	Pota	Stuck	Moss	FGA	Other
S ST S M RV	2003–2004	9	16.2	4.8	2.6	46.5	12.2	37.3	0.0	0.0	0.1	0.0	45.3	3.4	0.0	0.0	1.6

1. S shallow, S-IEZ shallow intermittently exposed, D deep, ST standing water, L low water velocity, S soft substrate, M mineral-based substrate, RV rooted vascular plants, NP no plants.
2. Number of replicates collected per habitat type.
3. Calli *Callitriche palustris*; Eleo *Eleocharis palustris*; Equis *Equisetum fluviatile*; Lemna *Lemna trisulca*; Myrio *Myriophyllum sibiricum*; Poly *Polygonum amphibium*; Pota *Potamogeton* spp.; Stuck *Stuckenia* spp.; Moss Aquatic Moss; FGA Filamentous Green Algae; Other Other Species.
4. SE = standard error.

Table 4-27: Summary of benthic macroinvertebrate community information for aquatic habitat types sampled in the Keeyask area, 1999–2004

Habitat Type[1]	Years	n[2]	Total Abundance (individuals/m²)				Mean Percent Composition of Major Groups[3]											
			Mean	±SE[4]	Min	Max	Oli	Amp	Hem	Eph	Ple	Tri	EPT	Chi	Cer	Gas	Pis	Oth
BWI-IEZ ST S M RV	2004	2	4,761	-	2,130	7,391	17.1	3.6	0.0	1.5	0.0	0.5	1.9	63.2	0.0	12.6	0.2	1.4
S-IEZ ST S M RV	1999, 2001–2002, 2004	17	5,900	1,217	424	15,957	23.4	16.3	0.0	3.8	0.0	0.4	4.1	39.1	0.8	10.8	4.4	1.0
S ST S M RV	2001–2002	6	1,399	298	522	2,522	1.9	5.8	0.0	32.3	0.0	0.5	32.8	21.9	1.6	11.5	23.7	0.8
S-IEZ L S M RV	1999	1	1,060	-	-	-	3.4	2.2	1.1	27.0	0.0	0.0	27.0	34.8	5.6	15.7	6.7	3.4
S L S M RV	1999	2	3,440	-	1,857	5,024	18.0	0.0	0.0	5.5	0.0	0.0	5.5	36.4	2.9	31.8	4.7	0.7
BWI-IEZ ST S M NP	2001–2002	3	3,768	1,750	880	6,924	4.3	47.7	0.0	3.6	0.0	1.2	4.7	33.8	2.5	4.3	1.8	0.8
S-IEZ ST S M NP	2002, 2004	4	3,603	560	2,185	4,815	9.6	4.2	0.0	9.0	0.0	2.1	11.1	66.1	2.8	3.6	1.9	0.8
S ST S M NP	2001–2002	4	1,546	184	1,337	2,098	3.0	1.1	0.0	38.8	0.0	1.1	39.9	23.6	0.5	16.2	15.6	0.2
S L S M NP	2002	3	2,217	1,025	196	3,522	4.6	0.7	0.0	17.5	0.3	2.3	20.1	8.3	1.1	18.0	45.9	1.3
D ST S M NP	2001–2002	3	917	672	239	2,261	4.0	0.0	0.0	37.9	0.0	0.4	38.3	23.7	2.4	5.5	24.9	1.2
D L S M NP	1999, 2001–2002	16	3,026	775	0	11,798	5.3	4.3	0.0	10.2	0.1	14.2	24.4	7.2	0.6	27.8	28.8	1.5

1. S shallow, S-IEZ shallow intermittently exposed, BWI-IEZ backwater inlet intermittently exposed, D deep, ST standing water, L low water velocity, S soft substrate, M mineral-based substrate, RV rooted vascular plants, NP no plants.
2. Number of replicates collected per habitat type.
3. Oli Oligochaeta; Amp Amphipoda; Hem Hemiptera; Eph Ephemeroptera; Ple Plecoptera; Tri Trichoptera; EPT sum of Eph, Ple, and Tri; Chi Chironomidae; Cer Ceratopogonidae; Gas Gastropoda; Pis Pisidiidae; Oth Other Groups.
4. SE = standard error.

Table 4-28: Summary of plant-dwelling macroinvertebrate community information for aquatic habitat types sampled in the Keeyask area, 2001–2004

Habitat Type[1]	Years	n[2]	Total Abundance (individuals/m^2)				Mean Percent Composition of Major Groups[3]											
			Mean	±SE[4]	Min	Max	Oli	Amp	Hem	Eph	Ple	Tri	EPT	Chi	Cer	Gas	Pis	Oth
BWI-IEZ ST S M RV	2001–2002, 2004	12	494	160	14	1,561	25.7	1.5	4.5	0.4	0.0	1.4	1.8	30.5	0.0	35.0	0.1	0.9
S ST S M RV	2001–2004	47	367	83	0	2,694	10.3	16.8	2.7	0.7	0.0	1.9	2.7	39.3	0.0	21.6	1.9	4.8
S L S M RV	2001–2004	20	600	150	23	2,611	25.4	1.5	4.5	0.4	0.0	1.4	1.8	31.4	0.0	34.4	0.1	0.9

1. S shallow, BWI-IEZ backwater inlet intermittently exposed, ST standing water, L low water velocity, S soft substrate, M mineral-based substrate, RV rooted vascular plants.
2. Number of replicates collected per habitat type.
3. Oli Oligochaeta; Amp Amphipoda; Hem Hemiptera; Eph Ephemeroptera; Plec Plecoptera; Tri Trichoptera; EPT sum of Eph, Plec, and Tri; Chi Chironomidae; Cer Ceratopogonidae; Gas Gastropoda; Pis Pisidiidae; Oth Other Groups
4. SE = standard error.

Table 4-29: Summary of aquatic macrophyte community information for aquatic habitat types sampled in the Keeyask area, 2001–2004

Habitat Type[1]	Years	n[2]	Total Abundance (g dry weight/m^2)				Mean Percent Composition of Major Species[3]										
			Mean	±SE[4]	Min	Max	Calli	Eleo	Equis	Lemna	Myrio	Poly	Pota	Stuck	Moss	FGA	Other
BWI-IEZ ST S M RV	2001–2002, 2004	12	56.7	10.8	5.0	135.3	0.0	13.9	0.0	12.9	41.8	0.0	10.1	14.9	0.3	0.0	6.1
S ST S M RV	2001–2004	47	27.9	3.7	0.0	94.5	0.0	10.0	0.8	12.7	3.2	2.1	50.5	16.0	0.5	1.3	2.9
S L S M RV	2001–2004	20	30.2	6.3	1.2	88.3	0.0	0.0	0.0	0.0	0.0	0.0	58.4	29.5	4.0	7.7	0.3

1. S shallow, BWI-IEZ backwater inlet intermittently exposed, ST standing water, L low water velocity, S soft substrate, M mineral-based substrate, RV rooted vascular plants.
2. Number of replicates collected per habitat type.
3. Calli *Callitriche palustris*; Eleo *Eleocharis palustris*; Equis *Equisetum fluviatile*; Lemna *Lemna trisulca*; Myrio *Myriophyllum sibiricum*; Poly *Polygonum amphibium*; Pota *Potamogeton* spp.; Stuck *Stuckenia* spp.; Moss Aquatic Moss; FGA Filamentous Green Algae; Other Other Species.
4. SE = standard error.

Table 4-30: Mean drifting invertebrate density and community composition information for large drift traps set in the Aquatic Environment Study Area in comparable sampling periods during the open-water season, 2003 and 2004

Location of Drift Traps	**Upstream of Birthday Rapids**	**Downstream of Birthday Rapids**	**Upstream of Gull Rapids (at the downstream end of Gull Lake)**	**Downstream of Gull Rapids (near the base of Gull Rapids)**	**Downstream of the Kettle GS**
n[1]	**20**	**20**	**20**	**16**	**16**
Mean Drift Density (individuals/100 m^3)					
Annelida (aquatic earthworms)	0.074	2.090	0.134	2.077	0.002
Crustacea (crustaceans)	0.383	1.008	0.543	0.365	0.619
Acarina (water mites)	0.023	0.037	0.021	0.136	0.027
Mollusca (snails and clams)	0.092	0.328	0.112	0.294	0.015
Platyhelminthes (flatworms)	0.000	0.000	0.000	0.000	0.006
Hydrozoa (hydrozoans)	0.002	0.004	0.005	0.000	0.016
Ephemeroptera (mayflies)	3.840	12.910	36.897	5.318	1.339
Trichoptera (caddisflies)	2.215	7.735	3.393	4.383	1.418
Plecoptera (stoneflies)	0.068	1.403	2.251	0.370	0.008
Diptera (true flies, including chironomids)	1.798	6.297	4.303	5.854	3.887
Total Aquatic Insects	**8.712**	**31.263**	**48.887**	**18.230**	**6.736**
Total Aquatic Invertebrates	**9.287**	**34.731**	**49.702**	**21.100**	**7.419**
Percent Composition (%)					
Annelida (aquatic earthworms)	0.801	6.017	0.270	9.842	0.025
Crustacea (crustaceans)	4.129	2.904	1.092	1.732	8.338
Acarina (water mites)	0.245	0.107	0.042	0.645	0.357
Mollusca (snails and clams)	0.992	0.946	0.225	1.395	0.200
Platyhelminthes (flatworms)	0.004	0.000	0.000	0.000	0.075

Table 4-30: Mean drifting invertebrate density and community composition information for large drift traps set in the Aquatic Environment Study Area in comparable sampling periods during the open-water season, 2003 and 2004

Location of Drift Traps	**Upstream of Birthday Rapids**	**Downstream of Birthday Rapids**	**Upstream of Gull Rapids (at the downstream end of Gull Lake)**	**Downstream of Gull Rapids (near the base of Gull Rapids)**	**Downstream of the Kettle GS**
n[1]	**20**	**20**	**20**	**16**	**16**
Hydrozoa (hydrozoans)	0.021	0.011	0.011	0.000	0.213
Ephemeroptera (mayflies)	41.350	37.172	74.235	25.204	18.045
Trichoptera (caddisflies)	23.854	22.271	6.827	20.771	19.111
Plecoptera (stoneflies)	0.733	4.041	4.530	1.753	0.108
Diptera (true flies, including chironomids)	19.364	18.132	8.657	27.742	52.392
Total Aquatic Insects	**93.809**	**90.015**	**98.359**	**86.397**	**90.792**

1. Number of samples collected per area for 2003 and 2004 sampling periods combined.

Table 4-31: Summary of benthic macroinvertebrate community information for aquatic habitat types sampled in the Stephens Lake area, 2001–2006

Habitat Type[1]	Years	n[2]	Total Abundance (individuals/m^2)				Mean Percent Composition of Major Groups[3]											
			Mean	±SE[4]	Min	Max	Oli	Amp	Hem	Eph	Ple	Tri	EPT	Chir	Cer	Gas	Pis	Oth
S-IEZ ST S M RV	2002, 2004	5	2,073	841	678	5,174	15.7	4.8	0.0	3.1	0.0	7.1	10.3	52.9	7.2	5.8	1.4	1.8
S ST S M RV	2001–2002	6	2,417	335	1,574	3,435	2.4	1.3	0.1	19.4	0.0	1.4	20.9	52.2	4.7	15.0	3.0	0.4
S ST S O RV	2001	1	7,109	-	-	-	0.0	25.1	0.0	28.1	0.0	0.9	29.1	42.4	1.1	1.5	0.8	0.2
S-IEZ ST S M NP	2002, 2004	9	2,399	606	43	5,791	15.5	4.9	0.0	3.9	0.0	1.3	5.2	56.6	1.0	15.3	1.2	0.4
S ST S M NP	2001–2002, 2004, 2006	22	1,611	287	148	5,217	12.8	0.8	0.1	17.5	0.0	1.6	19.0	50.9	1.8	7.8	4.5	2.1
S-IEZ ST S O NP	2006	5	8,331	2,949	2,696	19,130	4.8	1.7	0.0	3.6	0.0	0.4	4.1	83.1	0.9	1.7	3.3	0.5
S ST S O NP	2006	15	2,794	658	435	10,435	4.5	17.8	0.0	10.1	0.0	1.2	11.3	50.9	2.4	2.0	9.5	1.6
D ST S M NP	2001–2002, 2004	21	2,216	365	148	6,804	7.8	11.3	0.0	29.6	0.0	1.5	31.0	39.0	2.4	1.4	2.2	4.8
D ST S O NP	2001–2002	8	2,760	902	409	6,217	7.6	67.5	0.0	3.3	0.0	0.4	3.6	20.4	0.4	0.0	0.4	0.1

1. S shallow, S-IEZ shallow intermittently exposed, D deep, ST standing water, S soft substrate, M mineral-based substrate, O organic-based substrate, RV rooted vascular plants, NP no plants.
2. Number of replicates collected per habitat type.
3. Oli Oligochaeta; Amp Amphipoda; Hem Hemiptera; Eph Ephemeroptera; Ple Plecoptera; Tri Trichoptera; EPT sum of Eph, Ple, and Tri; Chi Chironomidae; Cer Ceratopogonidae; Gas Gastropoda; Pis Pisidiidae; Oth Other Groups.
4. SE = standard error.

Table 4-32: Summary of plant-dwelling macroinvertebrate community information for aquatic habitat types sampled in the Stephens Lake area, 2005–2006

Habitat Type[1]	Years	n[2]	Total Abundance (individuals/m^2)				Mean Percent Composition of Major Groups[3]											
			Mean	±SE[4]	Min	Max	Oli	Amp	Hem	Eph	Ple	Tri	EPT	Chi	Cer	Gas	Pis	Oth
S-IEZ ST S M RV	2005–2006	22	859	422	10	8,919	18.3	2.4	0.5	0.3	0.0	3.6	3.9	28.9	0.1	2.6	0.0	43.2[5]
S ST S M RV	2005	2	90	-	38	143	0.0	3.9	5.3	0.0	0.0	3.9	3.9	82.9	0.0	3.9	0.0	0.0
S-IEZ ST S O RV	2005–2006	4	224	154	36	686	13.0	8.0	0.3	0.3	0.0	0.5	0.8	32.6	0.3	5.6	0.3	39.3[6]
S ST S O RV	2006	1	721	-	-	-	7.6	12.2	0.0	0.7	0.0	0.3	1.0	20.5	0.0	14.2	0.0	44.6[7]

1. S shallow, S-IEZ shallow intermittently exposed, ST standing water, S soft substrate, M mineral-based substrate, O organic-based substrate, RV rooted vascular plants.
2. Number of replicates collected per habitat type.
3. Oli Oligochaeta; Amp Amphipoda; Hem Hemiptera; Eph Ephemeroptera; Ple Plecoptera; Tri Trichoptera; EPT sum of Eph, Ple, and Tri; Chi Chironomidae;
Cer Ceratopogonidae; Gas Gastropoda; Pis Pisidiidae; Oth Other Groups.
4. SE = standard error.
5. Hydrozoa (41.1 %).
6. Hydrozoa (34.7 %).
7. Hydrozoa (28.4 %); Diplostraca (12.9%).

Table 4-33: Summary of aquatic macrophyte community information for aquatic habitat types sampled in the Stephens Lake area, 2005–2006

Habitat Type[1]	Years	n[2]	Total Abundance (g dry weight/m^2)				Mean Percent Composition of Major Species[3]										
			Mean	±SE[4]	Min	Max	Calli	Eleo	Equis	Lemna	Myrio	Poly	Pota	Stuck	Moss	FGA	Other
S-IEZ ST S M RV	2005–2006	22	55.7	7.7	8.9	174.7	0.0	0.0	0.0	0.0	17.9	0.0	60.3	16.2	0.3	0.0	5.3
S ST S M RV	2005	2	71.9	-	31.0	112.8	0.0	0.0	0.0	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0
S-IEZ ST S O RV	2005–2006	4	27.3	13.4	2.4	61.3	0.0	0.0	0.0	0.0	19.6	56.1	5.8	0.0	16.4	0.0	2.1
S ST S O RV	2006	1	91.0	-	-	-	0.0	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0	0.0	0.0

1. S shallow, S-IEZ shallow intermittently exposed, ST standing water, S soft substrate, M mineral-based substrate, O organic-based substrate, RV rooted vascular plants.
2. Number of replicates collected per habitat type.
3. Calli *Callitriche palustris*; Eleo *Eleocharis palustris*; Equis *Equisetum fluviatile*; Lemna *Lemna trisulca*; Myrio *Myriophyllum sibiricum*; Poly *Polygonum amphibium*; Pota *Potamogeton* spp.; Stuck *Stuckenia* spp.; Moss Aquatic Moss; FGA Filamentous Green Algae; Other Other Species.
4. SE = standard error.

Table 4-34: Total benthic macroinvertebrate abundance in the existing environment (EE) and at post-Project (PP) time steps under different generating station operating scenarios

	EE[1]	Year 1 PP			Year 5 PP			Year 15 PP			Year 30 PP		
		Peaking Mode[2]		Base Loaded[3]	Peaking Mode		Base Loaded	Peaking Mode		Base Loaded	Peaking Mode		Base Loaded
		MOL[4]	FSL[5]		MOL	FSL		MOL	FSL		MOL	FSL	
Total Abundance (ind/habitat type)	4.5E+10	1.1E+11	1.2E+11	1.3E+11	1.1E+11	1.2E+11	1.4E+11	1.3E+11	1.5E+11	1.8E+11	1.2E+11	1.5E+11	1.8E+11
Loss(-)/ Gain(+) Total Abundance (ind/habitat type)	0.0E+00	6.3E+10	7.6E+10	8.8E+10	6.5E+10	7.9E+10	9.4E+10	8.0E+10	1.1E+11	1.4E+11	7.8E+10	1.1E+11	1.4E+11
Percent Loss(-)/ Gain(+) Total Abundance (%)	0.0	141.7	169.1	196.3	144.1	177.2	210.1	179.7	242.1	304.5	174.6	243.6	312.6

1. At the 95th percentile flow.
2. Assumes weekly cycling.
3. Assumes no cycling (*i.e.*, Full Supply Level [FSL] with no intermittently exposed zone [IEZ]).
4. Minimum Operating Level – no IEZ.
5. Includes IEZ.

Table 4-35: Total plant-dwelling macroinvertebrate abundance in the existing environment (EE) and at post-Project (PP) time steps under different generating station operating scenarios

	EE[1]	Year 1 PP			Year 5 PP			Year 15 PP			Year 30 PP		
		Peaking Mode[2]		Base Loaded[3]	Peaking Mode		Base Loaded	Peaking Mode		Base Loaded	Peaking Mode		Base Loaded
		MOL[4]	FSL[5]		MOL	FSL		MOL	FSL		MOL	FSL	
Total Abundance (ind/habitat type)	9.0E+08	6.9E+05	5.2E+06	9.8E+06	6.9E+05	5.2E+06	9.8E+06	4.1E+07	9.5E+07	1.5E+08	2.0E+08	4.7E+08	7.4E+08
Loss(-)/ Gain(+) Total Abundance (ind/habitat type)	0.0E+00	-9.0E+08	-9.0E+08	-8.9E+08	-9.0E+08	-9.0E+08	-8.9E+08	-8.6E+08	-8.1E+08	-7.6E+08	-7.1E+08	-4.4E+08	-1.7E+08
Percent Loss(-)/ Gain(+) Total Abundance (%)	0.0	-99.9	-99.4	-98.9	-99.9	-99.4	-98.9	-95.5	-89.5	-83.6	-78.1	-48.3	-18.5

1. At the 95th percentile flow.
2. Assumes weekly cycling.
3. Assumes no cycling (*i.e.*, Full Supply Level [FSL] with no intermittently exposed zone [IEZ]).
4. Minimum Operating Level – no IEZ.
5. Includes IEZ.

Table 4-36A: Residual effects on the aquatic macroinvertebrate community: construction period

Environmental Effect	Mitigation	Residual Effect(s)
Downstream of the Outlet of Clark Lake Aquatic macroinvertebrate distribution and/or abundance would be affected by: changes in water quality (increases in concentrations of TSS and metals; decreases in DO concentrations); and changes in physical attributes of aquatic habitat (downstream sedimentation, loss of habitat in footprint of supporting infrastructure)	A number of measures will be implemented to minimize effects of construction on water quality and aquatic habitat	Adverse, moderate magnitude, small to medium extent, and long-term decrease in aquatic macroinvertebrate distribution and/or abundance and in the production of drifting invertebrates (predominantly larval insects) from Gull Rapids
South Access Road Stream Crossings Benthic macroinvertebrate community would be affected by infilling of aquatic habitat at crossings	A number of measures will be implemented to minimize effects of construction on water quality	Adverse, large magnitude, small extent, and long-term loss of benthic macroinvertebrates at culvert locations

Table 4-36B: Residual effects on the aquatic macroinvertebrate community: operation period

Environmental Effect	Mitigation	Residual Effect(s)
Upstream of Outlet of Clark Lake No effect	Project design to avoid water level effects to Split Lake	None
Outlet of Clark Lake to Generating Station Aquatic macroinvertebrate distribution, abundance, and/or community composition would be affected by: flooding (loss of existing habitats, creation of new habitats); reduction in moderate and high water velocity aquatic habitats; conversion of existing hard substrates to silt/clay due to sedimentation in Gull Lake; increase in the frequency of water level fluctuations (reduction in overall magnitude); conversion of tributary habitat to bays; reduction in the extent and severity of ice scour; changes in surface water quality in off-current areas (e.g., decrease in dissolved oxygen concentrations)	Constructing reefs for fish spawning habitat which would also be colonized by aquatic macroinvertebrates	Positive (benthos) to adverse (plant-dwelling), moderate to large magnitude, small to medium extent, and long-term duration increase in benthos and decrease in plant-dwelling macroinvertebrates Adverse, large (small extent) to small (medium extent) magnitude, and long-term duration decrease in the production of drifting invertebrates (predominantly larval insects)
Downstream of Generating Station Aquatic macroinvertebrates would be affected by: alteration of flow patterns, water velocities and depths; reduction in the extent and severity of ice scour in the portion of the Nelson River to the inlet of Stephens Lake; direct loss of aquatic habitat due to dewatering of Gull Rapids and footprint of GS; change in density of drifting macroinvertebrates entering from Gull Lake; and no effect in Stephens Lake proper	Constructing fish spawning structure in tailrace (approximately 3 ha in size) which would also be colonized by aquatic macroinvertebrates	Adverse, small to moderate magnitude, small extent, and long-term duration decrease in the production of drifting invertebrates (predominantly larval insects)
North and South Access Road Stream Crossings Benthic macroinvertebrate community would be affected by infilling of aquatic habitat at crossings and inputs of sediment from erosion	Clear span bridge on Looking Back Creek; placement of culverts as per Manitoba Stream Crossing Guidelines; effective erosion control measures	Adverse, large magnitude, small extent, and long-term loss of benthic macroinvertebrates at culvert locations

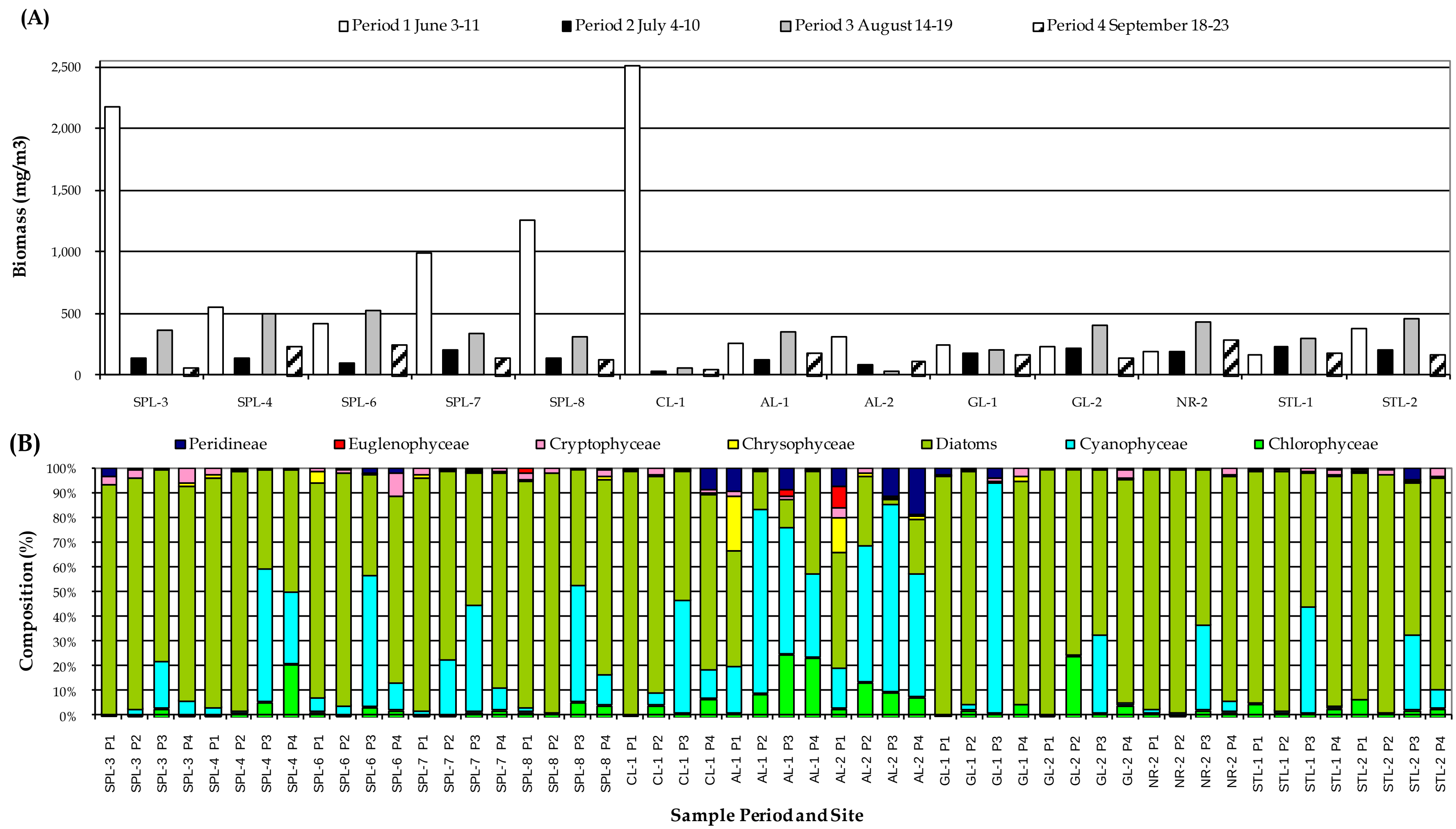


Figure 4-1: Phytoplankton community biomass (A) and composition (B) in samples collected from the Aquatic Environment Study Area in 2001

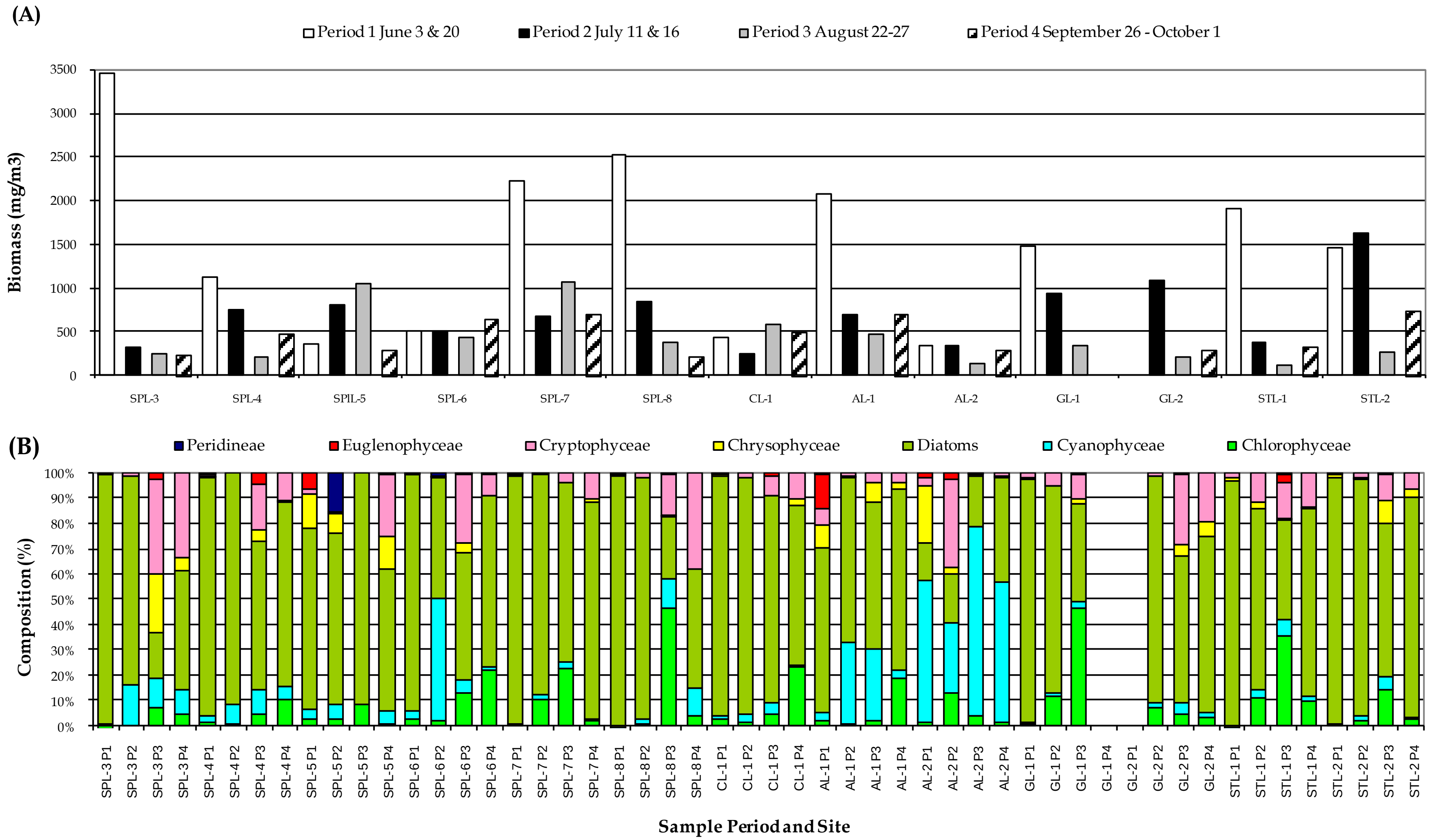


Figure 4-2: Phytoplankton community biomass (A) and composition (B) in samples collected from the Aquatic Environment Study Area in 2002

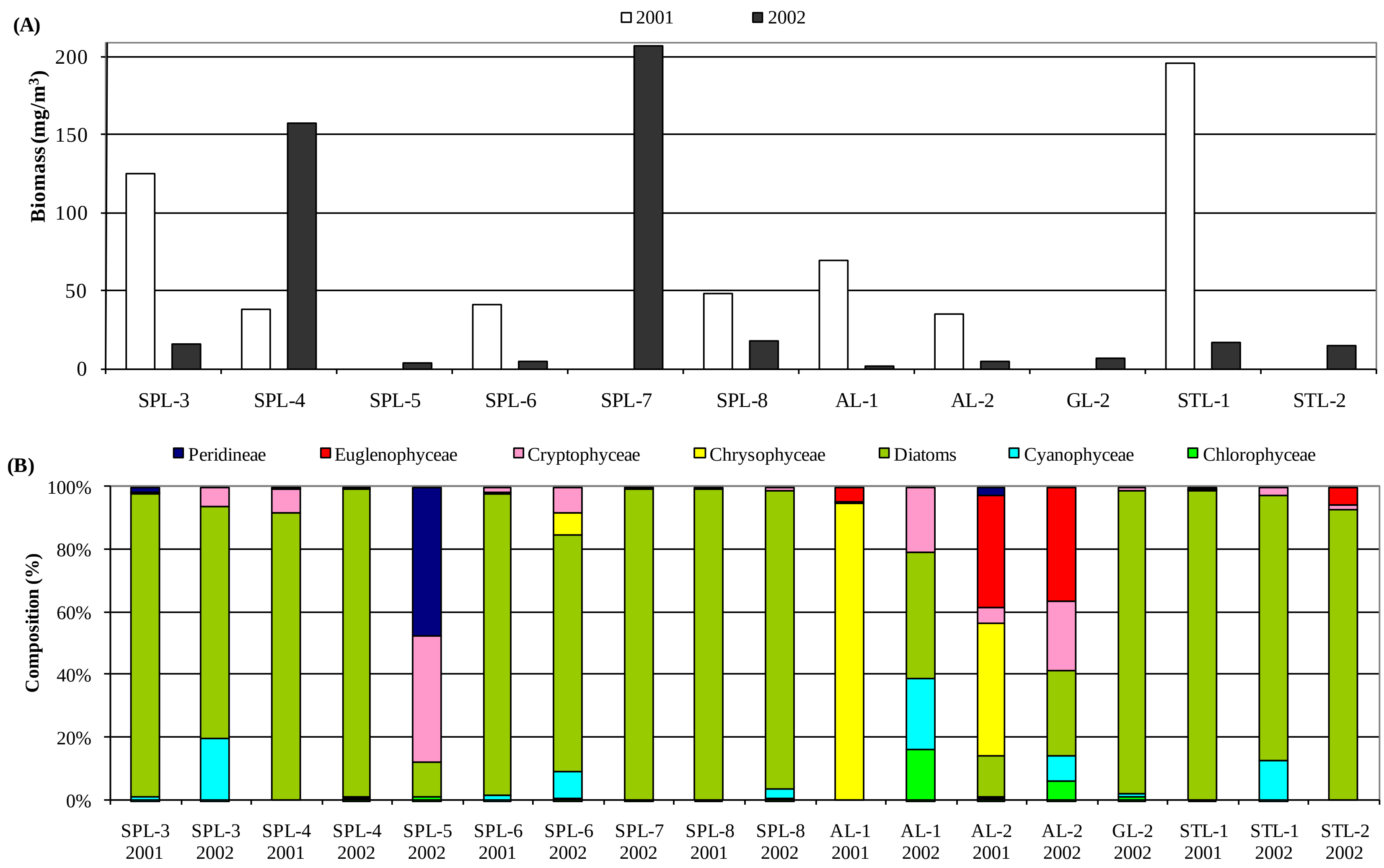


Figure 4-3: Phytoplankton community biomass (A) and composition (B) in samples collected from the Aquatic Environment Study Area in March, 2001 and 2002 (under ice- cover)

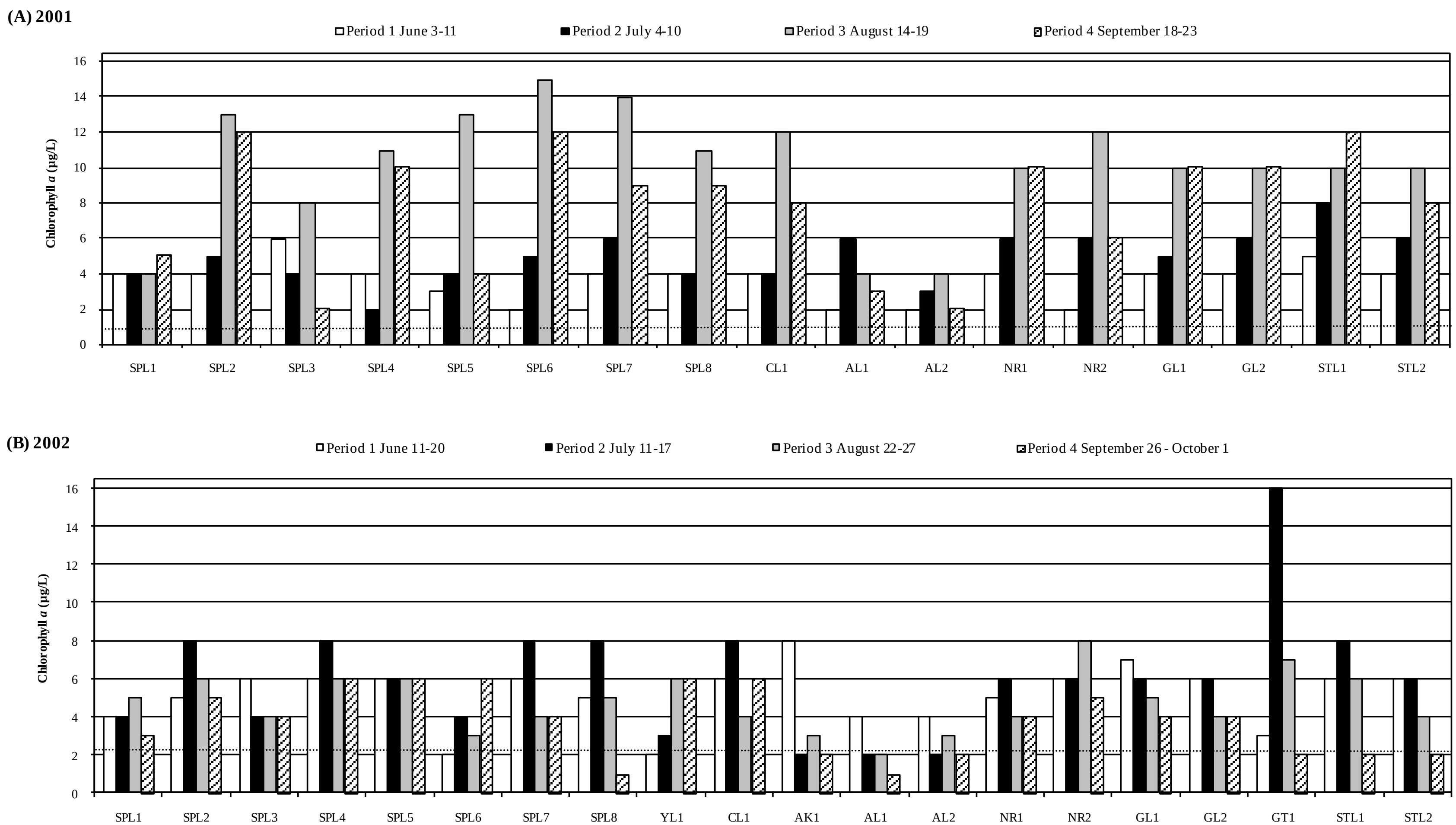
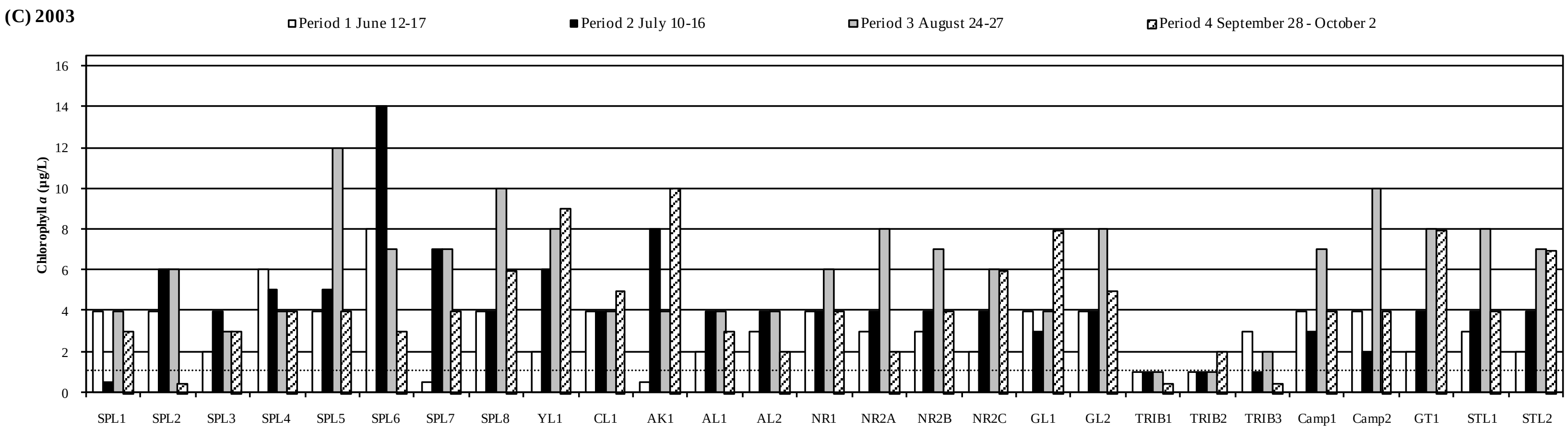


Figure 4-4: Chlorophyll *a* concentration (µg/L) in samples collected from the Aquatic Environment Study Area in: (A) 2001; (B) 2002; (C) 2003; and (D) 2004. Dashed line indicates the detection limit of the laboratory analysis method (samples which were below the limit of detection are plotted at half the detection limit)



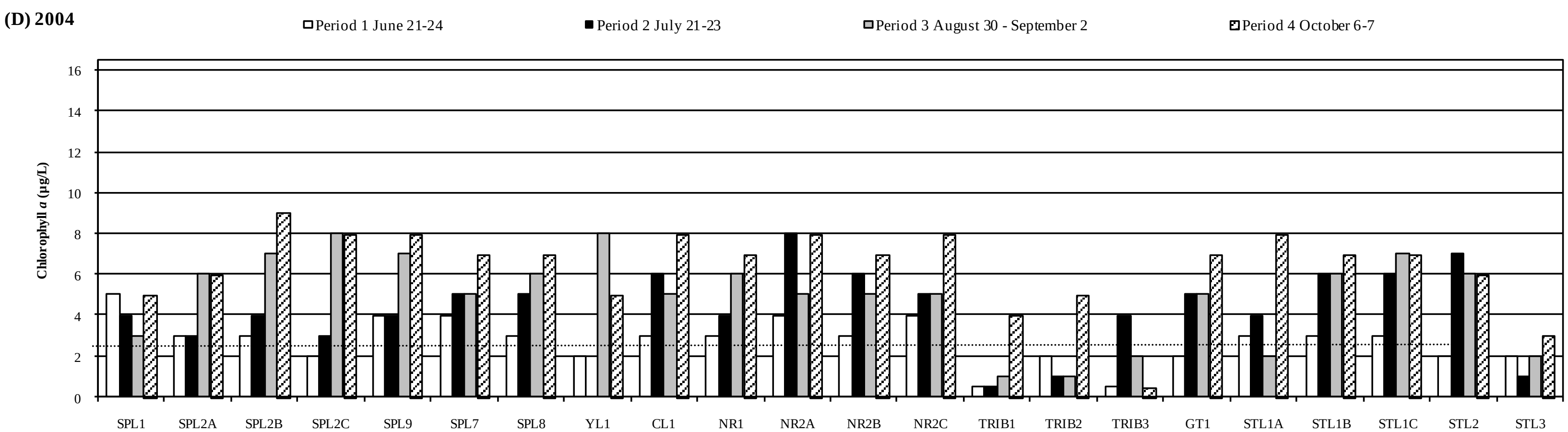


Figure 4-4: Chlorophyll a concentration (µg/L) in samples collected from the Aquatic Environment Study Area in: (A) 2001; (B) 2002; (C) 2003; and (D) 2004. Dashed line indicates the detection limit of the laboratory analysis method (samples which were below the limit of detection are plotted at half the detection limit)

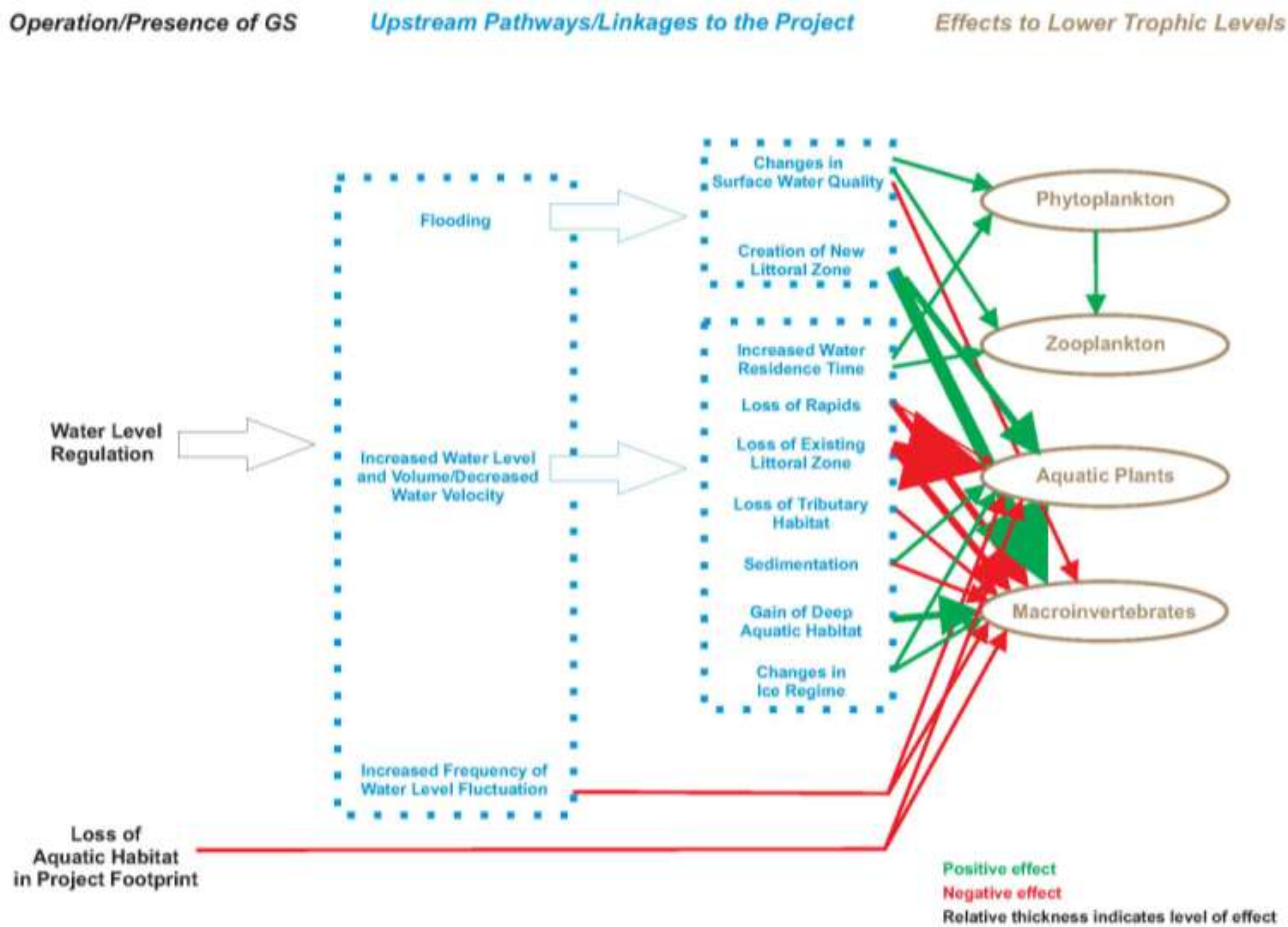


Figure 4-5: Operation-related pathways (*i.e.*, linkages to the Project) that were assessed for potential effects to the lower trophic level communities: Upstream of the Keeyask Generating Station

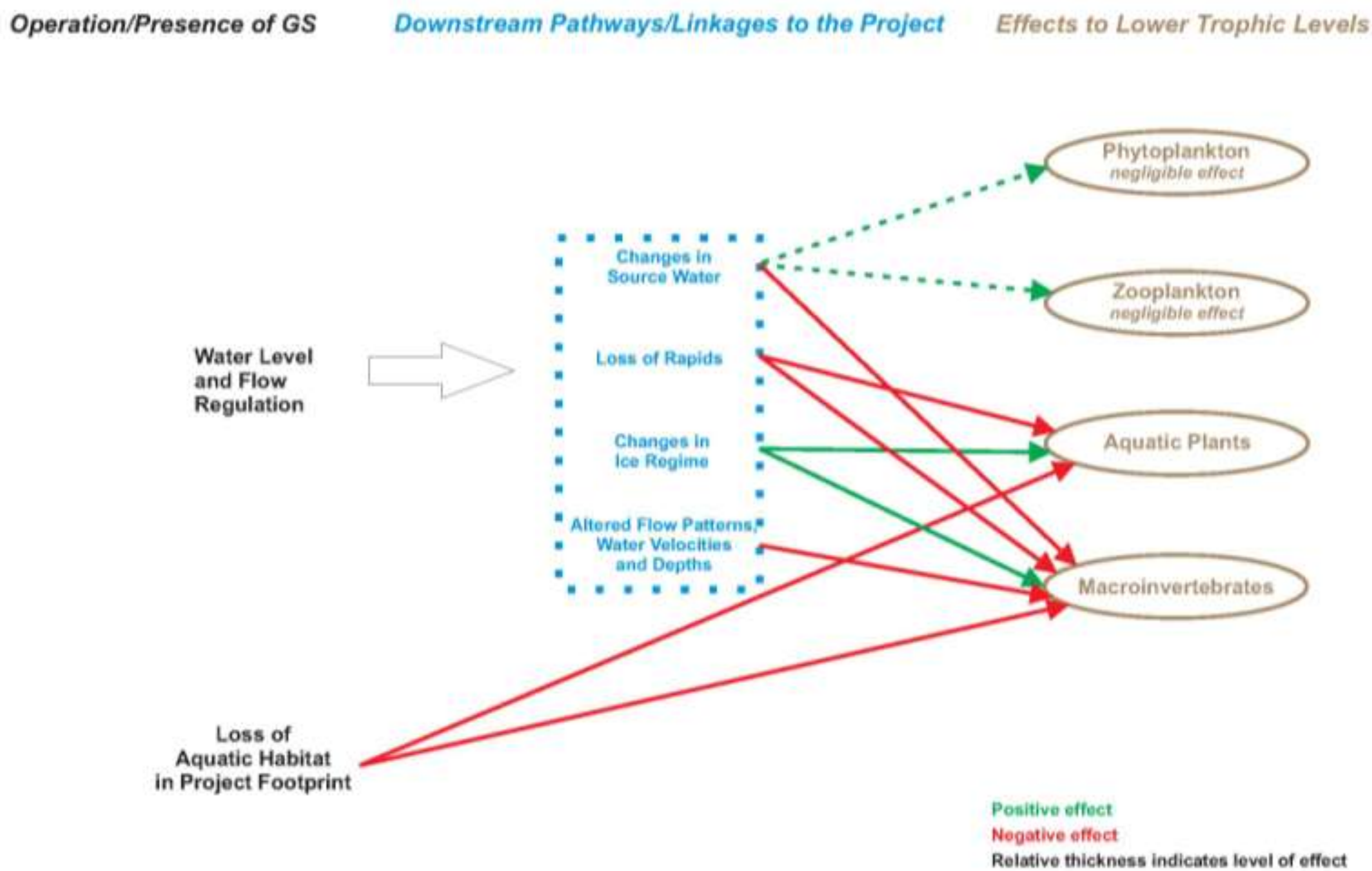


Figure 4-6: Operation-related pathways (*i.e.,* linkages to the Project) that were assessed for potential effects to the lower trophic level communities: Downstream of the Keeyask Generating Station

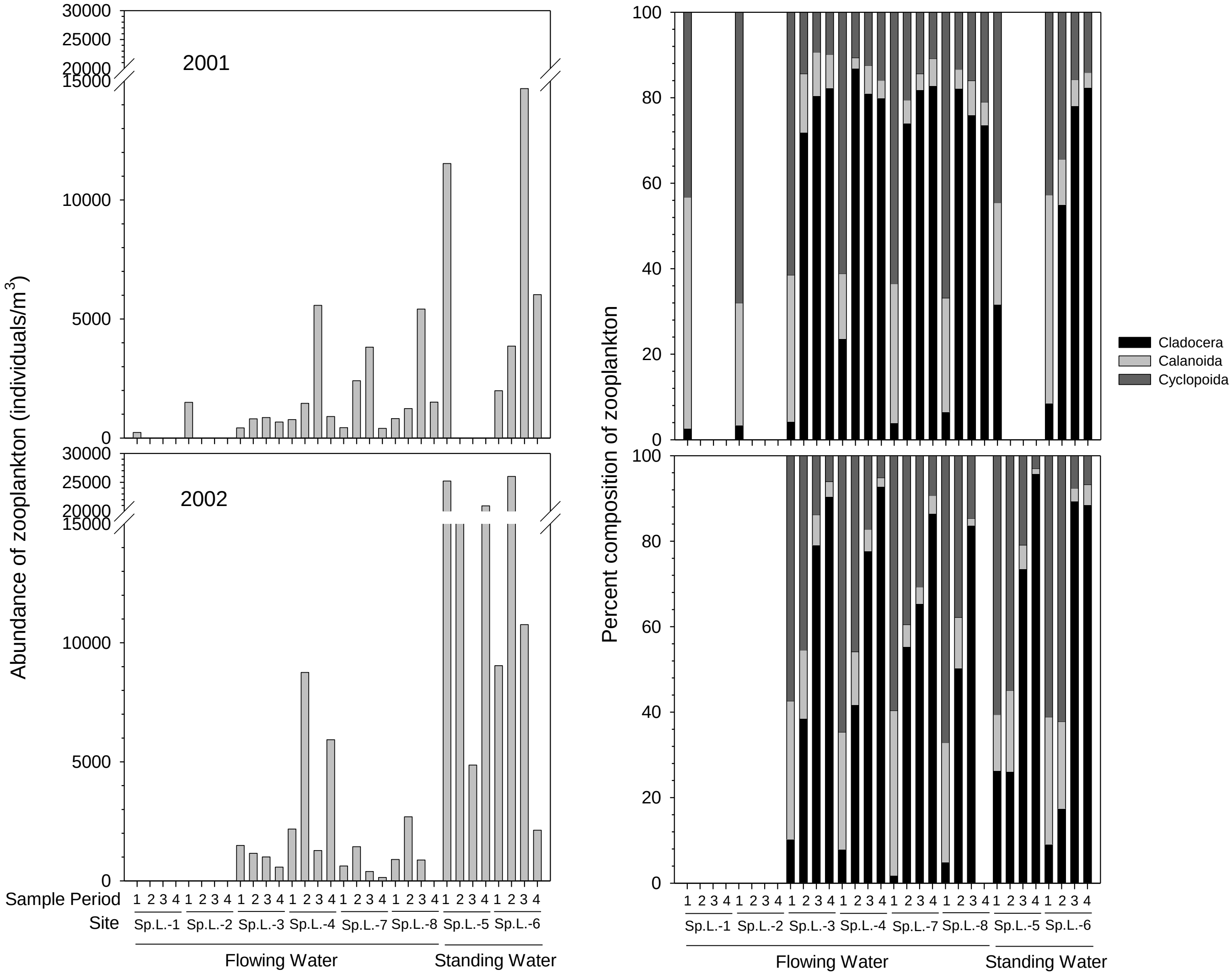


Figure 4-7: Total abundance and percent composition of zooplankton collected in vertical net tows from Split Lake, 2001–2002

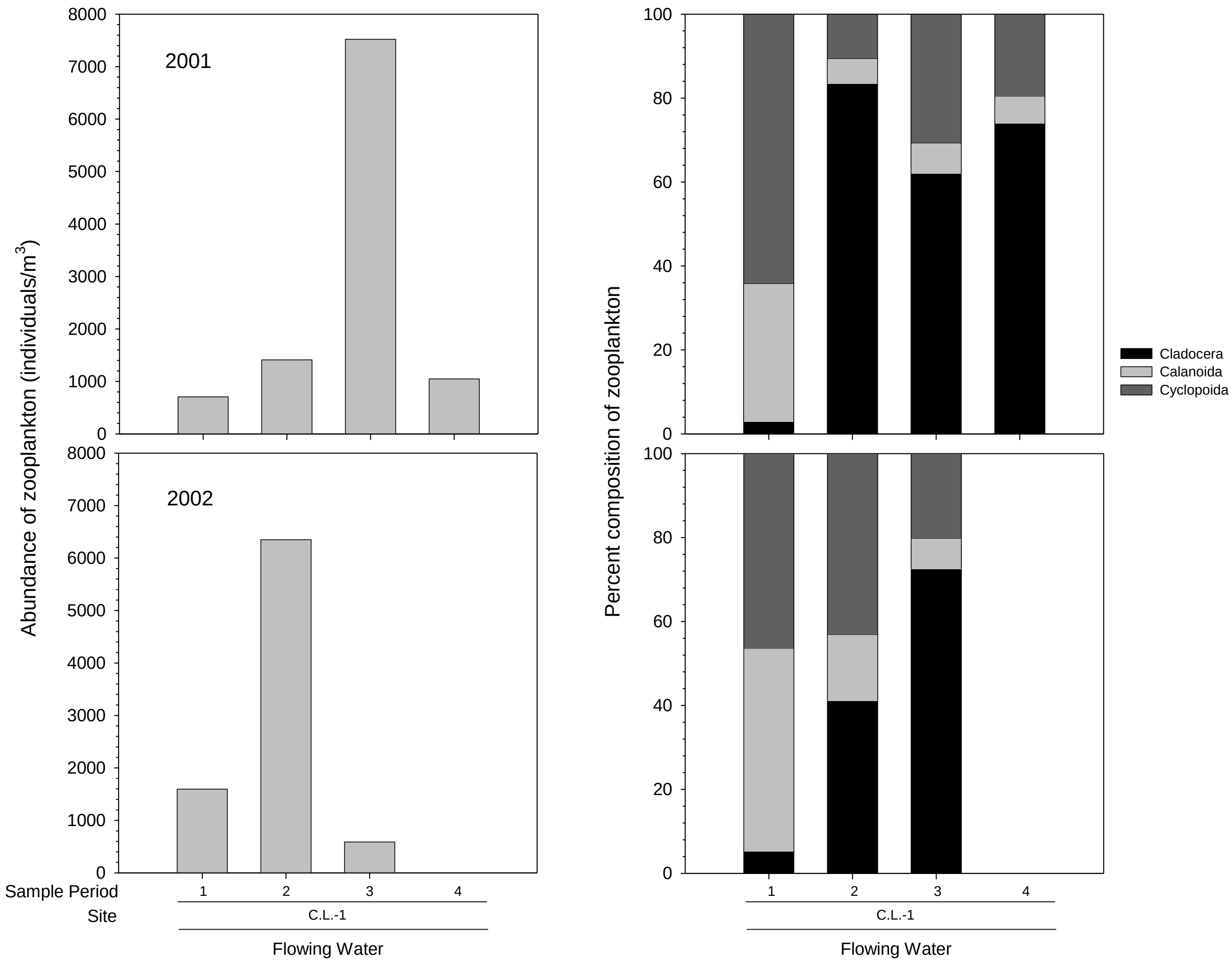


Figure 4-8: Total abundance and percent composition of zooplankton collected in vertical net tows from Clark Lake, 2001–2002

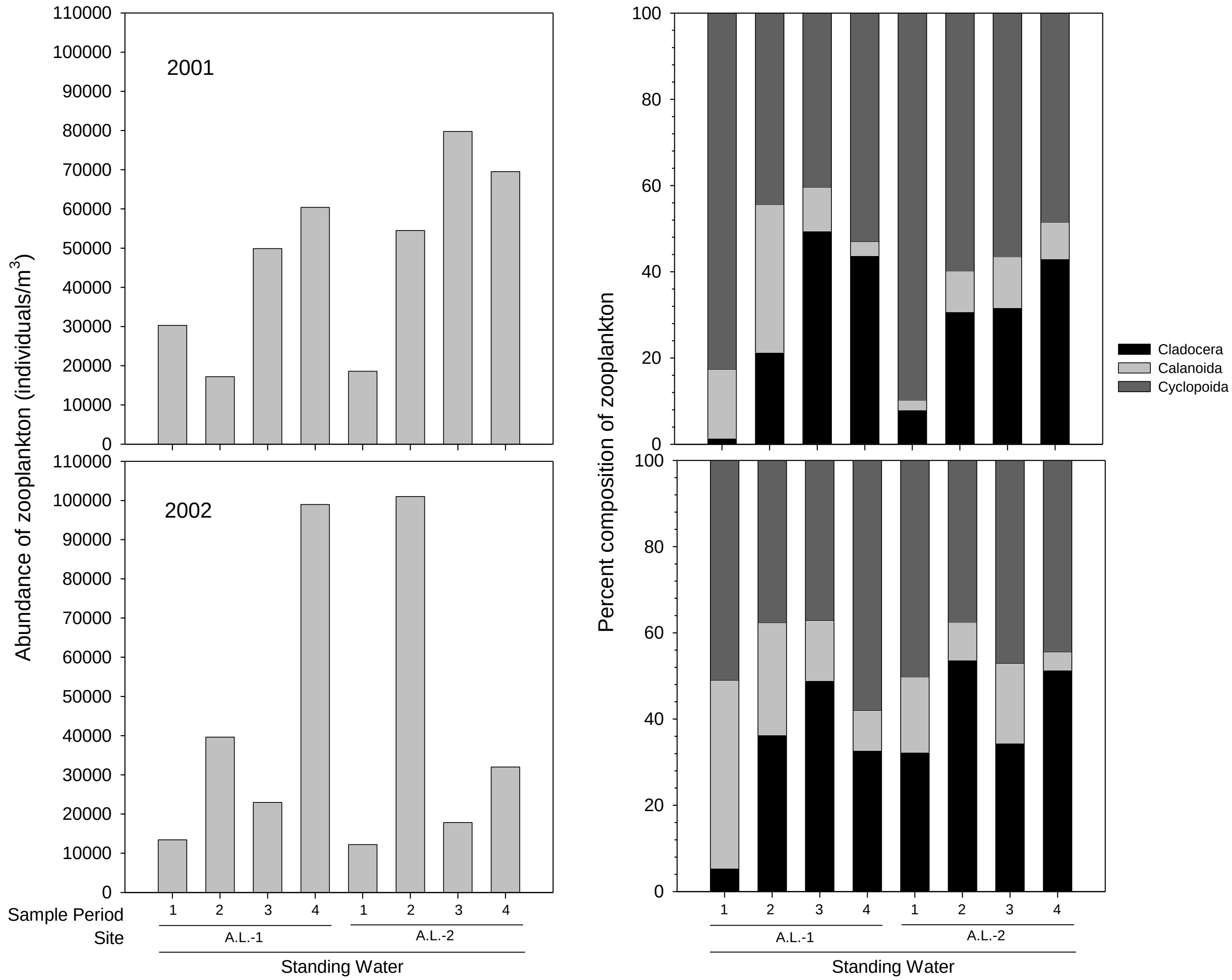


Figure 4-9: Total abundance and percent composition of zooplankton collected in vertical net tows from Assean Lake, 2001–2002

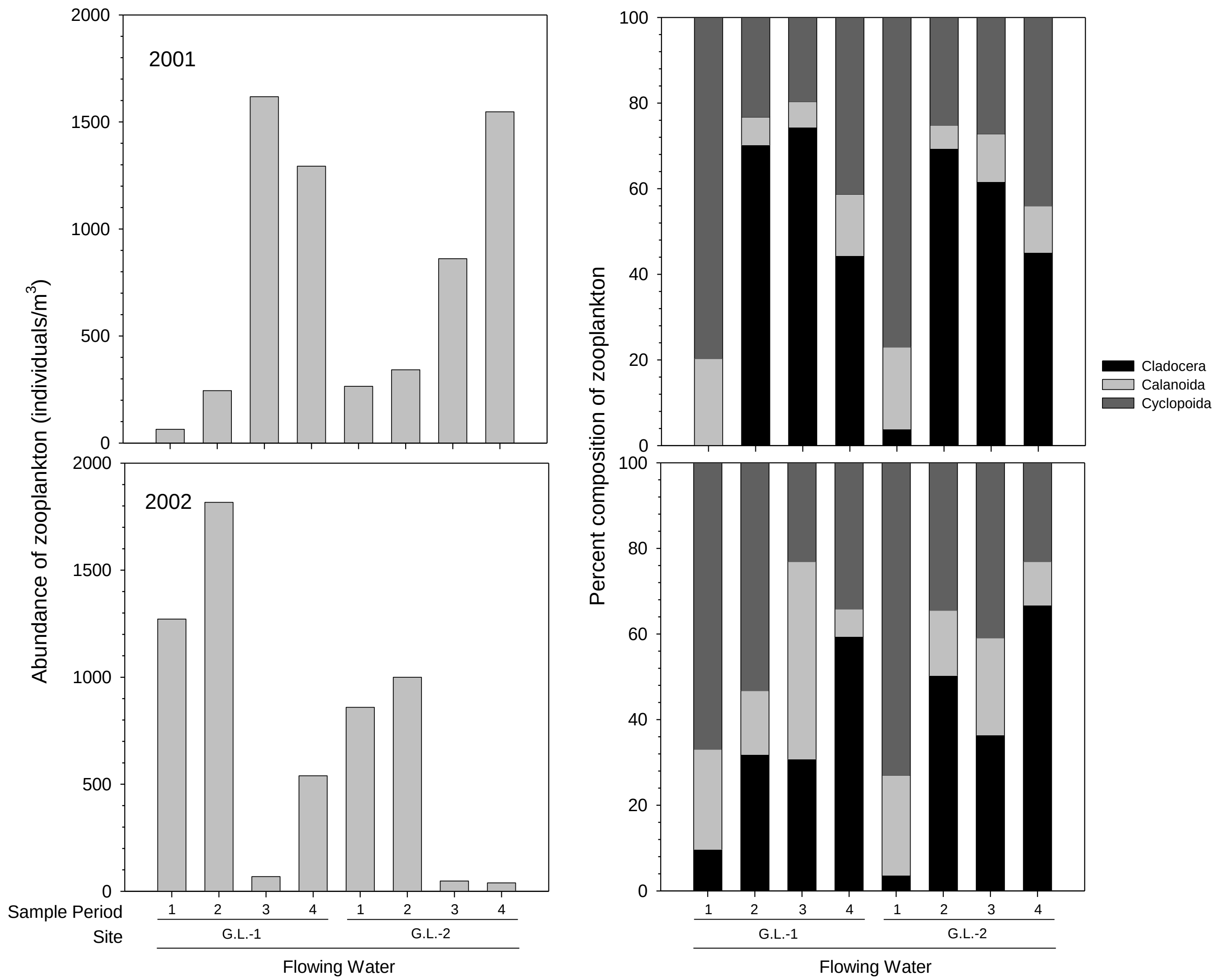


Figure 4-10: Total abundance and percent composition of zooplankton collected in vertical net tows from Gull Lake, 2001–2002

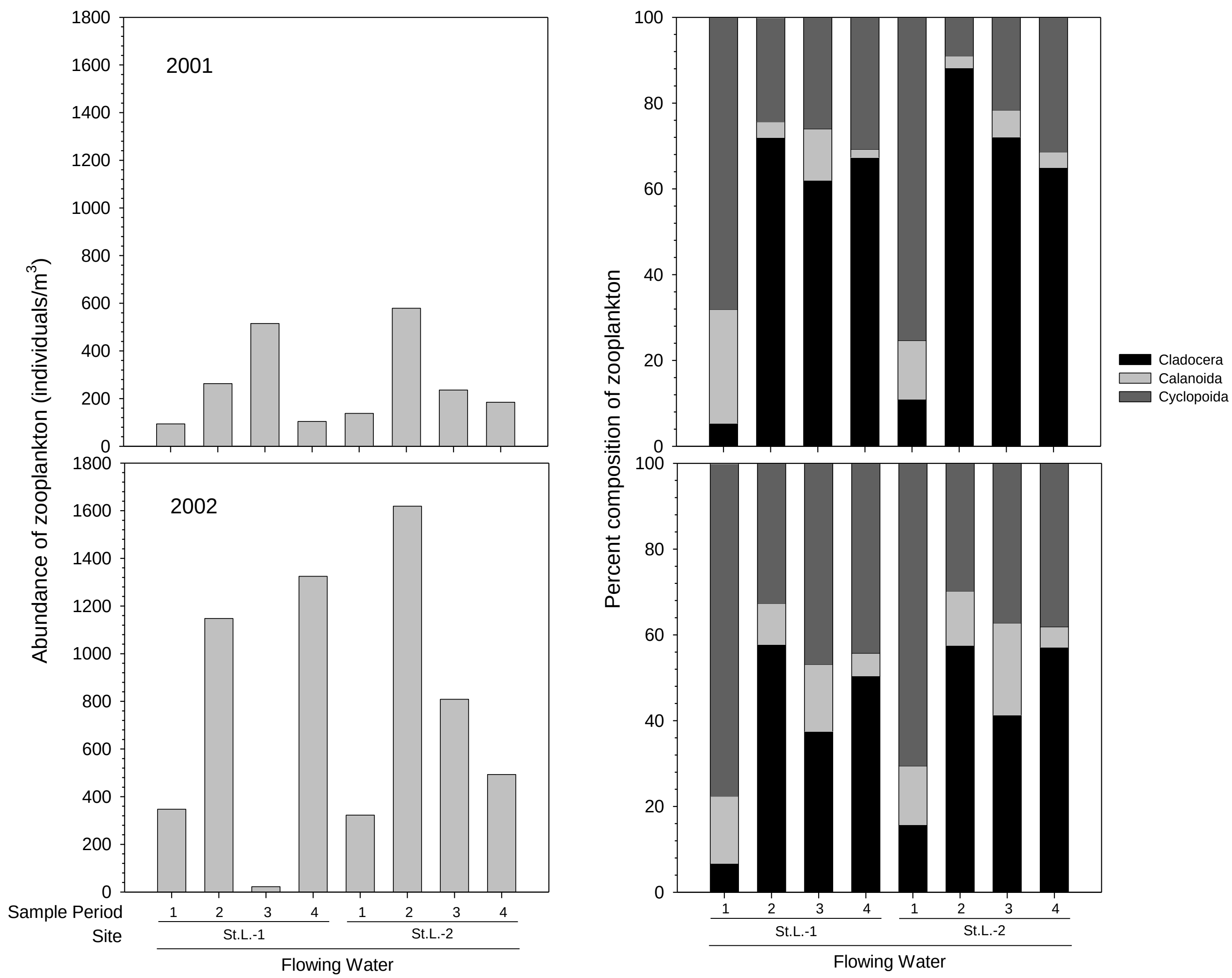


Figure 4-11: Total abundance and percent composition of zooplankton collected in vertical net tows from Stephens Lake, 2001–2002

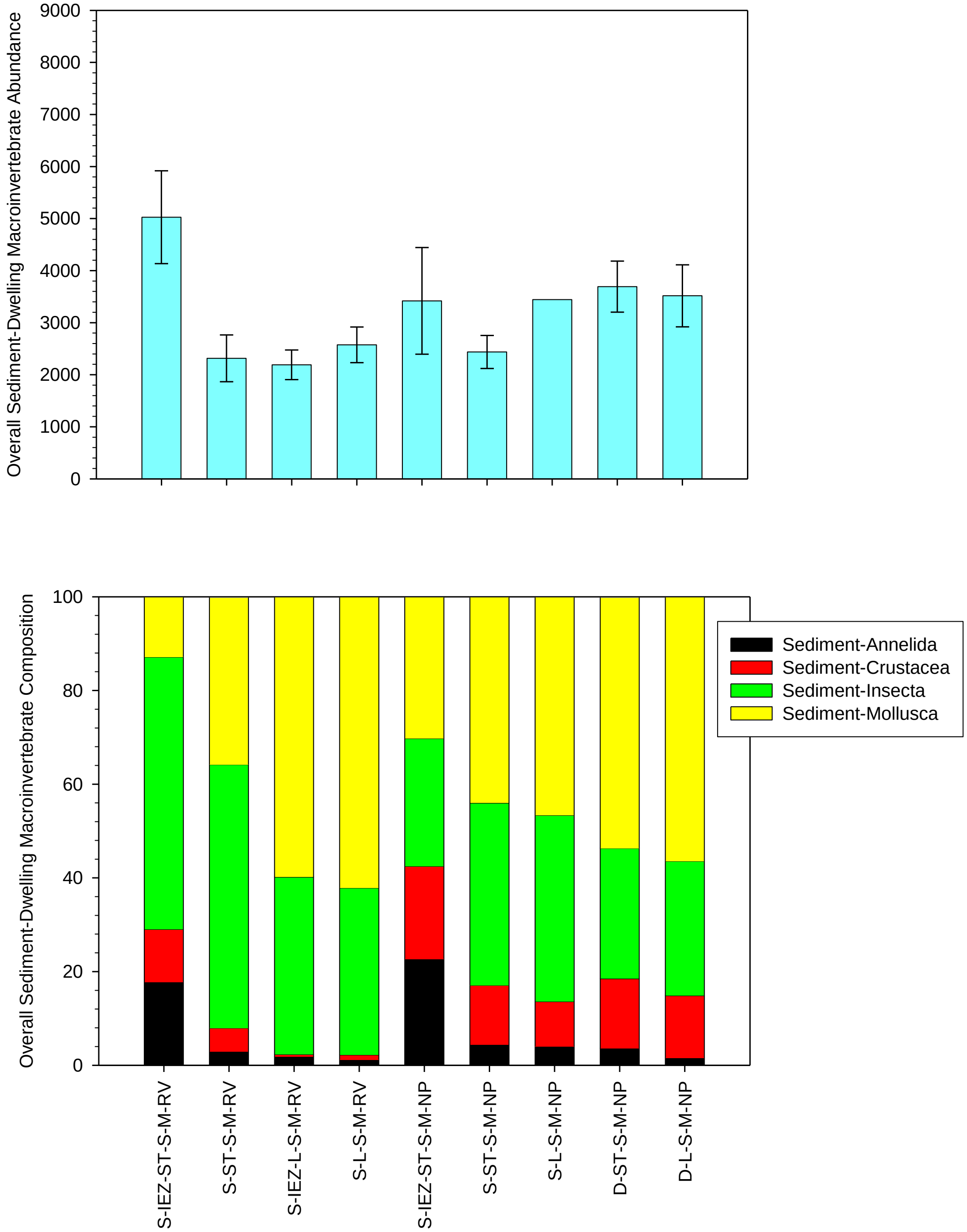


Figure 4-12: Overall abundance (individuals/m^2±standard error) and community composition (%) of benthic macroinvertebrates in the Split Lake area (Split and Clark lakes, and the York Landing arm of Split Lake) by aquatic habitat type, 1997–2004

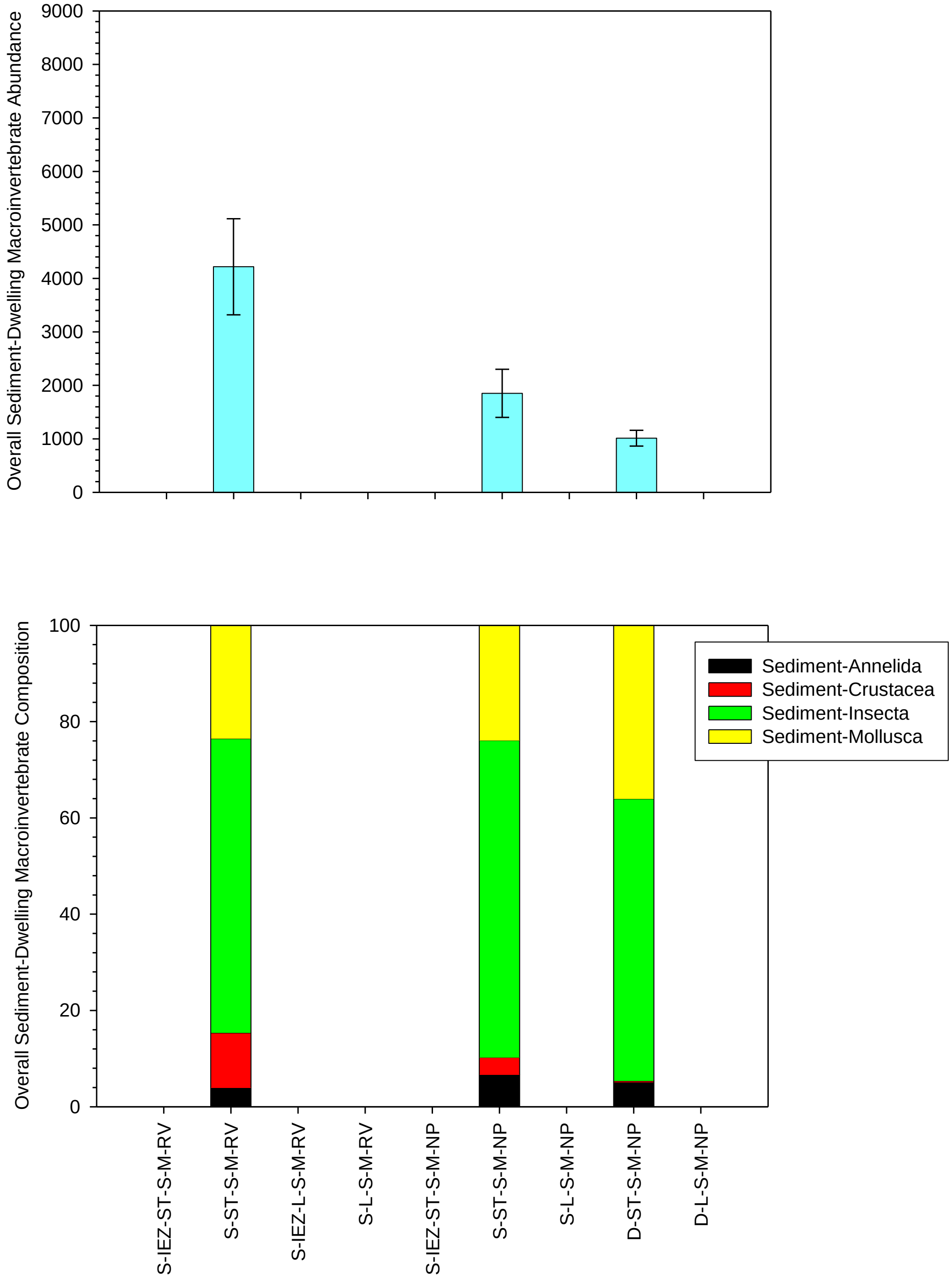


Figure 4-13: Overall abundance (individuals/m^2±standard error) and community composition (%) of benthic macroinvertebrates in Assean Lake by aquatic habitat type, 2001–2004

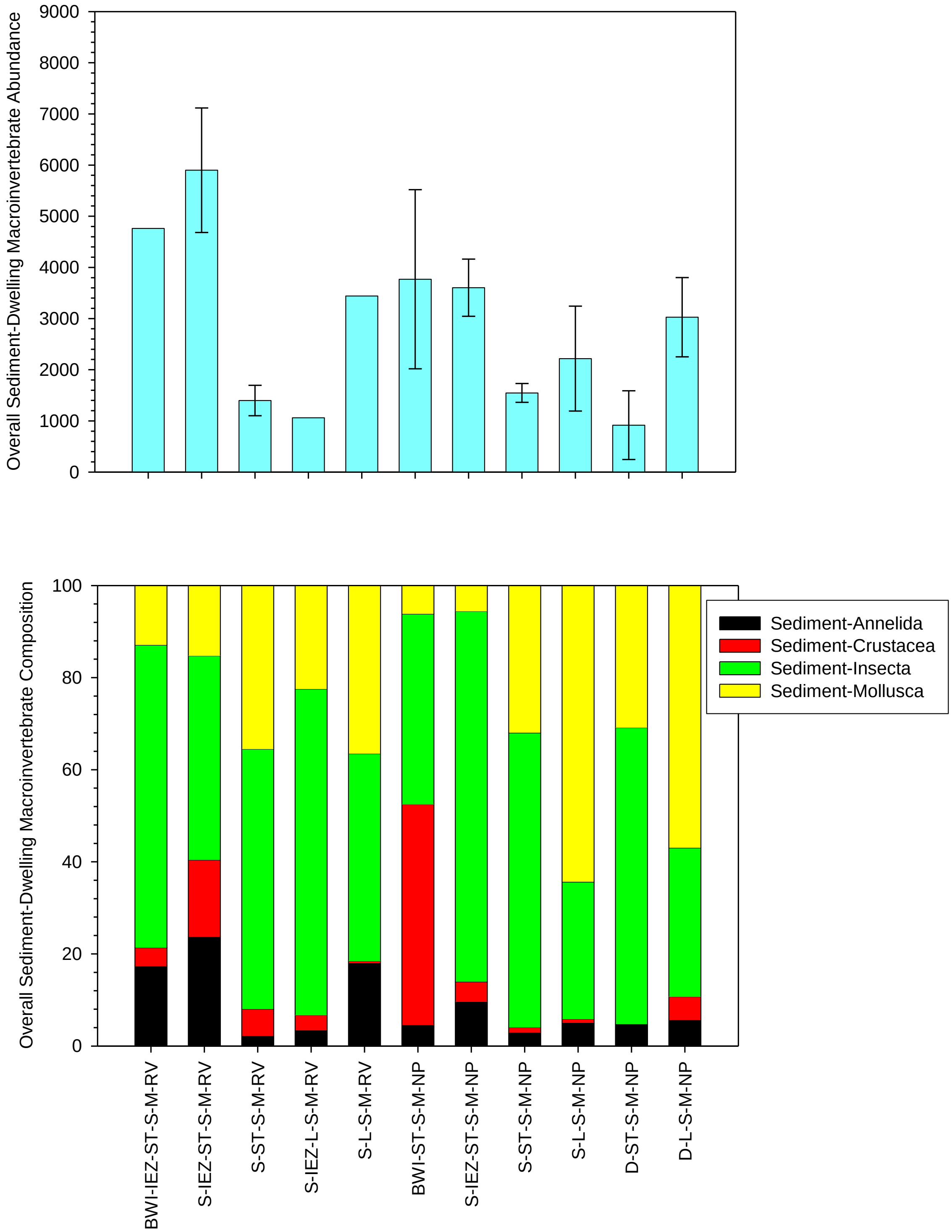


Figure 4-14: Overall abundance (individuals/m^2±standard error) and community composition (%) of benthic macroinvertebrates in the Keeyask area by aquatic habitat type, 1999–2004

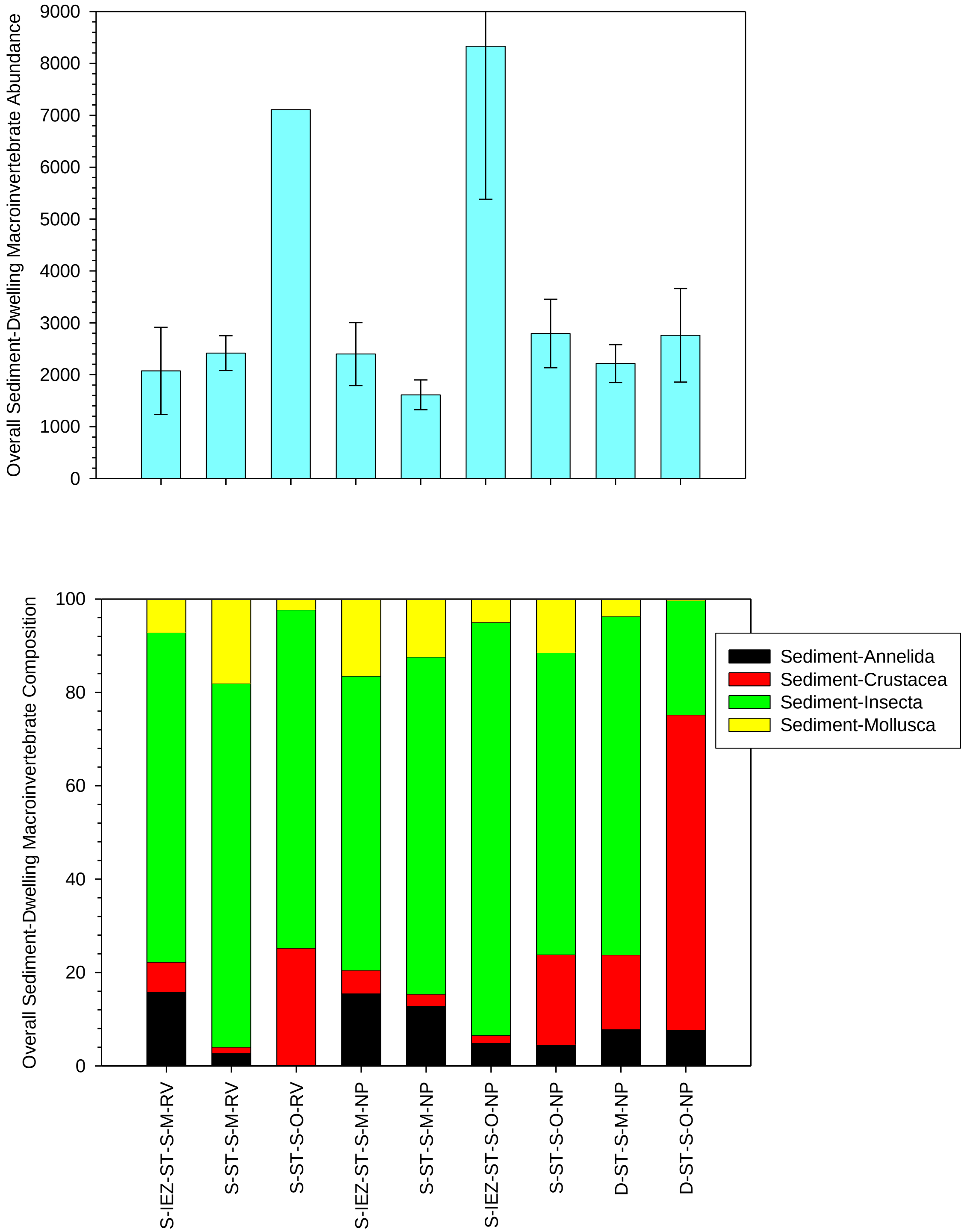


Figure 4-15: Overall abundance (individuals/m^2±standard error) and community composition (%) of benthic macroinvertebrates in the Stephens Lake area by aquatic habitat type, 2001–2006

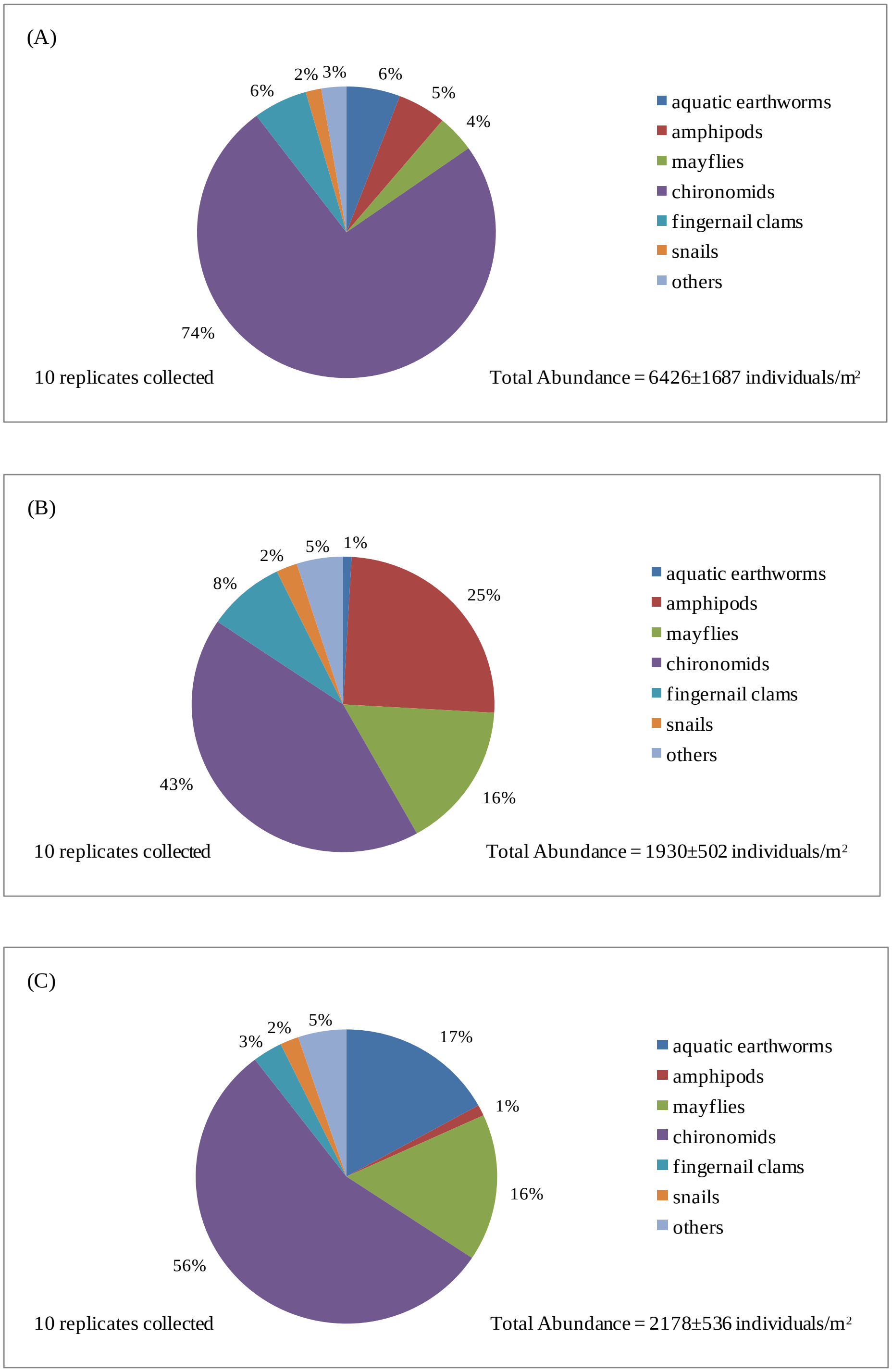


Figure 4-16: Composition of the benthic macroinvertebrate community in site-specific habitat types in Stephens Lake, 2006: (A) shallow, standing water, organic substrate that experiences DO depletion in winter; (B) shallow, standing water, organic substrate with adequate DO in winter; (C) shallow, standing water, silt/clay substrate with adequate DO in winter