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Status of eelgrass beds on the east coast of James Bay

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Preface

This report is intended to provide an objective summary of information available in published papers and unpublished reports, considering factors related to declines observed in the extent of beds of eelgrass, *Zostera marina*, along the east coast of James Bay, Québec. It was written to support Environment Canada's commitment to better understand the causes of changes in the coastal ecosystems of James Bay, and their implications for birds and other wildlife and for the residents of coastal communities.

In addition to providing a summary of available information, an objective of this review was to locate and provide access to as much published and unpublished information on this subject as possible. This report is based on a range of sources assembled by the author, in both French and English, many of which have not been widely accessible.

It is hoped that this review will further Environment Canada's objectives of providing for a thorough, science-based review of changes in the eelgrass beds along the east coast of James Bay, to enable a consensus on next steps to be taken to conserve this important resource.

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Decline of Eelgrass Beds on the East Coast of James Bay

Background

In the 1970s, the 250 km² of eelgrass beds in James Bay were healthy (FOPO 2008) and were considered the most productive habitat on the Quebec coast of James Bay (Curtis and Allen 1976). Hydroelectric development at the La Grande complex began in 1973. Because the north coast of James Bay was conducive to the growth of vast subtidal eelgrass beds, particularly within the freshwater plume from the La Grande River, a series of studies and monitoring activities were implemented to assess the impact of the project on the species. Studies on the use of James Bay coastal habitats by waterfowl were carried out in parallel with these studies on eelgrass, in collaboration with the Canadian Wildlife Service. In 1998–1999, a major decline in eelgrass beds was observed along James Bay, from Rupert Bay to Pointe Louis-XIV, and even in the region of Manitounuk Sound along the eastern coast of Hudson Bay (Lemieux et al. 1999). The cause of the decline was not determined. Since the development of the La Grande hydroelectric complex, and particularly since 1998, the Cree Nation has observed that eelgrass beds are less extensive and that the plants are less healthy (Interview, Cree community of Chisasibi 2013; FOPO 2008; Short 2008). The last report by Hydro-Québec on eelgrass monitoring on the east coast of James Bay indicates that not all beds had recovered in 2011 (Consortium Waska-GENIVAR 2011). The Cree community of Chisasibi confirmed that the eelgrass beds had still not recovered in 2013 (Interview, Cree community of Chisasibi 2013). The eelgrass beds provide important migration stopover and feeding areas for migratory geese, particularly Brant, Canada Geese and Snow Geese. These species are clearly less abundant than previously along the east coast of James Bay, an observation that the Cree attribute to the eelgrass losses observed in recent years.

General objective

The general objective of this report is to collect and review available scientific information as well as Aboriginal knowledge and perspectives on the effect of environmental changes observed on the east coast of James Bay on the distribution and abundance of eelgrass beds and waterfowl, particularly Brant.

The methodology employed for conducting this review was comprised of three activities: (1) review of existing scientific and grey literature on eelgrass; (2) review of existing documentation on Aboriginal knowledge and Cree perspectives on eelgrass; and (3) acquisition of Aboriginal knowledge and Cree community perspective through consultation activities by Environment Canada in Chisasibi in 2013.

1. Ecology of eelgrass

1.1 General description and distribution in eastern Canada

Eelgrass (*Zostera marina* Linnaeus) is an aquatic herbaceous plant that occurs along all marine coasts in the Northern Hemisphere. It is the dominant seagrass species found in coastal and estuarine areas of the western North Atlantic, from the Atlantic coast of Labrador, at 60° N, to North Carolina, at 35° N (den Hartog 1970). It is the only seagrass species in eastern Canada and it occurs anywhere conditions are favourable. However, eelgrass will generally be absent along rock faces, high-energy coastlines or areas of high turbidity (DFO 2009).

Seagrasses are underwater plants with an extensive network of underground roots and rhizomes. Eelgrass is a highly productive species and forms extensive beds in intertidal and subtidal zones (DFO 2009). It is capable of reproducing asexually through elongation of its rhizomes or sexually through seed formation (DFO 2009). The leaves of reproductive plants grow in the middle and upper part of the plant, which contain rows of yellow seeds that facilitate the visual identification of reproductive shoots. Vegetative shoots support several elongated, ribbon-shaped leaves arranged one on top of the other (Lalumière and Lemieux 2002). Flowering shoots and portions of rhizomes can be transported over large distances by currents (DFO 2009; Nellis et al. 2012). The proportion of reproductive shoots relative to vegetative shoots (asexual reproduction) is said to be an indication of the type of reproduction favoured by the eelgrass plant. It appears to rely on sexual reproduction when subjected to disturbance, such as eutrophication and salinity changes (Ewanchuk 1995, from Nellis et al. 2012).

In eastern Canada, eelgrass occurs primarily in the upper and lower estuary of the St. Lawrence River, Gulf of St. Lawrence, Chaleur Bay, Magdalen Islands and James Bay (Nellis et al. 2012). The largest eelgrass beds in the St. Lawrence are in the regions of Manicouagan and L'Isle-Verte in the lower estuary, on Crescent Island and in Aylmer Sound bay on the north shore of the Gulf, and in Baie de Cascapédia in Chaleur Bay (Martel et al. 2009).

1.2 Ecological importance and functions of eelgrass in eastern Canada

Protected status in Canada

Since 2009, Fisheries and Oceans Canada (DFO) has considered eelgrass in eastern Canada to be an *ecologically significant species*. This designation is justified by the fact that eelgrass is sufficiently abundant and widely distributed in eastern Canada to have a measurable influence on the overall ecology of estuaries, bays and adjacent ecosystems. It can be extremely difficult to restore large disappearing eelgrass areas, particularly if the losses are due to anthropogenic stresses, even if recovery efforts are taken (transplants, seeding). If the species were to be perturbed severely, the ecological consequences would be substantially greater than an equal perturbation of most other species associated with the community. DFO is of the view that eelgrass can have a controlling influence over key aspects of the nearshore marine ecosystem structure and function. Eelgrass is the only organism capable of creating habitat on the sand/mud flats of intertidal and subtidal areas within the salinity ranges it occupies (DFO 2009). In addition, eelgrass beds act as important fish habitat. To that end, eelgrass is protected from

harmful alteration, disruption and destruction (HADD) unless authorized under section (s.) 35 or s. 44 of the *Fisheries Act* (DFO 2012).

Ecological functions

Eelgrass beds attenuate wave energy and reduce current velocity. As currents are slowed, sediment particles fall out of suspension, resulting in sediment filtration, trapping and stabilization, and planktonic larvae are able to settle in the habitat created by the eelgrass. On a larger scale, the buffer effect of eelgrass beds may help protect shorelines from erosion (DFO 2009).

Eelgrass occurs at all stages of ecologic succession, i.e., pioneer, transition and climax (Lalumière 1987a). Due to the rapid renewal of its leaves and the presence of epiphytic algae at the surface of beds, eelgrass has an extremely high production rate. The habitats created by eelgrass are among the most productive ecosystems on the planet. Eelgrass tissue is most commonly consumed as detritus at the base of coastal food webs. Eelgrass fragments and particles that are dislodged from the plants are used by colonies of bacteria, fungi and protozoa, or are transported considerable distances by waves and currents, providing a critical source of organic matter to more nutrient-poor environments. Eelgrass occurs predominantly in monocultures throughout most of its range, but can overlap with widgeon grass (*Ruppia maritima*) in the upper portions of low-salinity estuaries in Atlantic Canada. There are more than 20 obligate species of marine algae adapted to eelgrass in Atlantic Canada. A number of other marine algae species are non-obligate, but are most abundant when found in association with eelgrass, which acts as a support. The most common macroscopic algae found on eelgrass leaves in Atlantic Canada are filamentous algae (e.g., *Ulothrix* spp., *Pilayella littoralis*, *Ecotocarpus* spp. and *Polysiphonia* spp.), which grow in dense or highly branched clumps providing refuge and food to small mobile herbivores, such as amphipods. The organisms that live on eelgrass blades and among the root-rhizome system are important in the diets of other organisms at higher trophic levels (DFO 2009).

The areal coverage and vertical distribution of eelgrass beds add spatial complexity to the landscape and provide important habitat for fish and invertebrate species. Eelgrass beds provide refuge and food for a number of fish species and serve as important spawning and nursery habitat for others (DFO 2009, 2012).

Eelgrass also provides feeding and resting habitat for several aquatic migratory bird species. Where it occurs, it makes up almost the entire diet of Brant outside its breeding range. Historical data suggest that if eelgrass declines were to become widespread, there would be major impacts on waterfowl feeding behaviour, migration patterns and over-winter survival (DFO 2009).

1.3 Environmental factors regulating eelgrass growth

The ecological niche occupied by eelgrass is defined by its tolerance to various chemical, biological and physical factors.

The upper limit of eelgrass on shores is determined by wave exposure, the extent of ice scour and desiccation. The lower limit is set primarily by the amount of light that reaches the leaves, but also by sediment movement and disturbance of the substrate by living organisms (DFO 2009). Shallow beds are generally exposed to more natural physical disturbance (desiccation, wave action, ice cover, sediment deposition) than deep beds (Grant and Provencher 2007; DFO 2009). The highest leaf biomasses are typically measured at depths of between 2 and 4 m, whereas the lowest expected values occur in shallow zones (0 to 2 m) (Krause-Jensen et al. 2000 and 2003, from Grant and Provencher 2007).

Hydrodynamics

Eelgrass distribution and abundance are influenced by current and tidal conditions. Specifically, shallow eelgrass beds exposed to wave action will exhibit variable shape and position over time. When exposure is high, the growth of beds is limited to deeper waters (DFO 2009). Low-density eelgrass beds and beds that are fragmented due to stress may be more vulnerable to flow regime variations, which larger contiguous dense beds can resist (DFO 2012).

The combined action of waves and currents results in two types of eelgrass beds: contiguous beds in areas of low hydrodynamic activity and isolated patches in areas of high hydrodynamic activity. Eelgrass is unlikely to occur as contiguous beds under high, sustained current conditions. Extremely strong currents expose the rhizomes and dislodge the plants. Eelgrass grows best at moderate water currents. To maintain an adequate rate of photosynthesis, eelgrass prefers currents of > 16 cm/s and < 120 to 180 cm/s. At current velocities between 120 and 180 cm/s, eelgrass beds established in shallow areas often form small raised elliptical patches, parallel to the current (Lalumière 1987a; DFO 2009).

The combined action of waves and current has a direct influence on sediment movement into and out of the beds. The presence of a current stimulates the transport of dissolved nutrients to the leaves. In beds in areas of high hydrodynamic activity, eelgrass shoot density and below-ground biomass are higher, which allows the plants to more effectively extract nutrients present in small concentrations. Sediments in beds exposed to low current velocities are therefore richer in organic matter and nutrients.

Ice scour

The erosive action of ice can have a significant impact on eelgrass distribution and even growth form (annual versus perennial). Given that the effect of ice is extremely variable from year to year and from site to site, this phenomenon is localized and variable, but remains a factor that influences the spatial and temporal distribution of eelgrass. At shallow water depths, ice movements scrape the seabed and can dislodge plants (Lalumière 1987a).

Salinity

Eelgrass is a true halophytic plant species, i.e., it is intolerant to salinities of < 5 parts per thousand (ppt) (Grant and Provencher 2007). The optimal salinity for growth of eelgrass is between 10 and 30 ppt, although it tolerates salinity levels of 5 to 35 ppt (Short 2008) and can even tolerate freshwater for short periods (DFO 2009). Low salinity levels can make the species susceptible to disease. According to DFO, eelgrass has an optimal salinity range of 20 to 26 ppt for photosynthesis (DFO 2009).

Laboratory research (a mesocosm experiment) was conducted to examine the influence of various salinity treatments on eelgrass growth over a two-month period. The plants were subjected to identical experimental conditions, except for salinity, which varied among the plants (0, 5, 10 and 20 ppt). The experiment showed that a reduction in salinity has an impact on eelgrass growth and morphology. The plants subjected to the 0 and 5 ppt salinity treatments showed reduced biomass, shoot density and leaf growth, and the plants subjected to the 0 ppt salinity treatment were all dead before the end of the experiment (Short 2008).

A literature review conducted by researcher Frederick T. Short in 2008 indicates that there are no studies of the long-term effects of reduced salinity on eelgrass growth and survival. However, given the documented influence of salinity on seagrass density (Thom et al. 2003), leaf size (Baden and Bostrom 2001), leaf production (Burdick et al. 1993; Pinnerup 1980) and photosynthetic activity (Biebl and McRoy 1971; Thorhaug et al. 2006), Short believes that exposure to reduced salinity for extended periods during the growing season will likely result in reduced eelgrass growth and abundance (Short 2008).

Temperature

The optimal temperature for eelgrass growth is between 10 and 20°C (den Hartog 1970; DFO 2009). However, it grows well over a wide range of temperatures (10 to 35°C) and survives in freezing temperatures. Photosynthesis decreases rapidly between 30 and 40°C, however. Elevated temperatures can weaken eelgrass and make it more susceptible to disease (Lalumière 1987a; DFO 2009). Laboratory studies have shown that temperatures above 28°C can affect its growth, resulting in reduced shoot density and size as well as reduced production of rhizomes and leaf tissue (Touchette et al. 2003, from Grant and Provencher 2007). Sexual production was documented for temperatures as low as 5°C.

Sampling conducted between 2005 and 2010 in six eelgrass beds in the St. Lawrence marine area reveals that temperature accounts for only a small proportion of the observed variations in eelgrass height (Nellis et al. 2012).

Turbidity

For optimal growth, eelgrass needs clear, i.e., oligotrophic, waters. Increased turbidity resulting from nutrient loading or suspended sediments reduces water clarity and promotes phytoplankton growth, which further reduces water clarity and ultimately the amount of light reaching the plants (DFO 2012).

Water turbidity is one of the key factors that can cause eelgrass mortality. Chronic high spring season turbidity was cited as the primary factor preventing the re-establishment of healthy eelgrass beds in a portion of Chesapeake Bay (Moore et al. 1996). A study by Zimmerman et al. (1991) suggests that extended periods of high turbidity in spring could have been responsible for the limited depth distribution of eelgrass in San Francisco Bay (from Moore et al. 1996).

Species of the genus *Zostera* have a high minimum light requirement. In contrast to other seagrasses, they do not tolerate low light levels for extended periods of time. Detrimental effects on eelgrass can be expected at < 60% surface light (DFO 2012), and destruction of eelgrass as fish habitat is expected at < 35% surface light (DFO 2012). According to Short et al. (1995, from DFO 2012), the minimum light limit for eelgrass survival is 10 to 20% surface light. However,

plants can survive for up to one month at low light levels, but are unlikely to survive under conditions of continuous complete shading of more than three days (DFO 2012). Under overwintering conditions, the species has lower light requirements. The maximum depth of water colonized by eelgrass is determined by the amount of light reaching the bottom. Sampling carried out between 2005 and 2010 in marine areas of the St. Lawrence indicate that light accounts for only a small proportion of the variation observed in eelgrass leaf length (Nellis et al. 2012).

The water quality components that have the greatest effect on eelgrass survival are nitrogen and phosphorus concentrations, the level of chlorophyll *a*, suspended sediments and water clarity (Neckles et al. 2009).

Suspended sediments

Increased turbidity resulting from sediment input or resuspension of sediments reduces water clarity (DFO 2012). Plants affected by turbidity stress will reduce their carbon reserves, which will lead to reduced biomass, growth, shoot density and area coverage, which can ultimately lead to destruction.

According to DFO (2012), total suspended solids levels of < 20 mg/L would cause no damage to eelgrass. However, this value varies with water depth (DFO 2012).

Nutrients

Short and Wyllie-Echeverria (1996) conclude that anthropogenic nutrient inputs to coastal areas are the primary cause of the present worldwide seagrass decline. For example, eelgrass decline in Ninigret Pond, Rhode Island, has been attributed to increased nutrient loading from residential development in the region (Short et al. 1996).

Eelgrass is intolerant of anoxic and eutrophic conditions (DFO 2009). Eelgrass rhizomes are normally effective in absorbing nutrients. In eutrophic waters, the rhizomes likely cannot meet the demand, resulting in increased concentrations of nutrients in sediments, which can disturb eelgrass growth.

Nitrate levels of $\leq 3 \mu\text{M NO}_3\text{-N}$ per day in the water column are generally toxic to eelgrass, and ammonium levels of $\leq 100 \mu\text{M NH}_4^+$ cause disruption of eelgrass (DFO 2012). Total nitrogen thresholds for survival of eelgrass have been estimated for certain estuaries. Concentration thresholds in estuaries in Massachusetts are generally about 0.32 to 0.38 mg/L (Neckles et al. 2009). The nitrogen threshold estimated for Great Bay in New Hampshire is similar, i.e., 0.30 mg/L (Trowbridge et al. 2009, from DFO 2012). Nutrient loading rates of $\leq 12 \text{ kg N/ha/year}$ are reported to have had no major effect on eelgrass beds in Waquoit Bay estuaries, a rate of 30 kg N/ha/year led to losses of 80 to 96% of eelgrass bed area, and a rate of $\geq 60 \text{ kg N/ha/year}$ caused total disappearance (DFO 2009, 2012). The extent of eelgrass beds is inversely related to nutrient inputs to the watershed (Neckles 2009).

Eutrophication can also result in a reduction in dissolved oxygen concentration in the water column, which can vary depending on the temperature and concentration of hydrogen sulphide in the water. For example, eelgrass can tolerate only 8 hours of anoxia in the water column at 20°C. After 36 hours under such conditions, eelgrass begins to die off (DFO 2012). Low oxygen

conditions may be a key factor in the rapid die-off of eelgrass (Greve et al. 2003, from DFO 2012).

Nature of sediments

Although most unconsolidated substrates are suitable for the establishment of eelgrass, the type of substrate influences the rate of growth and morphology of eelgrass. This species of seagrass can inhabit areas with sediments ranging from clay to sand (Thayer et al. 1984, from Lalumière et al. 1994).

Sedimentation

Eelgrass does not survive a rapid sedimentation event particularly well. At a burial depth as low as 25% of the average above-ground plant height, the probability of mortality can exceed 50%. The probability of mortality increases rapidly when burial is 50% of plant height, this mortality occurring in less than a month. Complete mortality would occur at 75% burial (Mills and Fonseca 2003, from DFO 2012).

Physical damage or removal

Physical damage or removal, caused by dredging, boat mooring scour, docks, piers, propeller scars and fishing practices (dragging, digging, trapping) is a stressor for eelgrass. Rhizome damage is the most substantive concern (DFO 2012).

1.4 Response of eelgrass beds to stress factors

As a stressor damages an eelgrass bed, the bed will go from the disturbance stage (recovery expected within a year) to the deterioration stage (recovery incomplete and taking more than one year), and then to the destruction stage. The effect of each stressor will depend on its intensity, duration, extent and frequency. As stress on an eelgrass bed increases, the bed can respond by becoming less dense, becoming patchy, declining in areal extent, or a combination of the three. A contiguous bed of eelgrass is considered to have an areal extent of hundreds to thousands of square metres, while patchiness is considered on a spatial scale of tens of square metres. Patchiness in eelgrass beds is the result of the division of a contiguous bed into smaller, more numerous units. It has been suggested that a loss of about 50% of the eelgrass cover corresponds with a transition from a contiguous bed to that of distinct patches (Fonseca and Bell 1998). Eelgrass can also respond to stress by reducing the growth rate of leaves or photosynthetic rate. If stress is reduced, eelgrass beds may fully recover to their initial state or some point close to it by reversing the steps along the model pathway (Figure 1; DFO 2012). An overview of possible eelgrass bed responses to different stressors is presented below (Table 1; DFO 2012).

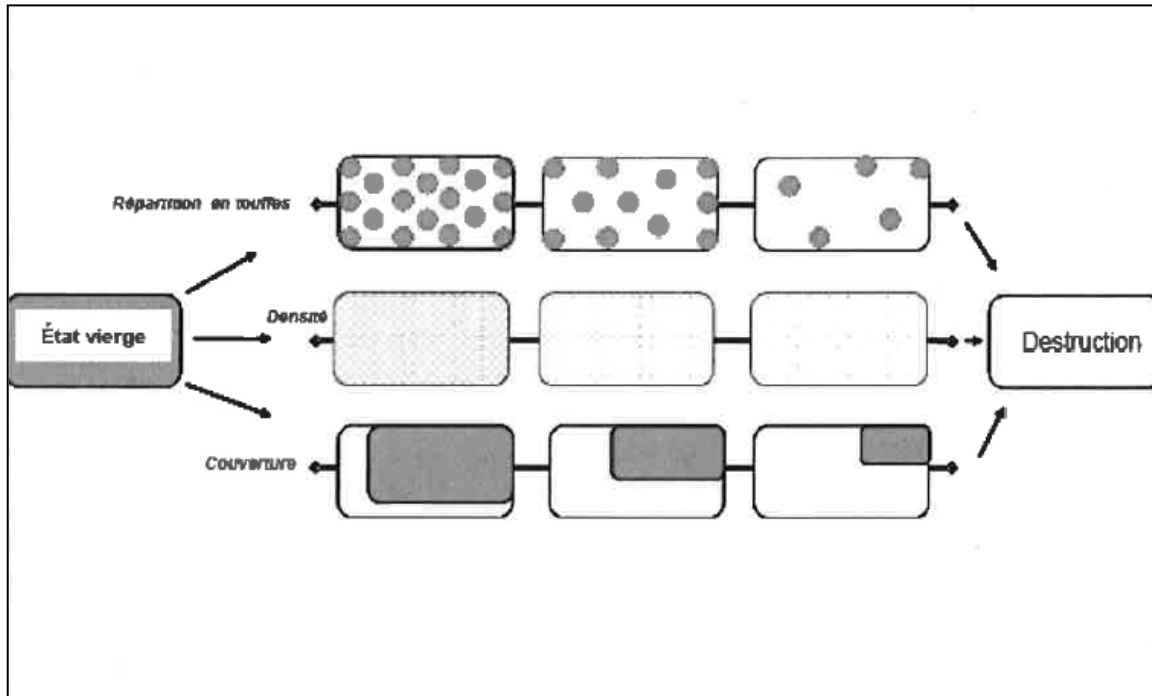


Figure 1: Simplified framework model of changes in form of eelgrass beds in response to stress. As a stressor increases in intensity, the eelgrass bed can be altered from the pristine condition on the far left (a contiguous high-density bed fully occupying the available area) to complete destruction (far right). If stressor intensity is reduced, the paths can be reversed, as indicated by the arrows along each path (from DFO 2012).

Translation key for Figure 1

État vierge = Pristine

Répartition en touffes = Patchiness

Densité = Density

Couverture = Area covered

Destruction = Destroyed

Table 1: Summary of stressor effects on eelgrass (from DFO 2012)

Stressor	<i>Eelgrass bed response</i>		
	<i>Patchiness</i>	<i>Density</i>	<i>Contiguous area</i>
Sedimentation ↑	↑	↓	↓
Turbidity ↑	↑	↓	↓ (deeper portions first)
Nutrients ↑	↑	↓	↓
Water flow* ↑ ↓	↑ ?	↔ ↔	↓ ↓
Physical damage or removal ↑	↑	↓	↓

* above and below optimal flow rate

1.5 Properties of eelgrass beds colonizing the east coast of James Bay: reference state

General description

In the 1980s, studies to characterize the eelgrass beds of the east coast of James Bay were carried out as part of the La Grande hydroelectric project. Apart from the studies by Curtis in 1975–1976, there are no other significant studies that can be used to provide a picture of eelgrass beds in conditions closest to their natural state. These studies were used in this section to describe the properties of the eelgrass beds of James Bay.

In the early 1970s, eelgrass beds were considered the most resource-rich habitat on the Quebec coast of James Bay (Curtis and Allen 1976). In 1988, the eelgrass beds of the east coast of James Bay were considered among the most extensive concentrations of this type of underwater bed in the world (Lalumière 1988a). Eelgrass grows in extensive beds along the east coast of James Bay, primarily in the subtidal zone, from the spring tide low-water mark to a water depth of up to 3 to 5 metres. The coast provides favourable conditions for the establishment of large eelgrass beds due to the combined effect of the protection provided by the many islands, the large number of gently sloping bays, the nature and relative stability of the sediments, the low amplitude tides, the moderate or high salinity of the water, and the suitable water temperature in sheltered bays (Lalumière 1987a; Dignar et al., 1991; FOPO 2008). The largest eelgrass beds on the east coast of James Bay were found in the region of Chisasibi (FOPO 2008).

Eelgrass growth and development

In the characterization study of the eelgrass beds of James Bay for the James Bay Energy Corporation in 1986–1987 (Lalumière 1987a, 1988a), most of the eelgrass showed a perennial growth habit. An annual growth habit was observed in only a few locations, including Dead Duck Bay. The perennial growth habit is characteristic of environments with little or no stress, in contrast to an annual growth habit, which is typical of stressed environments.

On the east coast of James Bay, eelgrass begins spring growth in May under an ice cover. Maximum eelgrass growth is reached in mid-June. It continues to grow rapidly in July and reaches its maximum biomass in September. To survive year-round, eelgrass plants need to grow rapidly and to fully utilize the available summer light in June and July for photosynthesis (Short 2008).

On the east coast of James Bay, eelgrass grows in oligotrophic water, primarily on sediments dominated by fine particles (silt and clay) containing a sandy fraction. In 1987–1988, summer salinity ranged from 10 to 20 ppt, and the annual temperature was between -0.1°C and 20°C. In winter, coastal salinity was more variable due to freshwater plumes from various tributaries (Lalumière 1988a). According to data collected by Roche ltée and Lalumière between 1982 and 1986, water turbidity was relatively low to moderate along the coast, although it could be high locally due to the influence of specific hydrometeorological conditions. Contiguous, higher-quality eelgrass beds were generally located in bays protected from wave action where subtidal slopes were gentle, and they were associated with currents of between 20 and 40 cm/s. As the current velocity approached 40 cm/s, shoot density increased (Lalumière 1987a; Dignard et al. 1991).

Eelgrass growth changes as a function of depth on the subtidal slope. Eelgrass beds are more stable at a depth of 1.5 m than at a depth of 0.5 m, where they are subjected to greater environmental stress caused by agents of erosion. The density of the branches (shoots and leaves) decreases with water depth, i.e., the shoots are denser and shorter in shallower water and become less dense and longer as depth increases (Lalumière and Lemieux 2002). Eelgrass stem density is known to decrease with increased depth due to the presence of higher light near the water surface (Krause-Jensen et al., 2000, from Grant and Provencher 2007). Annual variations in eelgrass biomass do not necessarily show the same trends from one depth to another (Lalumière 1988b). On the east coast of James Bay, eelgrass reproduces primarily vegetatively when it grows at depths of ≥ 1 m (Lalumière 1987a).

The exploratory studies carried out between 1982 and 1985 on the east coast of James Bay indicated that the coast provides good salinity conditions for eelgrass growth and that the eelgrass beds of James Bay are not distributed along a salinity gradient. The growth and distribution of eelgrass is influenced primarily by the hydrodynamic forces generated by tidal currents, wind (waves) and their combined action. Physical factors, such as the scalloped shoreline, the presence of many islands and shoals, the bathymetry of the sites, the slope of the shores, and the exposure to waves and prevailing winds, have an impact on these hydrodynamic forces, depending on the section of shoreline in question (Lalumière 1987a).

The effect of ice and the type of substrate on which eelgrass grows are also significant physical factors that affect the species' growth from site to site. Aerial photographs taken in the 1980s show signs of ice scour and suggest that the process is relatively widespread along the east

coast of James Bay (Lalumière 1986, 1987a). In 1988, it was noted that all of the eelgrass beds studied were mature (Lalumière 1988a).

Aquatic species associated with eelgrass beds

When currents are weak, green algae such as *Enteromorpha* sp., *Cladophora* sp., *Rhizoclonium riparium* and *Chorda filum*, and vascular plants such as *Ruppia maritima* and *Potamogeton pectinatus* are found in association with eelgrass, particularly in shallow waters. In the eelgrass bed characterization studies carried out in 1986–1987 (Lalumière 1987a), it was observed that widgeon grass, and particularly sago pondweed, were more abundant closer to the mouth of the La Grande River. Moreover, widgeon grass was found in association with eelgrass in shallow waters and disappeared quickly as the distance from the shore increased (Lalumière 1987a).

Eelgrass acts as a support for epiphytic plants (diatoms, other microscopic algae, bacteria, etc.) and animals (molluscs, cnidarians, bryozoans, annelids, etc.) at the bottom of somewhat complex food chains. A number of benthic species also grow in sediments colonized by eelgrass (DFO 2009).

Eelgrass beds provide habitat for a number of fish species, chief among them sculpins (*Myoxocephalus quadricornis*, *M. scorpius*), Arctic cod (*Gadus ogac*), cisco (*Coregonus artedii*), brook trout (*Salvelinus fontinalis*) and lake whitefish (*Coregonus clupeaformis*). Lake whitefish is more often associated with areas where eelgrass is absent, whereas Arctic cod prefer areas colonized by eelgrass. There is a greater availability of prey for fish within eelgrass beds than outside of them (Lalumière 1988a).

The extensive eelgrass beds along the east coast of James Bay are heavily used as feeding and resting areas by several migratory birds species, particularly waterfowl during migration (geese and ducks) and shorebirds (sandpipers, plovers, etc). The beds also provide feeding sites for guillemots and terns, which feed on the small fish that thrive within eelgrass beds (Short 2008). Coastal habitats are a particularly important migratory stopover site for two species, Canada Goose (*Branta canadensis*) and Brant (*B. bernicla hrota*), and provide a significant proportion of the food resources required for building the nutrient reserves these birds need in order to continue their migration, and for reproduction. On the east coast of James Bay, Brant feed almost exclusively in eelgrass beds during spring and fall migration (Reed et al. 1996).

2. Environmental changes on the east coast of James Bay

2.1 Development of the La Grande hydroelectric complex

Description of facilities

Construction of the La Grande hydroelectric complex began in 1973. The facilities were built in two phases. Phase I (1973 to 1985) consisted of the construction, under the supervision of the James Bay Energy Corporation (JBEC), of three power plants and associated reservoirs—Robert-Bourassa (formerly La Grande-2), La Grande-3 and La Grande-4—as well as two

additional reservoirs, Caniapiscou and Opinaca, formed by the partial diversion of rivers. During Phase I, the impact of the construction of the La Grande facility on eelgrass was not determined. However, a map of eelgrass distribution on the east coast of James Bay was produced in 1975 by the Canadian Wildlife Service to provide an overall picture of the status of eelgrass beds during this period (Curtis 1975).

Phase II of the La Grande complex was constructed from 1987 to 1996. The equipment plan included the construction of five power plants: La Grande-1, La Grande-2-A, Laforge-1, Laforge-2 and Brisay. Although the work began in 1987, power plant operations came on line gradually between 1991 and 1996. The six generators of the La Grande 2-A station were phased into service from September 1991 to October 1992, the La Grande-1 reservoir was impounded in October 1993, and the 12 generators of the La Grande-1 station were gradually phased into service from February 1994 to September 1995 (Lalumière and Lemieux 2002).

In 2007, Hydro-Québec began construction of the Eastmain-1-A and Sarcelle power plants and the Rupert diversion. The Rupert River was partially diverted to two new power plants and to the existing Robert-Bourassa, La Grande-2-A and La Grande-1 stations. The expected increase in the mean annual discharge of the La Grande River was approximately 450 m³/s, a 14% increase (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006).

Hydrology of the La Grande River and its plume

The filling of the Robert-Bourassa reservoir in November 1978 put an end to the natural hydrologic regime of the La Grande River (Saucet 2002). The main change observed in the river's hydrologic regime is the marked increase in flow, particularly in winter, which has resulted in an increase in the extent of the river's plume under the ice cover (Messier 2002). The plume is that part of the coastal zone where freshwater and brackish water extend over the denser saltwater. Wind and currents contribute to the mixing of the freshwater with the saltwater as the freshwater moves away from its source (Messier 2002). The increase in freshwater discharge at the mouth of the La Grande River could result in significant changes in water salinity, temperature and quality in James Bay and Hudson Bay, into which the freshwater discharge flows (Short 2008).

The waters of Hudson Bay, both surface and deep waters, flow into James Bay from the west and out to the east (Messier 2002). In a study characterizing the La Grande River's freshwater plume under ice cover, Ingram and Larouche (1987) observed that the plume waters were deflected northward as they entered James Bay, following the coastal circulation. Similarly, McDonald et al. (1997) reported that freshwater from the La Grande River that flowed into James Bay circulated primarily towards the north and extended to the southeast of Hudson Bay, following the hydrodynamic circulation of James Bay. As a result, eelgrass growing on the northeast coast of James Bay would be more likely to be affected by changes in the freshwater plume of the La Grande River. The report of the federal review panel for the Eastmain-1A and Rupert diversion project provides a summary of the opinions of experts and stakeholders on the changes observed in the hydrologic regime of the La Grande River and their possible causes. One expert was of the view that there were no data to support the possibility suggested by the NTK experts (Nunavuummi Tasiujarjuamiuguqatigiit Katutjiqatigiingit; Nunavut Hudson Bay Inter-Agency Working Group) that the plume of the La Grande River dispersed from the northeast side of James Bay. According to that expert, the existing oceanographic data showed no evidence of eastward expansion (Ingram 2006).

Discharge at the mouth of the La Grande River

Prior to the development of the La Grande hydroelectric complex, mean annual discharge from the La Grande River contributed roughly one third of the freshwater input of the east coast of James Bay. Since the start-up of the hydroelectric facilities, freshwater discharge from the La Grande River has provided half the total supply of all tributaries on the east coast of James Bay, and the increase is more significant in winter (Messier 2002).

Prior to the construction of the La Grande hydroelectric complex, the hydrologic regime of the La Grande River consisted of a severe low-flow period in winter and a pronounced spring runoff in June, followed by a relatively smaller summer low flow and fall runoff (Saucet 2002). Flows from the river peaked in mid-June, with an annual mean flow of 1689 m³/s (Hamilton and Whittaker 2005, from Short 2008), while the minimum flow was in late March (Déry et al. 2005, from Short 2008). Following Phase II of the hydroelectric project, the mean annual discharge doubled, reaching 3400 m³/s.

The report by Messier (2002), which summarizes the studies conducted on the plume of the La Grande River as part of environmental monitoring of the La Grande-2-A and La Grande-1 projects, contains a table of monthly discharge recorded at the La Grande-2 station (now the Robert-Bourassa station) between 1979 and 2000 (see Table 2). Figure 2 presents the discharge recorded at the La Grande River according to Whittaker and Hamilton (2005), in a study conducted by NTK.

Table 2: Monthly discharge (m³/s) at the La Grande-2 (Robert-Bourassa) site for the years of plume monitoring and operation of Phase I and Phase II (from Messier 2002)

	TRANSITION		PHASE I: REFERENCE STATE						
	79–80	83–84	85–86	86–87	87–88	88–89	89–90	90–91	Avg.
October	345	3119	2740	3170	4169	2713	2955	2176	2987
November	887	2751	3292	4080	3877	2713	3525	3216	3451
December	1997	3112	3522	4010	4219	3532	3982	3374	3773
January	1388	3151	3851	4113	4147	3633	2970	3811	3754
February	1477	2340	4110	3819	4157	3955	3413	3653	3851
March	1613	2628	3884	3741	4066	3573	3213	3453	3655
April	1633	1536	2810	3266	3594	3239	2601	3051	3094
May	1918	1778	2970	3684	3036	1689	2187	2178	2624
June	1752	1874	2586	3679	3082	2040	2542	2437	2728
July	2687	1674	2377	2986	2842	2504	2123	2777	2602
August	2354	2896	2415	3105	3010	2642	2332	2662	2694
September	1913	3143	2721	4022	2443	2581	2477	2641	2814
Mean	1664	2500	3107	3640	3554	2901	2860	2952	3169
Minimum	345	1536	2377	2986	2443	1689	2123	2176	
Maximum	2687	3151	4110	4113	4219	3955	3982	3811	

	TRANSITION	PHASE II: OPERATION						
	92–93*	94–95	95–96	96–97	97–98	98–99	99–00	Avg.
October	2527	1752	2837	2838	2536	1717	3244	2487
November	3338	2503	3952	3544	2733	2382	3689	3134
December	3505	3671	4165	2537	3760	2612	4069	3469
January	3844	4184	4568	3997	2876	3621	4550	3966
February	4718	4487	4629	3731	2499	2478	4557	3730
March	3497	2941	4200	3579	2860	2448	3784	3302
April	2613	3290	3242	2483	2288	2192	3842	2890
May	2199	2396	1626	2173	2504	2902	2662	2377
June	2262	2343	2614	2696	2707	3416	3291	2845
July	2346	2854	2612	2253	2659	4145	3259	2964
August	3012	2720	2864	2190	2443	3960	3398	2929
September	3062	2315	2616	2372	1832	3617	3203	2659
Mean	3077	2955	3327	2866	2641	2958	3629	3063
Minimum	2199	1752	1626	2173	1832	1717	2662	
Maximum	4718	4487	4629	3997	3760	4145	4557	

*PHASE II, transitional period

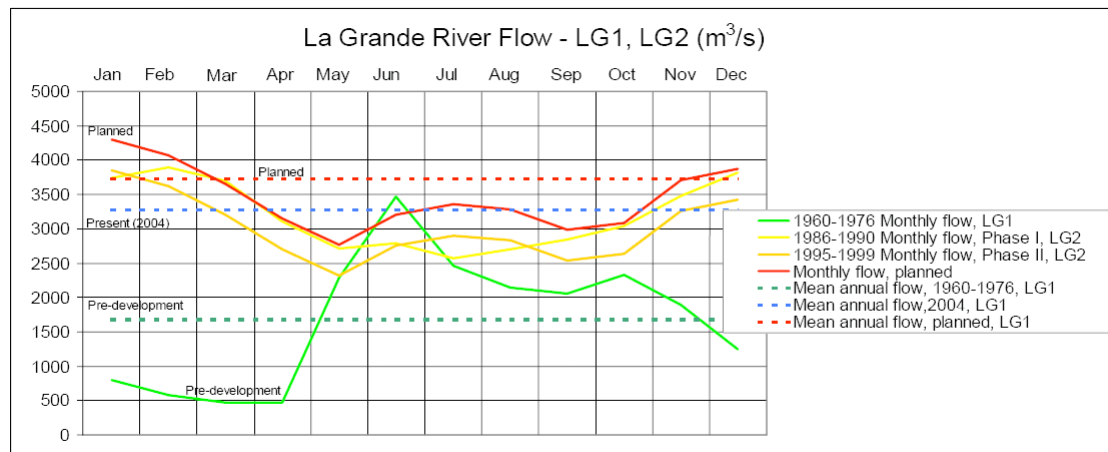


Figure 2: Historic, present and planned flows of the La Grande River according to Hamilton and Whittaker (2005) (from Short 2008)

At the meeting of the Standing Committee on Fisheries and Oceans of March 4, 2008, Alan Penn, science advisor to the Grand Council of the Crees, stated that over the last 10 years or so, the distribution of flows was different from what it had been at the time of the initial planning of the La Grande hydroelectric project. At the outset, the complex was designed to meet the electricity needs of the province only. Today, Hydro-Québec supplies electricity to northeastern North America. The overall effect of this has been a series of peaks that are more difficult to predict because they are responses to market demand and to the United States summertime energy consumption pattern (FOPO 2008).

Winter discharge

The plume area under the ice cover is larger than in open-water conditions. This can be explained by the absence of wind action and by the reduced effect of the tide on freshwater-saltwater mixing in winter (Messier 2002). Following the start-up of the power plants, the period of peak discharge from the La Grande River shifted from summer to winter (Messier 2002). Since the construction of Phase I of the La Grande complex, the spring runoff is stored in the reservoirs and discharge is maintained at a high level during the cold season (Saucet 2002). In the winter of 1986–1987, winter discharge increased from approximately 500 m^3/s to over 4000 m^3/s (Messier et al. 1988). Hydro-Québec carried out most of its monitoring of the freshwater plume of the La Grande River in winter (Messier 2002).

First-hand accounts collected from members of the Cree community living along the east coast of James Bay indicate that the freshwater plume under the ice now extends further into the bay than it used to (McDonald et al. 1997).

Summer discharge

The summer plume of the La Grande River was not studied in detail until 1983. The surface salinity distribution pattern was believed to be highly variable (Roche 1983, from Messier 2002). From the early 1980s to the summer of 1998, freshwater discharge to James Bay gradually increased yet remained relatively stable. Between October 1998 and September 1999, freshwater inputs decreased somewhat between December and March. During the same period, discharges increased by $1000 \text{ m}^3/\text{s}$ over the months of June, July and August, which was 1.7 times the average discharge measured in the summer of 1980 (Messier 2002: see Table 2). There was therefore a considerable increase in freshwater discharge to James Bay during the period of maximum eelgrass growth (Short 2008). Freshwater discharge to James Bay remained high over the summer of 2000 (Messier 2002, from Short 2008). It has not been possible to obtain information on monthly discharge rates from the La Grande River since 2000 (Short 2008).

Salinity of the plume of the La Grande River

Prior to the construction of the La Grande complex, salinity in the waters around the eelgrass beds was approximately 20 ppt (Lalumière et al. 1994). Following the start-up of the La Grande complex, the freshwater plume of the La Grande River resulted in a decline in the coastal salinity of James Bay.

In February 1984, the freshwater plume of the La Grande River under the ice was described. The areal extent of the plume was three times larger than for natural runoff levels. For February 1984 discharge levels, plume dilution of James Bay surface waters extended 100 km north of the river (see Figure 3). Salinity was reduced over a width of roughly 20 km along the coast. Ingram and Larouche (1987) estimated that the increase in winter discharge will lead to dilution of the nearshore surface waters as far as southeast Hudson Bay. The authors provided a summary of the areal extent of the plume as a function of winter discharge between 1976 and 1984, for different isohaline values (see Table 3).

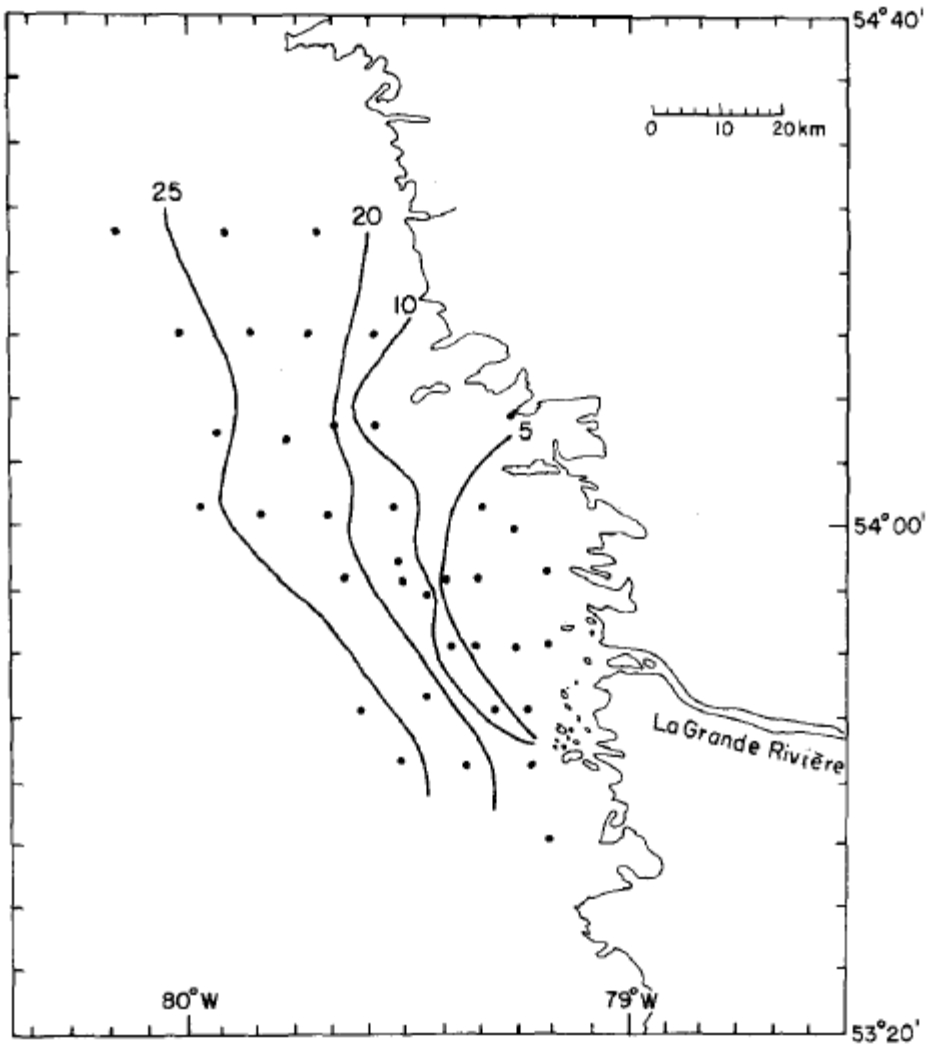


Figure 3: Distribution of the surface isohaline under the ice for February 6–16, 1984 (from Ingram and Larouche 1987)

Table 3: Areal extent of the La Grande River plume as a function of winter discharge recorded between 1976 and 1984, for different isohaline values (from Ingram and Larouche 1987)

Year	Discharge (m ³ /s)	A ₅	A ₁₀	A ₂₀	A ₂₅
1976	460	200	400	800	1800
1979	0	0	0	0	0
1980	1750	650	900	1300	2800
1984	3000	1200	1650	> 2300	> 4300

In 1987–1988, summer salinity ranged from 10 to 20 ppt, whereas in winter, coastal salinity was more variable due to freshwater plumes from various tributaries (Lalumière 1988a).

Hydro-Québec studied the winter plume of the La Grande River as a function of variations in its flow between 1986 and 1995. The extent of the impact of the freshwater plume on the salinity of the coastal waters of James Bay was set at the 20 isohaline, i.e., the freshwater plume is considered to extend to the curve where water salinity is equal to 20 ppt. In 1986–1987, it was concluded that, although the area of the plume had increased significantly for flows between 500 and 1500 m³/s, the land-fast ice formed a barrier beyond which intense mixing of freshwater and saltwater occurred. The effect of the fortnightly tidal variations on mixing and plume dynamics was considered more significant than that caused by variations in discharge (Messier et al. 1988). In 1987–1989, for estimated discharge values ranging between 3700 and 4000 m³/s, the area of the plume reached 2000 km². The central part of the plume, delineated by the 5 ppt isohaline, was located between the southern part of Dead Duck Bay (35 km south of the mouth of the La Grande River) and the southern part of Bay of Many Islands (30 km to the north). The 20 ppt isohaline extended from the Comb Islands (60 km south of the mouth) to Attikuan Point (50 km to the north). The core of the plume extended about 10 km further south in 1989 than in 1987 (Messier 2002; see Figure 4).

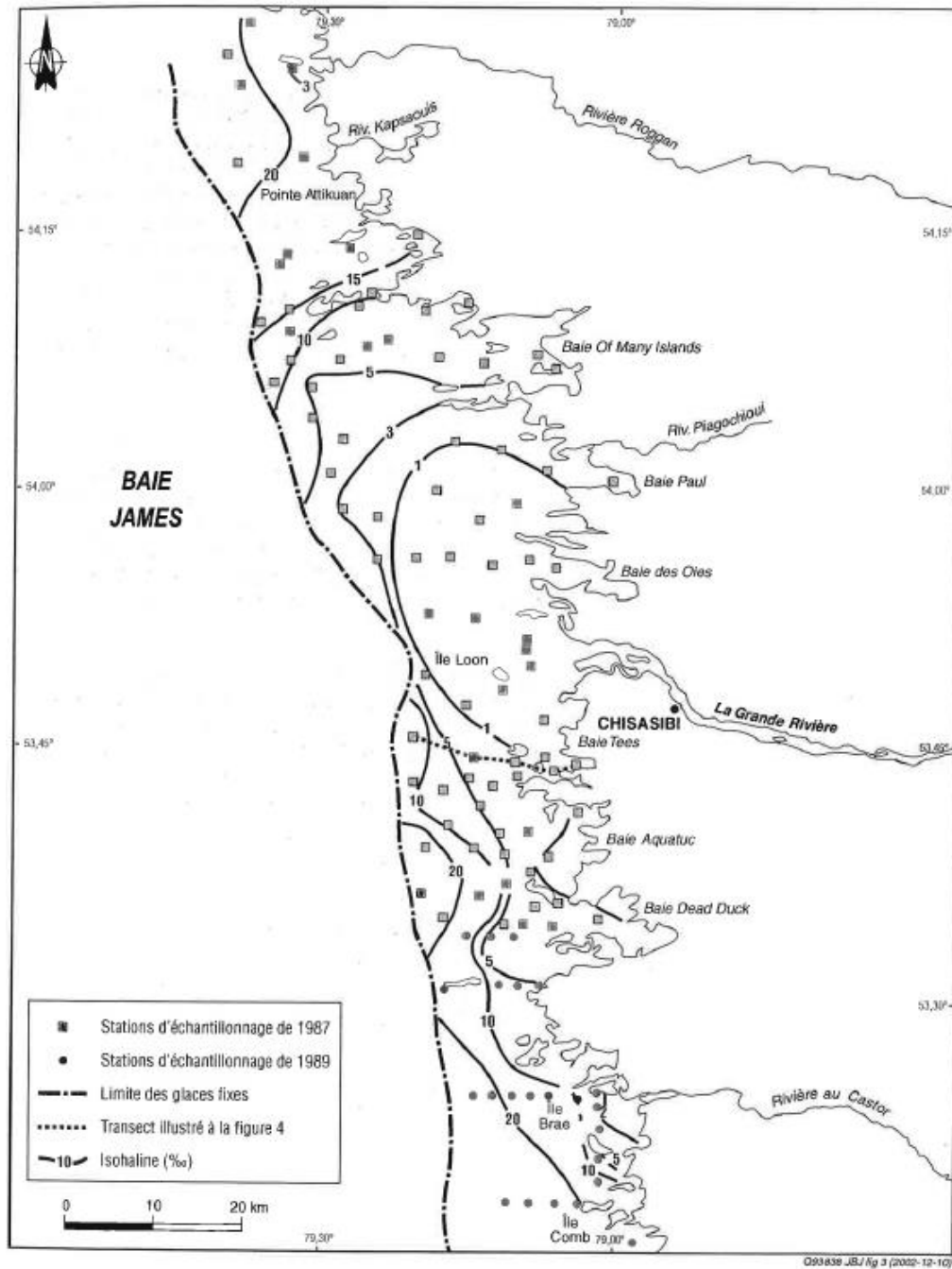


Figure 4: Distribution of surface salinity contours of the winter plume of the La Grande River based on 1987 and 1989 data (from Messier 2002)

Translation key for Figure 4

1987 sampling stations

1989 sampling stations

Fast ice boundary

Transect illustrated in Figure 4

Isohaline

James Bay

Roggan River

Kapsaouis R.

Attikuan Point

Bay of Many Islands

Piagochiou R.

Paul Bay

Baie des Oies

Loon Island

La Grande River

Tees Bay

Aquatuc Bay

Dead Duck Bay

Brae Island

Castor River

Comb Island

In 1993–1995, the typical discharges of the La Grande River were 4600 m³/s and 4400 m³/s, respectively, and the area of the plume ranged from 3200 to 3500 km² and from 2100 to 2800 km², respectively. In February 1993, the core of the freshwater plume, delineated by the 5 ppt isohaline, covered 1637 km². The 10 ppt isohaline covered 2055 km². Significant variability in the position of the 15 ppt and 20 ppt isohalines was measured (Messier 2002; see Figure 5).

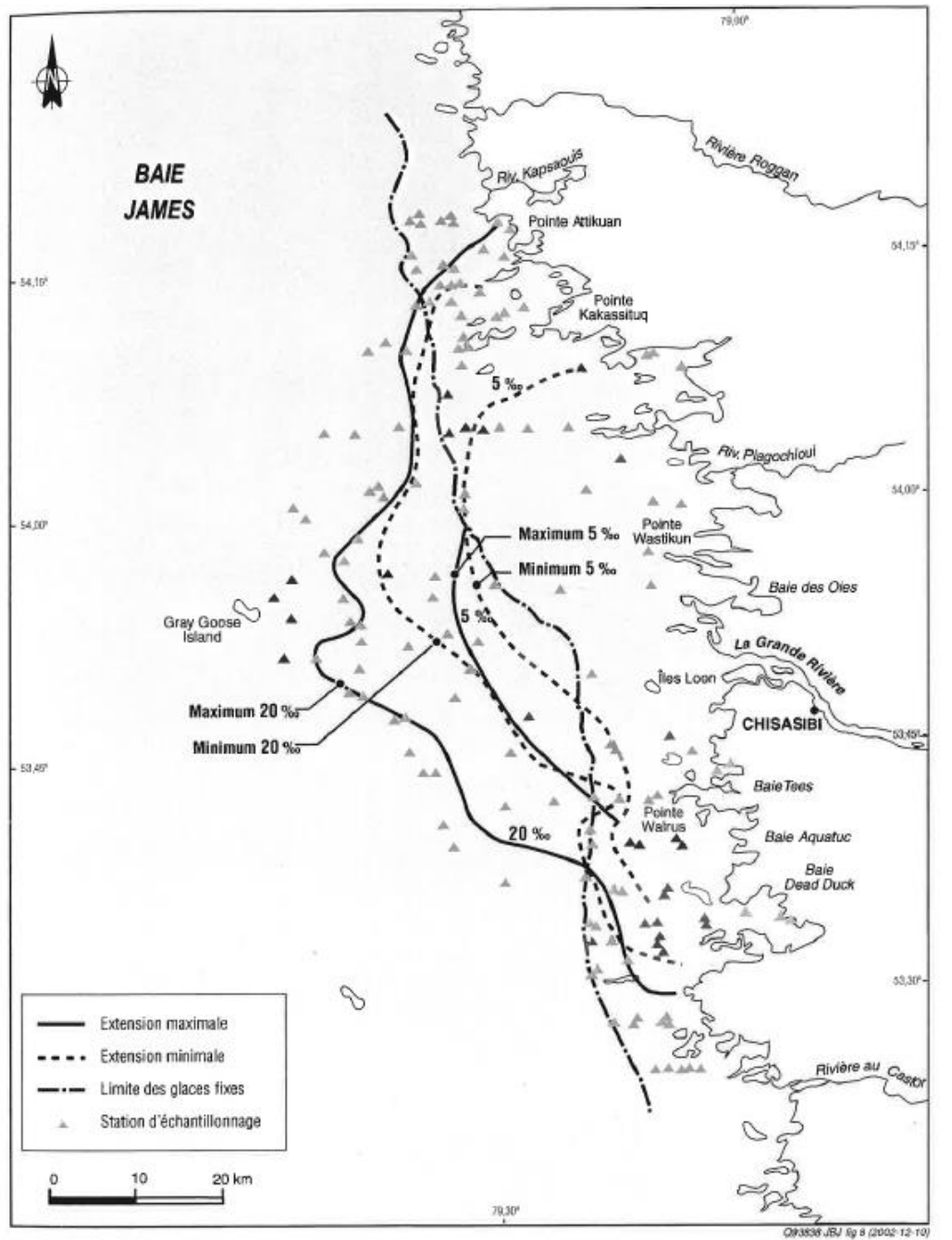


Figure 5: Plume of the La Grande River under ice cover in February 1995 (from Messier 2002)

Translation key for Figure 5

Maximum extent
Minimum extent
Fast ice boundary
Sampling station
James Bay
Roggan River
Kapsaouis R.
Attikuan Point
Kakassituq Point
Piagochioui R.
Wastikun Point
Baie des Oies
Loon Islands
La Grande River
Tees Bay
Walrus Point
Aquatuc Bay
Dead Duck Bay
Castor River

When the La Grande River discharge increased from 1700 m³/s to over 4400 m³/s, the area of the plume increased from 1600 km² to either 2100 km² or 3500 km², depending on the minimum or maximum extent observed. The ice plume extended over a strip of coast 15 to 20 km wide (Messier 2002).

The salinity of the bays was also studied in February 1993 and 1995. At the head of Bay of Many Islands, salinity ranged from 13 to 17 ppt, whereas it was only 5 ppt at its entrance. These results suggest that the presence of many islands and shoals contributes to intensifying the mixing of plume surface waters with the deeper saltier waters (Messier 2002).

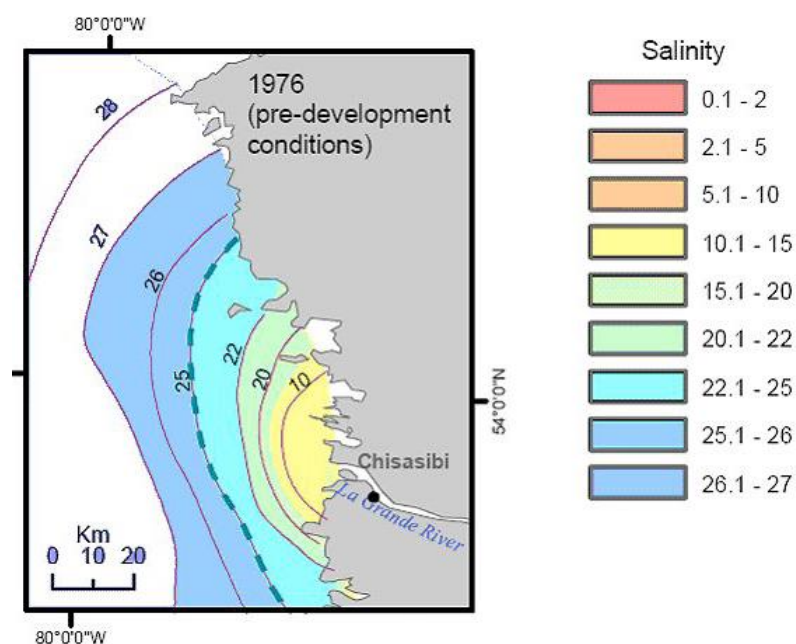
In April 1993 and 1995, the salinity of the waters of James Bay was higher than that measured in February. The presence of large areas of ice-free water promoted greater mixing of surface water and saltwater (Messier 2002).

These studies conclude, however, that tide and river discharge are not the only factors affecting the extent of the La Grande Rivière plume, although the other factors could not be specifically identified. The tidal regime is recognized as one of the main factors responsible for mixing. Wind is another important factor since it influences the width of the ice-free zones, which have a significant effect on mixing conditions and therefore on the area of the plume (Messier 2002).

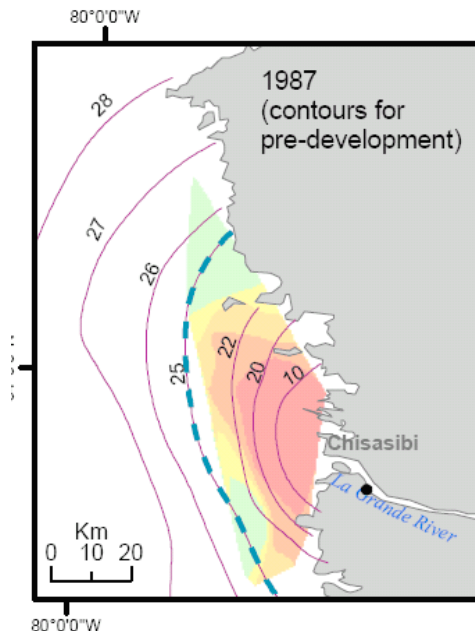
According to the work of Hamilton and Whittaker (2005, from Short 2008), who represent the NTK, salinity measured at the mouth of the La Grande River following the start-up of the hydroelectric complex was below 2 ppt, and there was a low-salinity area (≤ 10 ppt) extending out to a higher salinity area of 27 ppt. According to the authors, the projected area of the La Grande River plume will lead to the presence of an ever-increasing low-salinity area at the mouth of the La Grande River (see Figure 6). The report of the federal review panel on the Eastmain-1-A and Rupert diversion project echoes the words of the NTK, suggesting that the

water flowing into Hudson Bay has had a lower salinity level since the start-up of the La Grande complex and that the diversion of the Rupert River will intensify this decrease on the east coast of Hudson Bay and the Belcher Islands. The panel estimates the decreases in salinity to be between 1 and 3 ppt. The NTK report suggests that the strong winter flow of the La Grande River would be transported along the coastline up to Hudson Strait and the Labrador Sea, due to surface currents and the movement of large quantities of freshwater along the east coast of Hudson Bay. According to one expert consulted as part of the federal review, the La Grande River plume should not reach Hudson Bay (Gratton 2006), whereas another expert (Ingram 2006) was of the view that that these scenarios predicting present and future salinity patterns, cannot be supported owing to a lack of empirical data. The federal review panel concluded that there were no field observations confirming these patterns (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006).

A



B



C

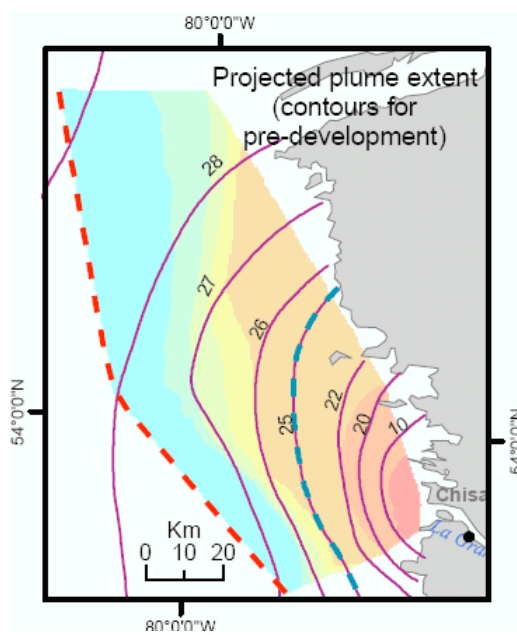


Figure 6: Position of salinity contours at the mouth of the La Grande River. A) Salinity of James Bay prior to the La Grande hydroelectric development. Salinity was 10 ppt at the mouth of the La Grande River, increasing to 25 ppt before mixing with the saltier waters of James Bay. B) Salinity of James Bay following start-up of the hydroelectric complex. Salinity was below 2 ppt at the mouth of the La Grande River, and a low-salinity area (10 ppt) extended out to the pre-development salinity contour of the 27 ppt isohaline. C) Projected distribution of salinity contours. The presence of an ever-increasing area of low salinity at the mouth of the La Grande River is anticipated (Hamilton and Whittaker 2005, from Short 2008).

More recently, in 2006–2007, coastal salinity status north of the La Grande River was examined as part of a study conducted by University of New Hampshire researcher Frederick T. Short. Between July 2006 and August 2007, salinity along the east coast of James Bay north of the mouth of the La Grande River was measured by volunteers from the Cree community of Chisasibi. The monitoring study revealed that surface salinities measured within the La Grande River plume were highest in late summer (August/September) and lowest in late winter (February/March) under the ice. The low salinities appeared to remain until June, depending on the flows recorded (see Figure 7). In winter, the freshwater plume extended further north; in summer, it contracted to the vicinity of station 001, where no eelgrass was found (see Figure 8). The results of this study suggest the presence of a spatial salinity gradient along the coast north of the La Grande River (Short 2008).

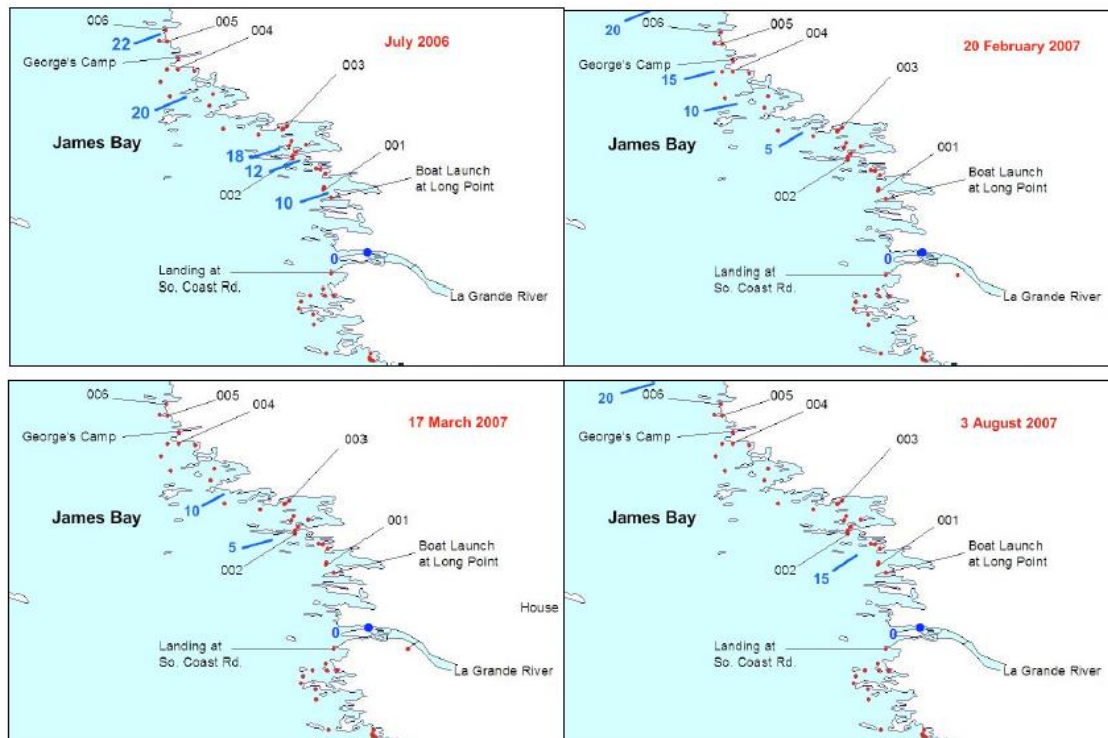


Figure 7: Salinity of the surface waters of James Bay north of the mouth of the La Grande River. Blue bars and numbers indicate the approximate position of isohalines within the La Grande River plume area (from Short 2008).

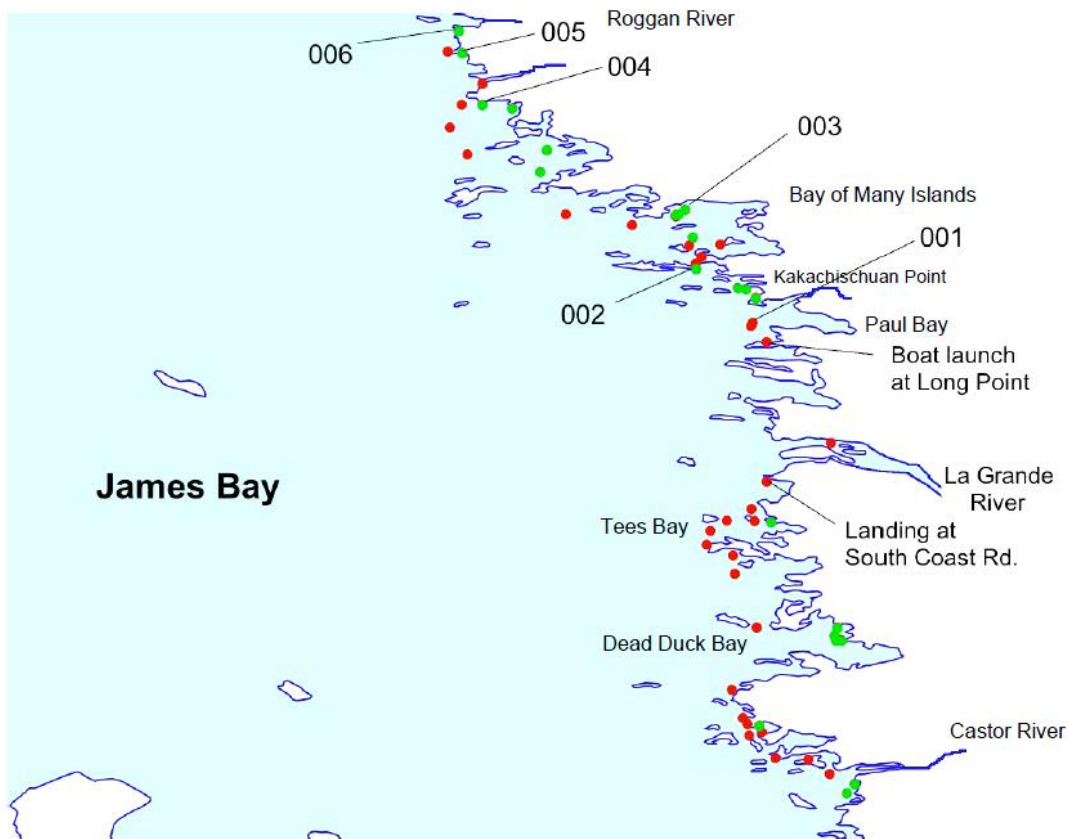


Figure 8: Sampling stations visited during the 2006 survey by the University of New Hampshire along the northeast coast of James Bay. Green dots are locations where eelgrass was found; red dots are locations with no eelgrass. Salinity and other environmental parameters were measured at all stations (from Short 2008).

In interviews held with the Cree community of Chisasibi on February 25 and 26, 2013, the participants noted a decrease in salinity in the bay in the Pointe Louis-XIV sector. They feel that the freshwater plume is more extensive than Hydro-Québec claims it to be (Interview, Chisasibi Cree community, 2013).

Water quality/turbidity

Increased discharge from the La Grande River can amplify shore erosion and lead to an increase in the quantity of suspended sediments within the freshwater plume, thereby increasing water turbidity.

Based on water samples collected at the mouth of the La Grande River in the summer of 1974 and winter of 1975, it was possible to conclude that the river did not contribute significantly to nutrient enrichment of James Bay (Grainger and McSween 1976, from Messier 2002). In the first winter following the start-up of the Robert-Bourassa power plant, an analysis conducted by Freeman et al. (1982) indicated that the increase in winter discharge seemed to have had little impact on nutrient concentrations in James Bay (Messier 2002).

Between July and September 1983, study of the physicochemical conditions of the plume water and marine coastal waters was carried out along the coast of James Bay for the James Bay Energy Corporation (Roche 1983, from Messier 2002). According to that study, concentrations of sestonic elements were higher in the La Grande River plume than in the waters of James Bay. Water quality monitoring conducted from 1987 to 2000 (Messier 2002) demonstrated that inputs of nutrients and sestonic elements from the La Grande River to the coastal waters of James Bay were not significant in winter (Messier 2002). According to Short (2008), the information provided by Hydro-Québec does not provide enough evidence to make conclusions one way or the other.

First-hand accounts gathered from the Cree community in the 1990s indicate that since the La Grande complex began operations, the increased discharge and sediment concentrations have directly affected Aboriginal fisheries. One member of the community stated that he could no longer set out his fishing nets in certain locations due to the large amount of debris that would get caught in them. In addition, residents of the La Grande River region indicated that they no longer used the river as a source of drinking water, which they describe as previously being very pure (McDonald et al. 1997).

Similar comments were heard in the interviews conducted at Chisasibi in June 2006. Some residents reported that, since the construction of the hydroelectric facility, they no longer consumed water directly from the La Grande River. Since the changes to the discharge of the La Grande River, the residents have noted ongoing erosion of the river's banks, particularly in winter (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006).

In interviews held with the Cree community of Chisasibi on February 25 and 26, 2013, the participants expressed their concern about the water quality of the La Grande River, which had become very murky (Interview, Chisasibi Cree community, 2013). According to their observations, the water from the La Grande River that flowed into the bay was much clearer before the La Grande complex began operations. They stated that they used to be able to see 20 feet deep in the bay before the facility was built. The only time the water used to be turbid was during the spring breakup. The Cree expressed concerns about the amount of suspended sediments in the water. They noted that the waters surrounding Bear Islands and Long Islands were more turbid than they used to be. They also attributed the lack of seals in the sector, a phenomenon observed by hunters, to the increased turbidity in James Bay.

The Cree also noted the presence of a viscous, greenish yellow substance deposited in layers on the bottom of James Bay that stuck to their fishing nets. At the northern end of the estuary, the viscous deposit is exposed at low tide. It was not observed near the mouth of the La Grande River but was clearly observed in the bay. The deposit had been observed before the La Grande complex began operations, but was 10 times smaller at the time. This change was observed after the construction of the La Grande complex (Interview, Cree community of Chisasibi, 2013).

Water temperature and ice regime

The La Grande River

The temperature variations measured in winter are affected by thermal stratification in the Robert-Bourassa reservoir, which in turn is heavily influenced by the meteorological conditions

preceding freeze-up in November and December (Saucet 2002). In the spring, the presence of the Robert-Bourassa reservoir results in a two- to three-week lag in the water warming curve. The maximum summer temperatures of the La Grande Rivière are 6 to 8°C lower (Saucet 2002). In winter, the increased discharge and effect of the reservoirs on water temperature have a significant effect on the ice regime by preventing ice cover from forming (Saucet 2002). The number of days on which ice cover is present on the river varies considerably. For example, the duration of the ice cover at Fort George fell from 125 days early in the operation of La Grande-2 in 1979–1980 to 74 days in 1986–1987 (JBEC 1988; from Saucet 2002). Safe crossing of the river by snowmobile at its mouth is no longer possible in spring (Saucet 2002).

First-hand accounts gathered from the James Bay Cree community in the 1990s indicate that the increased winter flows of the La Grande River prevent solid ice from forming on the river, which has become uncrossable near Chisasibi. According to the Cree, the amount of ice began to decrease when operations began at the La Grande-2A power plant (McDonald et al. 1997).

At the sessions held in Chisasibi in June 2006 as part of the Eastmain-1-A and Rupert diversion project, a number of Cree stated that they had noted major changes in the ice conditions of the La Grande River since the construction of the La Grande complex. The water does not freeze as much as it used to, and the ice cover is less solid since the winter discharge increased. The river never really freezes completely. Travelling in spring, fall and winter, especially by snowmobile, is less safe and more difficult (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006).

In the interviews held in the Cree community of Chisasibi in February 2013, the participants indicated that it had been possible to cross the La Grande River in November prior to the construction of the La Grande complex. Near the Chisasibi airport, there is now open water. It used to be possible to cross by snowmobile. At Fort George (Chisasibi), the ice used to break up in May. Today, the ice at the mouth of the La Grande River forms only in January.

Plume of the La Grande River in James Bay

On the east coast of James Bay, the ice forms in December. There are two types of ice: landfast ice, which is roughly 15 to 20 km wide, and drifting pack ice, which is located further offshore. According to Hydro-Québec, there is no relationship between the outer limit of the landfast ice and the flows of the La Grande River and other tributaries of James Bay. According to Hydro-Québec, the outer limit is determined by the drop-off from the coastal shelf (Hydro-Québec 2004).

There was a slight increase (less than 0.5°C) in winter water temperatures within the La Grande River plume and a major decrease in summer temperatures (11°C versus 16°C) (Hydro-Québec 2004). A 5°C reduction in the water temperature of the bay could substantially decrease eelgrass production and growth because the metabolism of eelgrass is reduced by half with every 10°C drop in temperature (Biebl and McRoy 1971; Short 1980).

The issue of ice formation and ice melt in James Bay and Hudson Bay is not clearly defined in the literature and is subject to various interpretations. The dates of marine ice freeze-up and breakup on Hudson Bay were studied at 36 sites across Hudson Bay and James Bay between 1971 and 2003. The results indicate a trend toward earlier breakup in James Bay and along the southern and western shores of Hudson Bay, while there is a trend toward later freeze-up in the

northern and northeastern regions of Hudson Bay. According to Gagnon and Gough (2005), these changes in the annual ice cycle coincide with both the regional temperature record and projections from general circulation models. Their study supports the 1998 study of Saucier and Dionne (from Gratton 2006), who report that climate variability is likely the most important parameter in explaining the variability of ice cover.

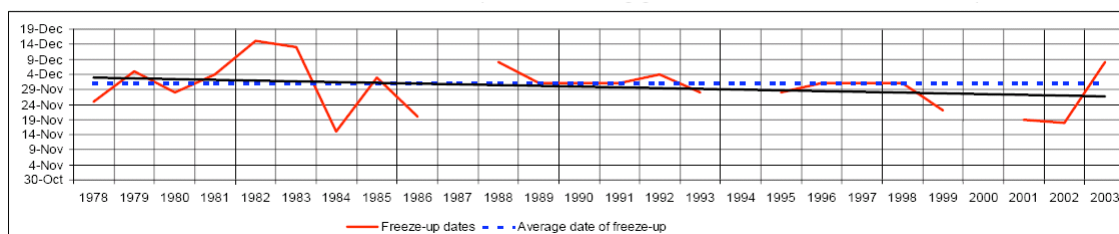
Gratton (2006), as an expert consulted by the Environmental Assessment Branch on the results of the above-mentioned impact study, supports the idea that climate variability is likely the primary factor in the variability of ice cover. However, he is of the view, as is a second expert consulted on the matter (Ingram 2006), that variations in ice cover should be compared against the variations in discharge of the La Grande River and against climate variations in order to better assess the role of each of these factors in the duration of ice cover (Ingram 2006).

First-hand accounts gathered from James Bay Cree report a change in ice conditions in the estuary and near the coast. The ice is not as solid as it once was, it forms later in the fall, and it breaks up earlier in the spring. In addition, open water areas form at unexpected locations (McDonald et al. 1997).

In interviews conducted in Chisasibi in February 2013, community members also mentioned that since the La Grande complex began operations, ice freeze-up occurred later in the fall and ice breakup occurred earlier in the spring (Interview, Chisasibi Cree community, 2013). They also noted the presence of sediments in the ice, which make it more porous and fragile, resulting in faster ice melt in spring (Interview, Cree community of Chisasibi, 2013).

On the other hand, lower-salinity waters in James Bay could have an impact on the duration of winter ice cover, since freshwater freezes at 0°C while saltwater freezes at -4°C (Hamilton and Whittaker 2005). The NTK experts are of the opinion that the ice cover on James Bay forms earlier in the year and ice breakup occurs later in the spring than in past decades, due to the lower salinity water of the La Grande River plume (see Figure 9 of Hamilton and Whittaker 2005, from Short 2008). This view, supported by Short (2008), was advanced by Hamilton and Whittaker (2005) on behalf of the NTK, in response to questions by the proponent in the environmental impact statement of the Eastmain-1-A and Rupert diversion project.

A



B

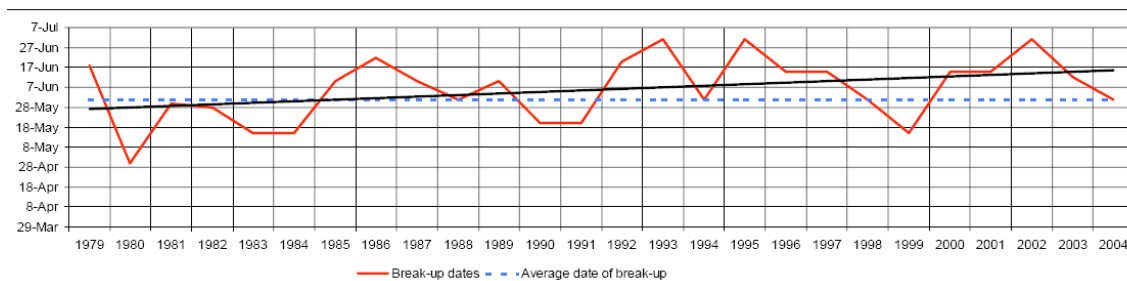


Figure 9: Changes in the La Grande River plume over recent decades. A) Freeze-up dates. B) Breakup dates (from Hamilton and Whittaker 2005, cited in Short 2008)

The NTK experts feel that the increase in ice concentration observed in the eastern portion of Hudson Bay and around the Belcher Islands could be explained by the inflow of freshwater to these areas from the La Grande complex (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006). Rapid freeze-ups consisting of more freshwater ice than saltwater ice are observed more frequently in Hudson Bay. In the winter of 2012–2013, sudden freeze-ups in eastern Hudson Bay, coinciding with record hydroelectric demand, led to entrapments of marine mammals in ice. Over 70 belugas were trapped south of the Belcher Islands, and an entrapment of killer whales occurred north of the Belcher Islands near Inukjuak (Arctic Eider Society 2013; NunatsiaqOnline 2013).

In winter, the salinity of Hudson Bay is normally about 30 ppt. Natural river inputs occur in the spring. Due to hydroelectric developments in Quebec, freshwater is held back in reservoirs to meet the increased energy demand during winter. As a result, the largest freshwater flows to the bays are in winter. One of the problems with freshwater plumes occurring in winter is that freshwater freezes quicker and at warmer temperatures than salt water, creating fast-freeze areas that can trap ducks and marine mammals (Arctic Eider Society 2013; and NunatsiaqOnline 2013). It should be noted that the relationship between hydroelectric development and increases in ice concentration has not been proven (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006).

Coastal expansion

First-hand accounts gathered from James Bay Cree in the 1990s indicate that coastal expansion has become very apparent along James Bay and the southwest coast of Hudson Bay. Herbaceous plants and trees are now growing in the tributaries of rivers in southern James Bay and in rivers altered by the construction of hydroelectric plants. The coastal vegetation has significantly altered available waterfowl habitat over the last 50 years, which may have led to changes to geese migration routes (McDonald et al. 1997).

2.2 Distribution and abundance of eelgrass beds

In recent decades, there have been changes in the distribution and abundance of eelgrass beds in James Bay. However, it is difficult to assess these changes due to the lack of available data. Most of the available information on the changes in the status of eelgrass beds comes from

reports produced by Hydro-Québec as part of its mandate, which includes eelgrass monitoring (Phase II of the La Grande complex and Eastmain-1-A and Sarcelle powerhouses and the Rupert diversion project). Some information on the state of these beds is available for 2006–2007 in the report by Short (2008). Information from Aboriginal traditional knowledge of the James Bay Cree community is also available.

Monitoring by Hydro-Québec

To assess the potential impacts of the La Grande hydroelectric complex on eelgrass, exploratory studies were initiated in 1982 (Roche ltée 1982) on the east coast of James Bay. Three field seasons were undertaken in the summers of 1982, 1985 and 1986, resulting in the production of a map of 1982 eelgrass distribution on a portion of the east coast of James Bay. In addition, eelgrass sampling was carried out in three sectors that served as reference stations for the seagrass bed monitoring program that was to follow (Roche ltée 1982). Eelgrass studies really began in 1986, when two new permanent sampling stations were established for eelgrass monitoring (Lalumière 1986).

In 1987–1988, a characterization study of eelgrass was conducted to better understand the main biological components associated with eelgrass beds and to characterize the biological and ecological relationships that exist in the beds. As a result of this study, it was possible to establish eelgrass distribution along the northeast coast of James Bay, define the extent of annual variations in eelgrass distribution and production, and standardize the monitoring method (Lalumière 1988a).

In 1989, the Canadian Wildlife Service and the James Bay Energy Corporation carried out a study of the habitats of the northeast coast of James Bay and their use by waterfowl (Dignard et al. 1991; Benoit et al. 1995, 1996; Reed et al. 1996).

Beginning in 1991, Hydro-Québec, through the consulting firm GENIVAR, began actual monitoring of the potential impacts of the La Grande hydroelectric complex project on eelgrass. Eelgrass was the main biological component of the environmental monitoring program indicated on the certificate of authorization for the La Grande-2-A power plant. Given that the operation of that station was to lead to an increase in the freshwater plume in winter, it was possible that several eelgrass beds would be exposed to lower salinities over the winter, potentially resulting in changes in their distribution and productivity.

Because the construction of the Phase II power plants had been completed since 1996, the James Bay Energy Corporation transferred its environmental monitoring obligations for Phase II to Hydro-Québec. The monitoring master plans expired in 2000. Summary reports on the various environmental components were produced following the completion of the monitoring programs, including a general summary report on changes to the eelgrass beds on the northeast coast of James Bay from 1988 to 2000, and more precisely from 1988 to 1995 and from 1999 to 2000 (Lalumière and Lemieux 2002). In 2004, Hydro-Québec re-evaluated the status of eelgrass beds in James Bay, on a purely qualitative basis, in response to the dramatic decline in eelgrass beds along the coast of James Bay in 1998 and 1999.

As part of the project to develop the Eastmain-1-A and Sarcelle powerhouses and Rupert diversion, the James Bay Energy Corporation and Hydro-Québec submitted an environmental monitoring program (2007 to 2023), including the monitoring of eelgrass along the northeast

coast of James Bay. The monitoring was divided into four periods: pre-Rupert River diversion and three years following its start-up (summers of 2011, 2014 and 2019). The Eastmain-1-A and Sarcelle power plants were scheduled to begin operation in 2011 and 2012, respectively (Consortium Waska-GENIVAR 2011).

Method used in the studies and monitoring

The eelgrass monitoring program conducted between 1988 and 2000 consisted of two main components: the distribution of the beds and eelgrass production (shoot density and leaf dry biomass). Saltmarsh monitoring was added to the program in order to measure the effects of isostatic uplift on coastal vegetation (Lalumière and Lemieux 2002).

Eelgrass distribution

Three maps of eelgrass bed distribution on the northeast coast of James Bay were produced during the monitoring program; they cover the periods 1986–1987, 1991–1992 and 1996. The maps were used to study the changes in eelgrass bed distribution over time. The 1986–1987 map represents the reference state of the eelgrass beds prior to the start-up of the La Grande-2-A power plant in October 1992, namely Phase II of the project (Lalumière and Lemieux 2002). It is a follow-up to the maps produced by Curtis in 1974–1975 and Roche Itée in 1982.

For the three years of monitoring, the assessment of eelgrass bed coverage comprised only two classes: high density (continuous eelgrass coverage or discontinuous coverage where plant coverage is >50%) and low density (loose coverage or patches where the proportion of denuded areas is >50%). For each year, 1:125 000 scale maps were produced based on the interpretation of aerial photographs (except for 1991–1992) and helicopter overflights. Several dives were carried out to determine the lower limit of some beds and to assess the presence of eelgrass when the water surface was too rough to obtain a good visual detection from the helicopter. The 1996 map is the most precise of the three (Lalumière and Lemieux 2002).

Starting in 1999, dives were conducted to assess the qualitative status of eelgrass beds. A total of 75 dive monitoring points were identified at low tide from Pointe Louis-XIV to north of Eastmain. These points were added to the monitoring program following the 1998–1999 decline, which resulted in an overall decline in eelgrass abundance at five of the six permanent stations. All or some of these points were revisited in 2000, 2004, 2009 and 2011 to obtain a qualitative assessment of the changes in eelgrass abundance. Eelgrass abundance was defined on the basis of the following classes: lack of eelgrass, sparse bed, discontinuous bed and continuous bed. Depending on the year, measurements of temperature, surface salinity and turbidity, among other parameters, were taken at the stations. The visual observations made during the overflights were complemented by aerial photographs (Lalumière and Lemieux 2002; Consortium Waska-GENIVAR 2011).

Productivity of eelgrass beds

Preliminary studies provided relevant information on the dry biomass and density of eelgrass shoots between 1982 and 1990. In 1982, three permanent eelgrass sampling areas were selected as part of exploratory studies. The actual monitoring program ran from 1991 to 2000; eelgrass samples were collected every year at each of the six permanent stations, except 1993,

when only the Attikuan I and Kakassituq stations were sampled. The six permanent sampling stations were selected to include areas both north and south of the La Grande River, and in sectors which, according to the proponent, were within and outside the winter freshwater plume under the operating conditions of the La Grande-2-A power plant (Lalumière and Lemieux 2002). Figure 10 illustrates the location of the six permanent sampling stations, while Table 4 presents their sampling periods since 1982.

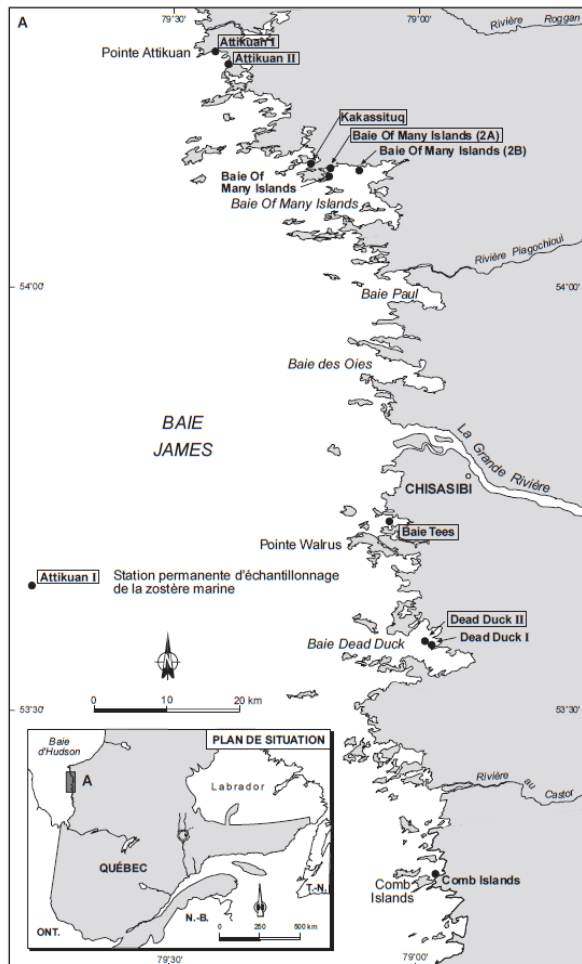


Figure 10: Six permanent sampling stations monitored by Hydro-Québec from 1988 to 2000 (from Lalumière and Lemieux 2002)

Translation key for Figure 10

Permanent eelgrass sampling station

Site plan

James Bay
 Roggan River
 Attikuan Point
 Bay of Many Islands
 Piagochioui R.
 Paul Bay
 Baie des Oies
 La Grande River
 Tees Bay
 Walrus Point
 Dead Duck Bay
 Castor River

In inset map:
 Hudson Bay
 Quebec
 NB
 NL

Table 4: Eelgrass sampling stations on the northeast coast of James Bay for the period 1982 to 2000 (from Lalumière and Lemieux 2002)

Year	<i>Sampling Stations</i>									
	Attikuan I	Attikuan II	Kakassituq	2A	2B	Bay of Many Islands	Tees Bay	Dead Duck I	Dead Duck II	Comb Islands
Eelgrass bed characterization and method standardization period										
1982			x					x		x
1985			x					x		
1986	x		x				x	x		
1987			x	x	x					
1988	x		x				x			
1989	x	x	x	x		x	x	x	x	
1990			x							
Monitoring period										
1991	x	x	x	x			x		x	
1993	x		x							
1994	x	x	x	x			x		x	
1995	x	x	x	x			x		x	
1999	x	x	x	x			x		x	
2000	x	x	x	x			x		x	

Permanent stations are indicated in boldface.

The sampling protocol used to monitor the eelgrass beds from 1991 to 2000 and in 2009 involved the location of plots by a diver, using precise survey equipment, and the collection of two eelgrass samples at separate depths along five transects perpendicular to the shore. For each sample, the number of vegetative and flowering shoots per square metre (N/m^2), the dry biomass of the shoots per square metre (g/m^2), and the mean dry biomass per shoot ($g/shoot$) were calculated (Lalumière and Lemieux 2002; GENIVAR 2010).

Gathering of Cree traditional knowledge on the species and the environment

One-on-one interviews were conducted on August 20, 2009, at the Niskamoon Corporation office in Chisasibi with two Cree trapline users, followed by two other individual meetings on November 5, 2009, at Chisasibi with the Cree tallymen of two other traplines. However, the traditional knowledge gathered was fragmented and, as a result, the entire coast could not be exhaustively documented (GENIVAR 2010). In 2011, interviews were conducted with about 10 Cree trapline users from July 25 to 29 (Consortium Waska-GENIVAR 2011).

Isostatic uplift

In order to monitor the changes in the distribution of eelgrass at shallow depths as a function of isostatic uplift, the status of the beds was verified at 10 control points located between Dead Duck Bay and Attikuan Point in 1993, 1999, 2000, 2004, 2009 and 2011 (Consortium Waska-GENIVAR 2011).

Changes in eelgrass beds documented by Hydro-Québec (JBEC) between 1975 and 2011

Baseline status of the eelgrass beds of James Bay prior to Phase II

In 1975, a map of eelgrass bed distribution along the northeast coast of James Bay was produced by Steve Curtis, an Environment Canada biologist who, in the early 1970s, was one of the first researchers to study James Bay eelgrass beds. The aerial survey clearly showed the presence of eelgrass in all saltwater or brackish water shoals and its almost complete absence near the mouth of the La Grande River. Eelgrass cover typically reached 11 to 40% but could be as high as 70 to 100% in more favourable sectors, such as the bays in the areas of Kakassituq Point, Dead Duck Bay and Comb Islands (Roche Itée 1982). In the early 1970s, Curtis and Allen (1976) described the eelgrass beds of Dead Duck Bay as being very dense. They also considered the area located roughly between the Roggan River and Kakachischuan Point (the area encompassing the Bay of Many Islands, Kakassituq Point and Attikuan Point) as being an area of national significance due to its extensive eelgrass beds, which attracted between 10 000 and 20 000 Brant and several thousand Canada Geese in the fall, as well as ducks. They also observed the presence of dense eelgrass beds around Comb Islands and Black Whale Island (Curtis and Allen 1976).

In 1982, a second map of eelgrass bed distribution was produced, this one by Roche Itée, to update part of Curtis' map. The areas with the highest eelgrass concentrations were between Attikuan Point and Kakassituq Point, and in the areas of Dead Duck Bay and Pointe des Oblats. In these areas, at low tide, eelgrass typically covered more than 70% of all mud bottoms less than 2 m deep. These exploratory studies show that the Kakassituq Point station had the highest eelgrass biomass values. A complete absence of eelgrass was observed at the mouth of the La Grande River, between Baie des Oies to the north and Tees Bay to the south. The coastal area from which eelgrass was absent was larger in the north sector than in the south sector. The results of the study conducted by Roche Itée indicate that eelgrass distribution remained relatively unchanged from 1974 to 1982 (Roche Itée 1982).

Between 1982 and 1986, there was no decline in eelgrass biomass. In 1986, the largest beds were surveyed at Kakassituq Point and Dead Duck Bay (Lalumière 1986).

In 1987, eelgrass still appeared to be absent from the large bays on either side of the mouth of the La Grande River, from Kakachischuan Point to Tees Bay. Elsewhere, eelgrass appeared to colonize most bays to varying degrees. The waters in which the eelgrass beds were located were definitely oligotrophic, typically characterized by low turbidity or by high turbidity over short periods. Summer salinity appeared to be suitable for eelgrass growth, whereas winter salinity was highly variable and particularly low near the freshwater plume of the La Grande River. The water temperature was suitable for the completion of the species' life cycle, and the water and sediment quality appeared to be suitable for its growth. According to Lalumière, a comparison of the three eelgrass distribution maps shows that the species' distribution had not changed significantly between 1974 and 1987 (Lalumière 1987b). The monitoring of permanent eelgrass stations by the James Bay Energy Corporation (JBEC) between 1988 and 1989 showed a slight decline in green leaf biomass but a significant increase in shoot density. The leaves were shorter, but there were more shoots. The study's authors concluded that there had been no decline in eelgrass production in the beds monitored (Lalumière and Belzile 1989). A comparison of the 1987 and 1991 eelgrass distribution maps by Lalumière et al. (1992) showed that the species' distribution had not changed significantly over that period. The location of the main concentrations was unchanged. However, significant differences were observed at shallow depths. The authors suggest that these changes are a reflection of the coastal dynamics and that isostatic uplift may potentially have a slow and gradual effect on eelgrass distribution and density. No general upward or downward trend was observed, with eelgrass production increasing at some stations and decreasing at others (Lalumière et al. 1992).

Phase II monitoring period: 1988–1996

According to reports provided by Hydro-Québec, eelgrass distribution was relatively stable over the monitoring period preceding the decline of 1998–1999, i.e., between 1988 and 1996. The locations of the main high-density and low-density eelgrass beds remained unchanged. However, eelgrass production in shallow waters seemed to have declined on the entire coast. The beds appeared to have moved offshore, moving by up to a few hundred metres in some locations (Lalumière and Lemieux 2002). Eelgrass production exhibited marked annual variations at all stations and for all depths. These variations differed from station to station, and even from one depth to another at a given station. As a general rule, dry biomass and number of shoots were more variable at the lowest depth sampled (0.5 m) than at the other depths. The number of flowering shoots was generally higher at 0.5 m than at other depths. At shallow depths (0.5 m and 1 m), downward trends in eelgrass production were measured for all stations except that of Kakassituq, while at depths of 1.5 m and 2 m, upward trends were measured for all permanent stations except Tees Bay.

For all sampling stations located at a depth of 1.5 m, except that of Tees Bay, there was a positive relationship between the number of shoots and the number of growing degree-days. The relationship was particularly strong for the station at Kakassituq, where the number of growing degree-days accounted for close to half of the annual variation in shoot density (Lalumière and Lemieux 2002).

The comparison of the eelgrass distribution maps for 1982, 1986–1987, 1991–1992 and 1996 can also provide other information about the nature of the changes that occurred in the beds over the time.

Phase II monitoring period: Period of major decline (1998–2000)

Following the major eelgrass decline in 1998–1999, the plants were slow to recover. There was a general decline in eelgrass along the entire coasts of James and Hudson Bays. Monitoring carried out in 1999 showed very tentative signs of recovery in some bays, and new shoots were observed. Leaf biomass had declined by 94 to 99% compared to 1995 at five of the six permanent stations. Only one station was spared, that of Tees Bay, where leaf biomass had even increased since 1995 at a depth of 1 m, as had the average number of shoots at all depths measured. The number of flowering shoots had also increased compared to 1995. In contrast, the mean dry biomass per shoot had declined at all depths and the shoots were the smallest ever recorded at that station since the start of monitoring. The south part of the Bay of Many Islands had also been spared. The number of contiguous beds fell from 57% prior to the decline to 8% in 1999. Sparse, patchy beds accounted for 72% of the beds in 1999, compared to 43% prior to the decline. Lastly, 20% of the points surveyed in 1999 contained no eelgrass. However, the eelgrass populations appear to have remained stable within protected bays, although the density of the beds declined. The beds were less extensive and less abundant at sites more exposed to wind and waves along the coast. In 2000, observations made at the monitoring points showed that at several locations, the percent cover had remained unchanged or had increased slightly. According to Hydro-Québec reports, the locations in question were north and immediately south of the La Grande River (Lemieux et al. 1999; Lemieux and Lalumière 2004).

Representatives of the Cree community of Chisasibi were interviewed by Hydro-Québec in October 1999. They stated that the decline in question had been observed in shallow bays in 1997. They suggested isostatic uplift, climate change and the impact of the La Grande River plume in the bay as possible explanations for the decline (Lemieux et al. 1999).

Post-decline period (2004–2011)

Following the 1991–2000 prescribed monitoring period, Hydro-Québec resumed monitoring in 2004, five years after the massive eelgrass die-off along the coast of James Bay. Low-altitude flights were conducted over the six permanent stations at spring low water to assess changes since 2000. No sampling was carried out at these stations, but photographs of each station were taken.

In 2004, eelgrass abundance remained stable at most monitoring points visited (50 to 70% of points had levels similar to those observed in 2000). North of the La Grande River, increased eelgrass abundance was observed to north of Attikuan Point, where the beds were making a very tentative recovery. In most eelgrass beds in the Bay of Many Islands, biomass was similar to what it had been prior to the decline. Immediately south of the La Grande River, eelgrass abundance was similar to levels prior to the decline, whereas in the area from south of Tees Bay to north of Eastmain, the recovery was slow. In Tees Bay, which had not been affected by the decline, the beds were still in good shape. The recovery was generally better in protected, gently sloping bays, where eelgrass growing conditions were favourable (Lemieux and Lalumière 2004).

In 2009, 60% of the eelgrass beds visited had abundance levels similar to pre-decline levels, 16% showed an uneven recovery, and 24% were slow to recover. It was not possible to verify the status of the eelgrass beds north of the La Grande River due to a ban on flights. The beds were in the process of recovering, although their abundance had not returned to pre-decline

levels. Eelgrass production was highly variable from station to station. Eelgrass production at the Bay of Many Islands and Tees Bay stations was similar to what it had been prior to the decline, while beds at the Kakassituq Point and Dead Duck Bay stations were less productive but were slowly recovering. The Attikuan I and II stations had not yet begun to recover. South of the La Grande River, recovery was slow at the station furthest from the mouth but was well underway near the mouth. In short, for the stations furthest north of the La Grande River, recovery was slow at all depths sampled, whereas the stations located closer to the mouth seemed to be recovering well (GENIVAR 2010).

In 2011, eelgrass continued to recover, but it was still not as widespread or as abundant as it had been prior to the decline. The recovery appeared to be slower north of the La Grande River than south of it, particularly in the section of coast between Pointe Louis-XIV and Kakassituq Point. An increase in eelgrass was observed at 40% of the points visited, while a decline was observed at 26% of the points. The relatively abundant presence of epiphytic algae (*Ulothrix* spp. and *Enteromorpha* spp.) at several sites along the coast was observed. This was the first time since the start of the studies (1982) that algal blooms were observed (Consortium Waska-GENIVAR 2011).

Table 5 presents the annual variations in mean dry biomass per shoot at the six permanent stations from 1988 to 2009.

Table 5: Annual variations in mean dry biomass per shoot (g/shoot) at the six permanent stations (from GENIVAR 2010)

Station and depth (m)	Year									
	1988	1989	1990	1991	1993	1994	1995	1999	2000	2009
<i>Attikuan I</i>										
0.5	0.11	0.12	-	0.23	0.10	0.11	0.12	0.67	0.05	0.13
1.0	0.40	0.33	-	0.38	0.30	0.28	0.35	0.62	0.10	0.11
1.5	0.53	0.31	-	0.37	0.58	0.29	0.49	-	0.09	0.16
2.0	0.34	0.33	-	-	0.39	0.33	0.59	-	-	0.21
Mean	0.34	0.27	-	0.33	0.34	0.25	0.39	0.65	0.08	0.15
<i>Attikuan II</i>										
0.5	-	0.11	-	0.13	-	0.14	0.09	-	-	-
1.0	-	0.38	-	0.40	-	0.35	0.36	0.43	0.06	0.05
1.5	-	0.38	-	0.48	-	0.36	0.39	-	0.06	-
Mean	-	0.29	-	0.34	-	0.28	0.28	0.43	0.06	0.02
<i>Kakassituq</i>										
0.5	0.20	0.17	0.29	0.24	0.25	0.35	0.27	0.21	0.10	0.12
1.0	0.68	0.41	0.51	0.41	0.48	0.45	0.60	0.46	0.17	0.26
1.5	0.79	0.48	0.63	0.53	0.48	0.53	0.63	0.86	0.19	0.30
Mean	0.56	0.35	0.47	0.40	0.40	0.44	0.50	0.51	0.15	0.23
2A (Bay of Many Islands)										
0.5	-	0.13	-	0.36	-	0.20	0.24	0.48	0.14	0.25
1.0	-	0.22	-	0.36	-	0.22	0.38	0.40	0.23	0.25
1.5	-	0.21	-	0.27	-	0.33	0.40	0.60	0.13	0.28
Mean	-	0.19	-	0.33	-	0.25	0.34	0.49	0.17	0.26
Tees Bay										
0.5	0.23	0.22	-	0.15	-	0.18	0.15	0.11	0.08	0.21
1.0	0.28	0.27	-	0.26	-	0.22	0.22	0.18	0.10	0.28
1.5	0.41	0.26	-	0.22	-	0.27	0.29	0.23	0.13	0.36
Mean	0.31	0.25	-	0.21	-	0.22	0.22	0.17	0.10	0.28
Dead Duck II										
0.5	-	0.18	-	0.12	-	0.04	-	-	0.05	-
1.0	-	0.39	-	0.43	-	0.44	0.42	0.15	0.12	0.20
1.5	-	0.49	-	0.65	-	0.65	0.56	0.31	0.03	0.22
Mean	-	0.36	-	0.40	-	0.37	0.49	0.23	0.07	0.14

Tables presenting the results obtained by dives at each of the monitoring points in 2011 and a qualitative comparison of the state of the beds since 1996 are available in the eelgrass monitoring report produced for Hydro-Québec in 2011 (Consortium Waska-GENIVAR 2011). For some years, surface water temperature, salinity and turbidity measurements were taken at the stations.

Effect of isostatic uplift in the salt marshes (1991–2000)

The objective of this study was to show the short-term changes on vegetation caused by isostatic uplift on the east coast of James Bay. The results revealed a significant expansion of vegetation cover and changes in the species colonizing the marshes. The observed changes in the vegetation of the coastal marsh studied revealed significant expansion of the vegetation under the influence of isostatic uplift. These changes occurred over short periods of time, with the lower marsh vegetation quickly moving toward the sea (Lalumière and Lemieux 2002; Hydro-Québec and GENIVAR 2005).

Status of eelgrass assessed in 2006–2007 by the University of New Hampshire (Short 2008)

In 2006–2007, Frederick T. Short of the University of New Hampshire began an investigation into the cause of eelgrass decline in James Bay, conducting an experimental laboratory study and two field surveys (Short 2008).

In July 2006, a team composed of University of New Hampshire researchers and representatives of the Cree community of Chisasibi visited 57 sites along the northeast coast of James Bay on either side of the mouth of the La Grande River. Eelgrass bed status, temperature and salinity at the sites were noted.

No eelgrass beds were found immediately north of the La Grande River, i.e., at Long Point, where surface salinity was 5 ppt. The first eelgrass bed was found just north of Kakachischuan Point, beyond Paul Bay, where the salinity was 11 ppt. Moving northward up the coast of James Bay to the Bay of Many Islands, salinities remained below 12 ppt. At several sites, eelgrass was mixed with *Ruppia maritima*, a species that tolerates lower salinities and is often found in brackish water. Around the Bay of Many Islands, salinity was around 18 ppt and eelgrass was dense with many flowering shoots. Northward from the Bay of Many Islands, eelgrass was found in many shallow nearshore areas extending to the Roggan River. Eelgrass beds were generally dense and flowering and appeared healthy (Short 2008).

With respect to stations south of the La Grande River, salinity at Tees Bay measured 19 ppt and eelgrass was sparse. However, it was abundant in Dead Duck Bay and south to beyond the Castor River, where salinities were generally around 20 ppt (Short 2008; see Figure 8).

Status of eelgrass beds according to Aboriginal traditional knowledge

During the environmental assessment of the Eastmain-1-A and Rupert diversion project, the Cree community reported that there had been no signs of eelgrass recovery since its decline in 1998 (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006).

A member of the Cree community of Chisasibi who appeared before the Standing Committee on Fisheries and Oceans in March 2008 stated that eelgrass had disappeared following the start-up of the La Grande complex, then reappeared in 1995 and subsequently went into decline (FOPO 2008).

Short's (2008) first-hand accounts gathered from Aboriginal interviewees indicate that before the construction of the hydroelectric complex, eelgrass was extensive and healthy even near the mouth of the La Grande River where it empties into James Bay. In the early 1980s, the Cree observed that eelgrass was sparse, particularly north of the La Grande River, and less abundant than previously. Some eelgrass beds had disappeared, while others had become less dense (Short 2008).

In 2009, the few Cree users of James Bay interviewed by Hydro-Québec felt that eelgrass abundance had declined along the coast. Although it had recovered at some locations, it remained less dense and occurred to a larger extent in deep bays and less often in shallow bays than previously (GENIVAR 2010). In 2011, the Cree interviewed by Hydro-Québec were of the opinion that the eelgrass beds were less abundant in 2011 than they had been over the

previous four decades. In their view, the species had only partially recovered since the 1998–1999 decline (Consortium Waska-GENIVAR 2011).

On February 25 and 26, 2013, three meetings were held with some 15 representatives of the Cree Nation of Chisasibi (Interview, Chisasibi Cree community, 2013). The members of the group provided an oral presentation of some of their traditional knowledge regarding eelgrass beds on the east coast of James Bay. The participants stated several times that the eelgrass decline began in 1975, at the time the La Grande complex began operations. According to their accounts, there used to be extensive eelgrass beds as far as Pointe Louis-XIV. One of the Cree participants recounted that when he used to go fishing for Greenland cod at Comb Island with his grandfather, there were eelgrass beds up to 30 feet deep. After the construction of the La Grande complex, large clumps of eelgrass were found “stranded” on the shore. At least two Cree hunters present at the meeting—one who hunted in the Pointe Louis-XIV sector north of the La Grande River, and the other who hunted in a sector south of the river—had observed this phenomenon but never prior to the construction of the La Grande complex. The representatives of Chisasibi who were interviewed had also observed that the eelgrass leaves were shorter and narrower than before, and were found in small patches. The rhizomes seemed dead. Pieces of rhizomes freed from the ice during the spring thaw were observed. Sparse patches of eelgrass are now found scattered north and south of the mouth of the La Grande River. In recent years, the beds have been stable, with no deterioration or improvement. Eelgrass still occurs in the Bay of Many Islands sampling area, but its abundance is only 50% of what it was prior to the beginning of operations at the La Grande complex (Interview, Cree community of Chisasibi, 2013).

2.3 Potential causes of eelgrass decline on the northeast coast of James Bay

In 1998 and 1999, members of the Cree community of Chisasibi reported a marked eelgrass decline along the northeast coast of James Bay. Monitoring conducted by Hydro-Québec in 1999 confirmed a significant decline in the distribution and abundance of eelgrass from Rupert Bay to Pointe Louis-XIV, and even in the area of Manitounuk Sound, along the east coast of Hudson Bay (Lemieux et al. 1999).

Chisasibi Cree community members state that beds are currently less abundant and less healthy than in the 1970s, 1980s and 1990s. They feel that the species has only partially recovered since the decline of 1998–1999 (Consortium Waska-GENIVAR 2011). In their view, the eelgrass beds located north of the mouth of the La Grande River had suffered a more significant decline than those south of the mouth (Short 2008) and would recover more slowly (Consortium Waska-GENIVAR 2011).

Reasons for the decline

Four main hypotheses were put forward to explain the eelgrass decline in James Bay: wasting disease, climate change and weather conditions, isostatic uplift, and changes to the hydrologic regime at the mouth of the La Grande River, including reduced salinity and increased turbidity of coastal waters located near the river’s mouth. It was also suggested that the eelgrass decline on the northeast coast of James Bay was caused by the combination of temperature changes, a

reduction in water clarity in the plume of the La Grande River, a reduction in salinity, and the cumulative impacts of these factors (Short 2008).

One of the experts consulted during the environmental impact assessment of the Eastmain-1-A and Rupert diversion project noted that changes had been observed in the salinity, temperature and ice conditions in the sector of the La Grande River plume over the past 30 years. Isostatic uplift and climate variations that occurred during the same period could also have altered the eelgrass environment. He felt that comparative studies should be conducted in sectors that had not undergone these changes to determine the relative significance of each of the factors. He was of the view that the studies conducted to that point did not make it possible to differentiate the effects of interannual climate variability from those of variations in river discharge caused by hydroelectric developments. This expert also noted that the available data were insufficient to properly monitor the changes in the coastal environment of James Bay (Ingram 2006).

Wasting disease

Given the speed and extent of the eelgrass decline, some scientists believe that it was triggered by the wasting disease caused by *Labyrinthula zosterae*, a marine slime mould-like protist. This organism is endemic in eelgrass populations and can almost always be found growing to some degree within healthy eelgrass beds. It occurs on the leaves and can spread from plant to plant by direct contact (Short 2008). It occurs as small brown spots on the leaves that develop into black patches. The plant eventually dies, and the leaves break off in large numbers. Photosynthesis slows down, and the bed is ultimately ravaged after several weeks (den Hartog 1970). The presence of signs of the disease on eelgrass plants was not verified at the time of the decline of 1998–1999 (Lemieux et al. 1999).

Wasting disease outbreaks rarely occur at salinities of ≤ 10 ppt (Short et al. 1988; Muehlstein et al. 1988; FOPO 2008) because the protist in question is unable to tolerate low salinities (Short et al. 1988). Outbreaks occur in higher-salinity areas, whereas in low-salinity areas of eelgrass habitat, the plants are unaffected by the disease and survive to repopulate the more saline areas after an episode of the disease subsides (Short 2008). According to Short (2008), the salinities measured in the La Grande River plume could not trigger a major outbreak of wasting disease. A salinity of at least 20 ppt would be required for such a situation to occur. The salinity of the eelgrass beds in 1998 is not available (Short 2008).

Some scientists believe that invasion by *Labyrinthula zosterae* may have been triggered by unusually high summer and winter temperatures, which may have reduced the plants' resistance to the micro-organism. Summer temperatures recorded at the La Grande airport were particularly high in 1998 (Lemieux et al. 1999). On the other hand, the colder waters of the La Grande River that flow into James Bay would have not been very conducive to the growth of the micro-organism. Hydro-Québec notes that the waters surrounding the eelgrass beds at the Tees Bay station, which is 10 km south of the mouth of the La Grande River, are colder and have a salinity of < 10 ppt, which could potentially explain why it was the only station immune to the disease (Lemieux et al. 1999). However, temperature is not the main factor in the development of infection by *Labyrinthula zosterae* (Short et al. 1988).

In the 1930s, 90% of the eelgrass beds along the Atlantic coast of North America and Europe were decimated by wasting disease; they took three decades to recover (Short et al. 1988; DFO 2009). In the early 1980s, wasting disease was detected along the northeast coast of the United

States. There were very localized reports of declines associated with this disease. Infected plants were identified in Europe, but no decline was reported (Short et al. 1988). *Labyrinthula zosterae* does not appear to have contributed to the eelgrass declines currently observed in eastern Canada (Hanson 2004).

Climate change and meteorological conditions

One of the hypotheses put forward to explain the eelgrass decline in James Bay is climate change, which can manifest itself in various ways, including increases in water surface temperature, temperature variations, sea level, and frequency and intensity of storms (Orth et al. 2006), as well as increases in water CO₂ concentrations and UVB radiation levels (Neckles et al. 2009).

The increase in global temperature can alter growth rates and other physiological functions of seagrasses, leading to a shift in their distribution and to increased eutrophication, which can result in plant community changes (Short and Neckles 1999).

The rise in sea level is associated with an increase in water depths and a higher probability of seawater intrusion into estuaries or freshwater bodies, which could also cause considerable changes in the distribution of plant communities. Increased water depth, which reduces the amount of light reaching eelgrass beds, could directly reduce plant productivity where plants are light-limited. In addition, variations in water motion and tidal circulation could decrease the amount of light reaching the plants by increasing turbidity or stimulating the growth of epiphytes (Short and Neckles 1999). Isostatic uplift and the rate of sediment deposition on the sea bottom can offset the impact of sea level rise on eelgrass beds by influencing the water level (Shaughnessy et al. 2012).

Increases in CO₂ in coastal waters can promote the growth of seagrasses in regions where carbon is limited, but they can also promote the growth of algae. Exposure of plants to UVB radiation may inhibit photosynthetic activity in certain species (Short and Neckles 1999).

A recent study based on a hydrodynamic model tested at Hog Island Bay, in Virginia, indicates that eelgrass is likely to tolerate a rise in sea level but not an increase in the frequency of days when summer water temperature exceeds 30°C. Summer high-temperature events can cause rapid eelgrass losses, regardless of water depth (Carr et al. 2012).

Long-term studies carried out in the Pacific Northwest during a time of climatic variation (El Niño and Pacific decadal oscillation) revealed that small variations in temperature and mean sea level were correlated with substantial changes in eelgrass density, growth, biomass and areal extent, indicating that climate change can have an impact on eelgrass. Following an El Niño event in 1997–1998, eelgrass beds along the U.S. Pacific coast recovered in two to four years, which suggests that eelgrass can be resilient to strong climatic variation (Neckles et al. 2009).

According to Ingram, one of the experts consulted in the federal assessment of the Eastmain-Rupert diversion project, the most obvious climate changes that occurred in the marine environment of James Bay and Hudson Bay are related to air temperature and to the thickness, extent and duration of sea ice cover resulting from the general warming observed in recent years. However, studies are needed to determine whether these observed changes in

sea ice conditions are attributable to hydroelectric development, climate change or interannual atmospheric variability (Ingram 2006).

In the 1990s, James Bay Cree observed that fall temperatures quickly moved from one extreme to the other. The cold weather arrived earlier, but the lakes froze later and snowfall increased. In the areas located near the reservoirs, the winters became colder (McDonald et al. 1997). In interviews with representatives of the Cree community of Chisasibi in February 2013, participants stated that they felt that the warming of the continent had increased the temperature of the La Grande River and thus the waters in which eelgrass grows.

Some scientists reject the idea that eelgrass decline in James Bay is attributable to climate change because they feel the decline in James Bay was too rapid to be explained primarily by climate change. Although it is recognized that climate effects are responsible for year-to-year variations in eelgrass biomass, climate induces small changes. If major temperature or sea level changes had occurred in James Bay, they would have been noticed (Short 2008).

Meteorological conditions in 1998–1999

The summer of 1998, i.e., the period during which there was a significant widespread eelgrass decline in James Bay, was characterized by unusually high temperatures and very low winds. The recorded wind speed from April to June at the La Grande airport was particularly low compared to previous years, i.e., 1988 to 1999 (analysis of La Grande airport data by Lemieux et al. 1999). In 1998, eelgrass beds may have been disturbed by a reduction in water mixing due to the low winds and high temperatures.

In the winter of 1997–1998, the water temperatures were exceptionally warm (0.9°C at Robert-Bourassa and 0.5°C at the outlet of the La Grande-1 plant). Water temperatures downstream from La Grande-1 are typically between 0 and 0.15°C. The ice cover melted in March, despite the low turbine flows (Saucet 2002).

Isostatic uplift

Isostatic uplift is the gradual rising or “bouncing back” of the Earth’s crust from the weight of the glaciers that once covered the continent. Isostatic uplift affects the entire region of the east coast, which encompasses James Bay and Hudson Bay. It occurs at a rate of 1.0 to 1.5 cm per year on the east coast of James Bay (Tushingham 1992) and 0.5 to 1 cm per year in the sector of the La Grande River (Hydro-Québec 2004). The average water level appears to be continuously declining relative to the continent (Tushingham 1992; Lalumière and Lemieux 2002). Isostatic uplift can cause stress to eelgrass in shallow areas by exposing it to ambient air more frequently, causing it to dry out (Short 2008).

Exploratory studies conducted in 1982 on the east coast of James Bay revealed that isostatic uplift could play an important role in changes to the shores of the east coast of James Bay. The topography of the various stations modulates the response of eelgrass to isostatic uplift. The impact of isostatic uplift on eelgrass therefore depends on the slope profile, local hydrodynamics and the degree of protection against wave, wind and ice action (Lalumière and Lemieux 2002).

Uplift of approximately 11.7 cm occurred between 1969 and 1982, which would have allowed for the westward expansion of vegetation by about 150 m (Roche Itée 1982). An analysis of the

water levels at the Churchill tide gauge station in Hudson Bay (Ancil 1999, from Lemieux et al. 1999) indicated an overall drop in the water level between 1962 and 1998, which was attributed to continental uplift. The mean tidal range was at the peak of a cycle between 1995 and 1999. Tidal range is the difference between high tide and low tide; it has a cycle of approximately 19 years (Ancil 1999). Some scientists believe that the combination of a high tidal range and a record low water level could have promoted hydrodynamic conditions that were unfavourable to eelgrass growth (Lemieux et al. 1999). The average water level at the Churchill tide gauge station in Manitoba appeared to have been stable from 1992 to 2003 (Lemieux and Lalumière 2004). However, a reanalysis of the level of isostatic uplift recorded at Churchill indicated a mean rate of topographic height change of 11.4 mm/year between 1996 and 2001, and a mean relative sea-level height change of -9.72 mm/year between 1940 and 2001 (Wolf et al. 2006). Eelgrass monitoring by Hydro-Québec indicated a decline in eelgrass abundance at shallow depths (0.5 m) and the loss or decline of eelgrass beds in certain very gently sloped bays (Lemieux and Lalumière 2002). In addition, a study conducted by Hydro-Québec between 1991 and 2000 to verify the presence of short-term changes on vegetation caused by isostatic uplift revealed the presence of significant and rapid expansion of vegetation towards the sea and changes in the species colonizing the marsh. Some of the changes observed could be attributed to isostatic uplift (Lemieux et al. 1999; Lalumière and Lemieux 2002; HQ and GENIVAR 2005).

Some scientists reject isostatic uplift as the cause of eelgrass decline in James Bay because they believe that the decline was too sudden to be explained primarily by such a gradual long-term geological process. While it is true that isostatic uplift can cause eelgrass at the most shallow edge of its distribution to be stressed, scientists believe that the losses of eelgrass observed at depths of 1 and 2 m cannot be attributed to the very small decreases in depth that occur as a result of isostatic uplift since the eelgrass habitat at these depths is still well within the range at which the plants are able to thrive (Short 2008).

Changes to the hydrologic regime of coastal waters

The reduced salinity and increased turbidity of the coastal waters caused by changes in freshwater discharges from the La Grande River are the two primary changes in the hydrologic regime that were identified as being possible causes for the eelgrass decline in James Bay.

According to the experts of NTK and Makivik Corporation, seasonal changes in the hydrologic regimes of the La Grande and Nelson rivers (which contribute 50 to 60% of the discharge of the tributaries of James Bay and Hudson Bay from December to April) upset the marine ecosystems of Hudson Bay (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006). The report of the federal review panel for the Eastmain-1-A and Rupert diversion project indicates that Fisheries and Oceans Canada believes that better knowledge of the factors governing the physical and biological oceanography of James and Hudson bays is needed to assess the impact of the changes in the flow of the La Grande River on the marine ecosystem and the resources of these bays. The review panel determined that there was no evidence to rule out the possibility that the operation of the La Grande complex had an impact on eelgrass beds on the east coast of James Bay (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006).

Reduction in salinity

One possible explanation for the eelgrass decline is the reduction in coastal salinity caused by the increased levels and frequency of freshwater discharge to James Bay from the La Grande River.

A study to monitor the salinity of the coastal waters was conducted in 2006–2007 along the coast of James Bay north of the mouth of the La Grande River. The results suggest the existence of a salinity gradient in the bay resulting from the dispersion of the freshwater plume of the La Grande River (Short 2008). A previous study carried out for JBEC in 1986 concluded that eelgrass beds along the east coast of James Bay were not distributed along a salinity gradient (Lalumière 1986).

Although the long-term effect of reduced salinity on eelgrass growth and survival has not been studied (Short 2008), the literature (Biebl and McRoy 1971; Pinnerup 1980; Burdick et al. 1993; Baden and Bostrom 2001; Thom et al. 2003; Thorhaug et al. 2006) suggests that extended periods of exposure to seawater dilution during the growing season result in reduced eelgrass growth and abundance (Short 2008). The application of different salinity treatments to eelgrass in the laboratory showed that salinity below 5 ppt reduced leaf growth and biomass and that a salinity of 0 ppt was lethal to eelgrass after exposure of two months (Short 2008). A slowdown in growth also means there is a smaller amount of energy that can be invested in the production of belowground biomass, i.e., in the energy reserves that enable eelgrass to survive the winter (Short 2008).

The increase in discharge levels from the La Grande River and seasonal fluctuations in the timing of discharges can reduce the salinity of the coastal waters surrounding the eelgrass beds and therefore disrupt eelgrass growth. In 1998–1999, significant changes were made to discharges from the La Grande River. Prior to 1998, the highest discharges were in the winter. Between October 1998 and September 1999, winter discharge levels decreased, while summer discharge levels increased by over 1000 m³/s from the previous year. In May 1999, the water level reached 2 metres, which was unusually high, the norm being 1.5 metres in summer (Saucet 2002). The water levels reflect the variations in discharge. In 2000, summer discharges remained high (Short 2008). Increased freshwater inflows occurred at the start of the eelgrass growing season, which may have affected the health and distribution of beds at a critical point in their development (Short 2008). According to certain scientists, all of these changes could have triggered the massive decline in eelgrass beds along the James Bay coast in 1998–1999 (Short 2008).

The increased discharge of the La Grande River can also lead to reduced eelgrass growth under the ice due to reduced salinities at the ice/water interface within the freshwater plume, over larger areas (Ingram 2006). In addition, the NTK experts believe that the reduced salinity of coastal waters could have an impact on the duration of winter ice cover in the bay. Prolonged ice cover means that eelgrass is trapped under the ice for a longer period of time, which reduces its photosynthetic activity and, as a result, its growth. To survive year-round, eelgrass plants must grow rapidly and fully utilize the summer light in June and July for photosynthesis (Short 2008). Another expert (Gratton 2006) consulted for the federal assessment of the Eastmain-1-A and Rupert diversion project is of the view that climate variability is the most important factor in ice cover variation, a conclusion supported by Gagnon and Gough (2005). However, Gratton believes, as does a second expert consulted on the matter (Ingram 2006),

that it is important to investigate the relationship between variations in ice cover and variations in discharge and climate in order to better assess the role of each factor on the duration of ice cover. Ingram also believes that comparative studies carried out in regions not influenced by changes in discharge could provide a better understanding of the factors responsible for the decline (Ingram 2006).

Increased turbidity

One possible explanation for the eelgrass decline in James Bay is increased turbidity caused by an increase in suspended sediment concentrations in the water column. An increase in suspended matter reduces water clarity. Clear water provides ideal conditions for eelgrass growth because more light reaches further into the water column. The two greatest threats to eelgrass throughout its range are nutrient pollution and suspended sediments, both of which reduce water clarity (Short 2008).

Increased freshwater discharges to James Bay can lead to more suspended sediments in coastal waters with eelgrass habitat (Short 2008). A study conducted at the mouth of the Great Whale River showed that concentrations of particulate organic matter were positively correlated with discharge and that river waters had lower nutrient concentrations than Hudson Bay coastal waters following spring ice breakup (Hudon et al. 1996). However, Short notes that an increase in suspended sediments is not obvious in the coastal waters of James Bay and that such an increase is usually associated with increases in industrial development and human population, neither of which has occurred at James Bay (Short 2008). According to Short, the information gathered to date is insufficient to determine whether there has been an increase in suspended sediments caused by erosion of the banks of the La Grande River or whether any such increase could reduce the productivity of eelgrass beds within the river plume (Short 2008). Hudon et al. (1996) also indicate that rivers flowing through tundra to Hudson Bay and Ungava Bay normally carry a specific dissolved organic carbon load that is half that of rivers flowing through forested basins to the Gulf of St. Lawrence. However, the environmental impact study of the Eastmain-1-A and Rupert diversion project reports that bank erosion is already very active on the La Grande River downstream from the La Grande-1 power plant and is a major concern in the area. Hydro-Québec planned to install granular blankets in certain sections of the river to reduce the sediment load, particularly off Chisasibi. An erosion monitoring program on the La Grande River downstream from the La Grande-1 complex was also planned (Hydro-Québec 2004).

Representatives of the Cree Nation of Chisasibi noted a decline in the water clarity of the La Grande River. They attribute the increased turbidity to erosion and landslides along the river, which result in the presence of suspended matter, particularly in winter. Such situations had not previously been observed (FOPO 2008). In interviews conducted with the Cree community of Chisasibi on February 25 and 26, 2013, the participants once again expressed their concerns about the water quality of the La Grande River, which they described as very murky. In their view, water turbidity is a key cause of eelgrass decline in James Bay. They believe that the turbidity could be caused by the accumulation of mud from the reservoirs (Interview, Communication with the Cree of Chisasibi, 2013).

2.4 Status of eelgrass beds on the Atlantic coast and causes of observed declines

Eelgrass is in decline in several parts of eastern Canada and the northeastern United States. Declines are most frequently reported in the southern regions and are often related to water quality degradation. Eelgrass decline in more northern areas is generally less dramatic (Neckles et al. 2009).

Eelgrass decline has been largely attributed to excessive nitrogen enrichment, which stimulates blooms of phytoplankton and filamentous algae, such as *Ulva* spp. and *Enteromorpha* spp., which have the potential to displace seagrasses as the dominant plants in eutrophic waters (Ward et al. 2005). On the U.S. east coast, considerable efforts have been made to establish nitrogen concentration thresholds above which eelgrass survival is at risk. These thresholds are estuary-specific. For estuaries south of New England, 50% eelgrass loss is anticipated at loads above a mean threshold of 50 kg N/ha/year, and eelgrass loss of over 75% is anticipated at loads above 100 kg N/ha/year (Neckles et al. 2009). On the U.S. east coast, there are several examples of eelgrass recovery following water quality improvements, such as in Tampa Bay, Florida, and in three locations in Massachusetts: Boston Harbor, New Bedford Harbor and Gloucester Harbor (Neckles et al. 2009).

There has been a widespread decline in eelgrass distribution and abundance in the Maritime Provinces. No one single factor has been identified as the cause of the regional decline (Hanson 2004). Invasion by green crab, water quality degradation and aquaculture are the main possible causes for decline in eelgrass beds in several parts of New Brunswick and Nova Scotia. The most dramatic eelgrass decline observed in the Maritime Provinces was the 95% decline in Antigonish Harbour, Nova Scotia, between 2000 and 2001. It was attributed to the presence of green crab (Seymour et al. 2002, from Hanson 2004). Eelgrass declines documented in several estuaries of Prince Edward Island are primarily attributable to a reduction in water quality (warmer summer water temperatures, nitrogen input from agricultural runoff, and sediment input from erosion) and in some cases to invasion by green crab, which could also affect the stability of eelgrass beds in Newfoundland, where green crab has recently been detected (Neckles et al. 2009). Eelgrass beds in Newfoundland and Labrador have not suffered major declines (Hanson 2004). In Quebec, apart from James Bay, eelgrass abundance in the estuary and Gulf of St. Lawrence has been stable or increasing over the past 20 years (Neckles et al. 2009). During this period, there was a marked increase in eelgrass distribution in the Manicouagan sector, where new beds even became established (Martel et al. 2009).

In eastern Canada, there are insufficient data to properly assess changes in the health of eelgrass beds in recent decades (Hanson 2004). The Oceans and Coastal Management Division of Fisheries and Oceans Canada, Quebec Region, established the Eelgrass Monitoring Network (EMN) in conjunction with outside partners in 2005. The network covers only eelgrass habitats in coastal regions of the St. Lawrence. The Community Aquatic Monitoring Program (CAMP), a community network in the southern Gulf of St. Lawrence supported by Fisheries and Oceans Canada (Gulf Region), was officially launched in 2004. It is focused on the estuaries of the Maritimes (Nellis et al. 2012). There are also two international eelgrass monitoring networks: SeagrassNet (global membership) and Seagrass-Watch (members located around the Indian Ocean).

Given the continuous decline in eelgrass beds in eastern Canada, it is becoming increasingly important to identify the causes of the decline and their impacts on ecosystems (Hanson 2004).

According to several experts, protecting existing eelgrass beds and improving environmental conditions to allow for their restoration and recovery is becoming increasingly critical (Neckles et al. 2009).

3. Abundance and distribution of waterfowl and other waterbirds

In recent years, the Cree community living on the east coast of James Bay has noticed a decline in the abundance of migratory geese and waterbirds along the coast (McDonald et al. 1997; FOPO 2008; Consortium Waska-GENIVAR 2011). The available information on the observed changes in waterfowl and sea duck populations in James Bay comes primarily from first-hand accounts gathered from the Cree community.

Waterbirds

Prior to the observed ecosystem changes in James Bay, the Red-throated Loon was much sought after by hunters for food (Interview, Cree community of Chisasibi, 2013). Today, it is no longer hunted at the Cree community of Chisasibi because it is no longer seen. According to members of the community, the species no longer uses the same sectors due to the changes that have occurred in the La Grande River. The members interviewed stated that this species tended to follow the ice, which now melts too quickly. The fact that the ice is gone from the river in late April explains why this species no longer stops there (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006). Red-throated Loons used to arrive in James Bay at the end of May, after migrating geese. They followed the Arctic Terns, which also migrated north at the same time (McDonald et al. 1997). The representatives of the Cree community also reported that they no longer saw Arctic Tern due to the loss of eelgrass beds, which provided habitat for the fish species that were a food source for the tern (Interview, Cree community of Chisasibi, 2013). Surf Scoters also used to use the area, but this is no longer the case (Interview, Cree community of Chisasibi, 2013).

Canada Goose

According to Curtis and Allen (1976), the Canada Goose was the most abundant species of goose along the Quebec coast of James Bay in the early 1970s. Representatives of the Cree community who were interviewed over the years believe that the migration corridor used by Canada Geese of the Atlantic population (short necks) has shifted to the east (McDonald et al. 1997; Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006; Interview, Cree community of Chisasibi, 2013). However, that hypothesis has not been studied (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006). Some members of the Cree community believe that geese populations have declined along the coastline since 1988 (McDonald et al. 1997). Others observed a three-year decline following the construction of the hydroelectric project in 1975 (Interview, Cree community of Chisasibi, 2013). It is believed that the geese are now taking a more inland corridor and that some use the reservoirs of the hydroelectric complex (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006). They are observed at Brisay, in the vicinity of the Caniapiscau reservoir (Interview, Cree community of Chisasibi, 2013). Moreover, Hydro-Québec had observed an

increase in the number of geese in the sector of the reservoirs (Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006). Some geese still fly along the coast, but without stopping, and they tend to fly off the coast more than in the past (McDonald et al. 1997). The Cree of Chisasibi indicated that they sometimes saw these geese in the spring flying along a north-south trajectory. It is believed that these individuals arrived at their breeding grounds too early and were backtracking to feed further south (Interview, Cree community of Chisasibi, 2013). One member of the community stated that the last good goose hunt occurred approximately four years ago. When the geese stopped flying over Chisasibi, they were replaced by the resident goose population (long neck: Interview, Cree community of Chisasibi, 2013), which undertake a later moult migration, i.e., in June (McDonald et al. 1997; Interview, Cree community of Chisasibi, 2013). They arrive in large numbers along the coasts, at the mouth of rivers and on islands in the bay (McDonald et al. 1997). It is believed to be the only Canada Goose population that now passes over Chisasibi. They were not observed in the Chisasibi sector when eelgrass was abundant. These geese are hunted despite the fact that the Cree do not find them to be as flavourful (Interview, Cree community of Chisasibi, 2013). The community has also noted that the hunting seasons have become shorter. In the fall, the geese leave for the south earlier (McDonald et al. 1997; Interview, Cree community of Chisasibi, 2013). Movements of Canada Geese (short necks) are still observed in the Chisasibi sector, but only adults and in smaller numbers (Interview, Cree community of Chisasibi, 2013).

In Nova Scotia, there is a case in which eelgrass loss has resulted in a decline in the number of migrating Canada Geese in the province. During the fall migration, a large population of Canada Geese used to migrate over northern Nova Scotia and spent the winter in the southern part of the province. These geese fed heavily in the eelgrass beds. In 2001, there was a significant decline in eelgrass in Antigonish Harbour attributed to invading green crab in the estuary. It was accompanied by a 50% reduction in the number of geese normally present during the fall migration. This decline also occurred on the wintering grounds for several winters. However, the reduction in the migrating population is probably the result of a shorter time spent in the estuary, rather than a decline in the total number of migrating birds (Seymour et al. 2002).

Brant

Of the geese species that occur in James Bay, Brant is most closely tied to fluctuations in eelgrass abundance and distribution. Brant were known to use the east coast of James Bay during spring and fall migration in order to feed on eelgrass, their preferred food (Reed et al. 1998). Today, the decline in the Brant population on the east coast of James Bay may be the result of the eelgrass decline (Interview, Cree community of Chisasibi, 2013). No studies have been conducted to determine whether the eelgrass decline that occurred in 1998–1999 along the east coast of James Bay led to a reduction in Brant populations. Moreover, no long-term monitoring of the state of the Brant population that uses James Bay has been carried out. The members of the Cree community of Chisasibi interviewed by Hydro-Québec in October 1999 stated that Brant had stopped only for a short time to feed along the east coast of James Bay in the fall. They remained for the night and then left. They appeared to be in poor physical condition and the migration corridors appeared to have changed, with the birds taking a more inland route (Lemieux et al. 1999).

The North American population is divided into four subpopulations. Brant observed in James Bay are part of the Atlantic population (Figure 11). In Quebec, the species is considered a common migrant but a rare breeder. It breeds in the Lower and Middle Arctic, particularly on

Baffin and Southampton islands, as well as on the coasts of Foxe Basin, where it begins laying eggs in mid-June. It winters on the east coast of the Atlantic, from Massachusetts to North Carolina (Reed 1995, 1998).

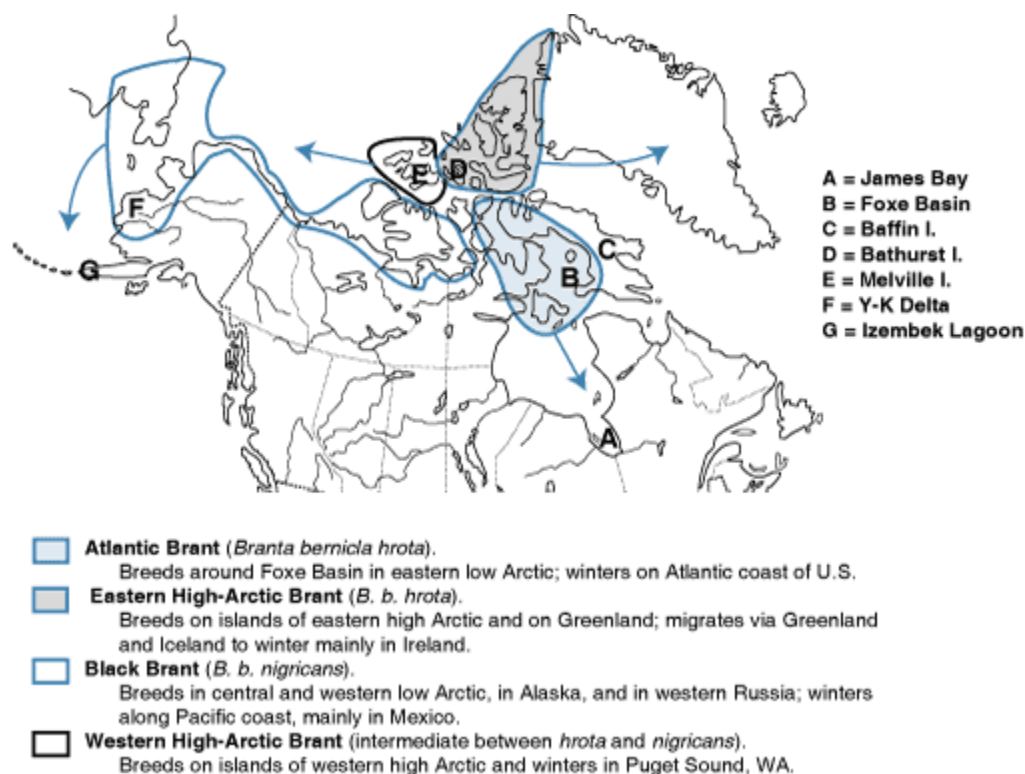


Figure 11: Distribution of the four subpopulations of Brant in North America (from Reed et al. 1998: <http://bna.birds.cornell.edu/bna/species/337>)

Atlantic population

Given the close link between eelgrass and Brant, and the decline of both in James Bay, the changes in the diet of Brant and an overview of available information on its population trends in recent decades are described in the sections below.

Diet

Eelgrass is the preferred food of Brant during the non-breeding season (Reed et al. 1998). Its populations are therefore largely dictated by eelgrass distribution and abundance (Ward et al. 2005). Brant can also feed on other species of grass (particularly *Ruppia maritima*), green algae (particularly the sea lettuces *Ulva lactuca* and *Enteromorpha*) and certain plant species that grow in saltmarshes (particularly *Spartina alterniflora*). Green algae dominate the diet of wintering Atlantic Brant (Ward et al. 2005). On breeding and moulting sites, these birds feed on grasses, forbs and mosses (Reed et al. 1998).

At James Bay

Body condition of females prior to their arrival at the breeding sites has an effect on their reproductive success. Body reserves can provide 11 and 22% of the energy requirements of the males and females, respectively, during the incubation period (Ankney 1984, from Reed et al. 1998). The energy reserves accumulated by Brant before arriving at their breeding grounds, particularly along James Bay, are therefore critical. A study has shown that a number of Brant relied on the food resources at James Bay to build the necessary reserves to complete their spring migration and to ensure good reproductive success (Vangilder et al. 1986, from Benoit et al. 1996). Eelgrass does not contain a high level of protein, but it provides a sufficient amount of energy to meet the birds' requirements during migration. Plants located further north, such as at James Bay (40–50°N), have a higher energy content than those in the south (20–30°N). It is possible that Brant prefer to feed in eelgrass beds not for their nutritional value, but because they are widely available (Reed et al. 1998).

A study conducted in 1990–1991 on the northeast coast of James Bay showed that Brant used eelgrass beds exclusively or almost exclusively during the spring (99.8% of the dry weight of all food ingested) and fall (95% of the dry weight of all food ingested) migration. In the aerial survey of June 6, 1990, in the Bay of Many Islands, over 95% of the 150 Brant observed were in the eelgrass bed–tidal flat microhabitat, with the remainder being in adjacent areas of open water (Reed et al. 1996). Another series of surveys conducted in the spring of 1994–1995 confirmed that eelgrass beds were the primary feeding habitat of Brant. Open-water areas and the shores of islands and the mainland were used primarily for resting (Benoit et al. 1996).

In its entire range

Prior to the outbreak of the wasting disease epidemic that devastated eelgrass beds in the 1930s, Atlantic Brant fed primarily on eelgrass (85%). Following this major event, their diet consisted primarily of sea lettuce (75%) and only 9% eelgrass (Cottam et al. 1944, from Ward et al. 2005). This change in diet was noticed by hunters, who complained that the flavour of the meat was negatively affected (Hanson 2004).

In the 1970s, Atlantic Brant began to make heavier use of inland areas, feeding on cultivated grasslands and on clover on golf courses in winter. However, this behaviour is displayed by only 5% of the wintering Atlantic Brant and is generally confined to sites near the coast (Ward et al. 2005). In contrast to most geese species, Brant still rely on marine habitats and native intertidal plants during the non-breeding season, particularly eelgrass and sea lettuce (*Ulva lactuca*), which has become an important food source on migration and wintering grounds, particularly in New York and New Jersey. The increase in abundance and availability of this macroalga, and the corresponding degradation and loss of eelgrass habitats along the Atlantic coast of the United States, resulted in a new dependence of Atlantic Brant on this alga (Reed et al. 1998; Ward et al. 2005).

Of the four Brant populations, Atlantic and Eastern High Arctic Brant have experienced the greatest degradation of their winter habitats and have also shown the most plasticity in feeding behaviour (Ward et al. 2005). Black and Western High Arctic Brant are the most dependent on eelgrass on their wintering grounds, probably because this plant species is two to three times more widespread on the Pacific coast than on the Atlantic coast of North America and Western Europe (Green and Short 2003).

Population status

Since there is no long-term monitoring of the Brant population that uses the east coast of James Bay, it is impossible to precisely assess the changes in the size of the population over the years. The available information comes from a number of isolated studies carried out at James Bay in recent decades, and from Aboriginal traditional knowledge.

However, long-term monitoring of the total Atlantic Brant population has been carried out, including individuals that are likely to use James Bay during migration. Since 1955, the U.S. Fish and Wildlife Service has conducted an annual assessment on Brant wintering grounds. Aerial and ground surveys are conducted every January. Since 1976, a productivity index based on the calculation of the juvenile/adult ratio is also produced in November. These surveys are not very precise because they typically consist of a single annual count and do not take account of climate variations or seasonal phenology (Reed et al. 1998). However, they are considered a good indicator of long-term fluctuations in the population. No survey of Brant productivity has been conducted on the breeding grounds (Castelli et al. in prep.).

History of the status of the population and use of the different habitats of the northeast coast of James Bay

Prior to 1930, the global Atlantic Brant population likely numbered about 300 000 individuals (Reed 1977). In the 1930s, wasting disease decimated 90% of the eelgrass populations along the Atlantic coast of North America and Europe. Recovery of eelgrass beds took three decades, and some areas where the species had been very dominant before the outbreak failed to recover (Short et al. 1988; DFO 2009). As a result, the Brant population declined considerably. Only 10% of the population survived the winter of 1933–1934. The Brant population increased concomitant with the gradual eelgrass recovery and a hunting moratorium between 1933 and 1952 (Ward et al. 2005). The Brant either moved north to areas where eelgrass was more abundant and/or switched to alternative foods (Cottam et al. 1944, from Ward et al. 2005; Reed et al. 1998). Although some birds may have starved, the greatest factor in the decline of the Atlantic population was the reduced breeding effort by malnourished birds (Kirby and Obrecht 1982, from Ward et al. 2005). By the mid-1950s, Atlantic Brant had reached the level present before the eelgrass die-off (Atlantic Flyway Council 2002, from Ward et al. 2005).

Poor reproductive success in 1971 and 1972, combined with overharvesting, led to a population decline over the winter of 1972–1973 (Lemieux et al. 1999). Aerial surveys were carried out on the Quebec coast of James Bay between 1972 and 1975 to obtain counts of waterfowl, including Brant. The survey produced highly variable data, depending on the year and sector in question. In the spring, Brant remained at James Bay for a period of two weeks between the second half of May and early June. Rupert Bay was the most heavily used site (2300 individuals). In the fall, Brant arrived in late August, and their numbers peaked in mid-October. Brant were observed along all sectors of the James Bay coast. The most heavily used sector was that of Chisasibi (14 000 individuals). It is estimated that 5000–15 000 Brant used Dead Duck Bay in the fall (Curtis and Allen 1976). Dense eelgrass beds around the Comb Islands and Black Whale Island attracted hundreds of Brant in the fall (Curtis and Allen 1976). The area located roughly between the Roggan River and Kakachischuan Point (encompassing the Bay of Many Islands, Kakassituq Point and Attikuan Point) was then considered an area of national importance owing to its vast eelgrass beds, which attracted 10 000–20 000 Brant each fall (Curtis and Allen 1976).

Curtis and Allen (1976) reported that Brant were not a particularly sought-after food species for the Cree in the early 1970s and that harvests of the species were variable. Native Harvest Research data from 1973 to 1975 estimated the Aboriginal harvest to be around 3000 birds in the spring and 5000 in the fall (Curtis and Allen 1976). However, Brant harvests in James Bay in the 1970s were larger in Quebec than in Ontario (NHRC 1982 and 1988, from Castelli et al. in prep.).

During the winters of 1976–1977 and 1977–1978, exceptionally cold weather gripped the east coast of the United States, covering intertidal areas in which eelgrass grows with thick ice (Ward et al. 2005). Surveys were carried out in the spring of 1977 to estimate losses. The surveys covered the Atlantic coast of the northeastern United States, the Maritime Provinces, the Gulf and estuary of the St. Lawrence, the north shore of Lake Ontario, and the James Bay and Hudson Bay coasts. A total of 35 313 Brant were observed in the surveys carried out in northeastern North America in May 1977. At James Bay, 3544 brant were counted, 1842 of which were on the Quebec side (Reed 1977). These results confirmed the existence of a decline in the Atlantic Brant population within a six-month period, in response to the exceptionally cold weather the previous winter. This event led to famine in the Brant population and caused a two-third decline in its numbers (Dupuis 1977; Reed 1995). Juveniles were more vulnerable to the lack of food than adults (Reed et al. 1998). In each of these winters, there was about a 30% decline in Brant use of estuaries in the north and a corresponding increase in more southerly wintering areas, where intertidal areas were ice-free. This was the first time Atlantic Brant were recorded feeding on cultivated grasses and golf courses (Hindman and Ferrigno 1990, from Ward et al. 2005). Reduced food resources in the 1970s, combined with hunting pressure and reduced recruitment, were likely responsible for a crash in the Atlantic population, similar to the 1930s crash. The population subsequently increased, from 42 000 birds in 1979 to 165 000 in 2003 (Ward et al. 2005).

Winter surveys by the U.S. Fish and Wildlife Service indicate that the population reached 185 000 individuals in 1991–1992 (Reed et al. 1998). However, the population fell to approximately 100 000 individuals in January 1993 as a result of low reproductive success in 1992 (Reed 1995).

The results of studies conducted on the northeast coast of James Bay in the spring of 1994 and 1995 were used to more clearly define the use of the shore of James Bay by Brant and to estimate the number of Brant that stopped there in the spring. Four coastal bays were visited, namely Dead Duck Bay, Aquatic Bay, Bay of Many Islands and Kakassituq Bay. On June 12, 1994, 507 Brant were counted along the shore between the Castor River and Pointe Louis-XIV. It was probably an exceptional year due to the abundance and persistence of the ice. Winter surveys conducted in November 1994 indicated that the population included a small proportion of juveniles, suggesting a poor breeding season in 1994 (Benoit et al. 1995). Globally, Brant primarily used the sector north of the La Grande River. The most heavily used sectors were the Bay of Many Islands (north sector) and Dead Duck Bay (south sector). These bays supported extensive areas of eelgrass beds and were partially free of ice. The characterization of Brant use of the coast of James Bay continued in 1995. Brant on spring migration made heavy use of the northeast coast. In surveys conducted on June 2, 7 and 8, it was estimated that over 12 000 birds were using the habitats of the coast. In the first survey, conducted on June 2, 1995, 4300 Brant were counted; 3410 were in the north sector and 839 in the south sector. Densities in the two sectors were comparable, namely 509 and 577 birds/100 km², respectively. In the second survey, conducted on June 7 and 8, 1995, larger

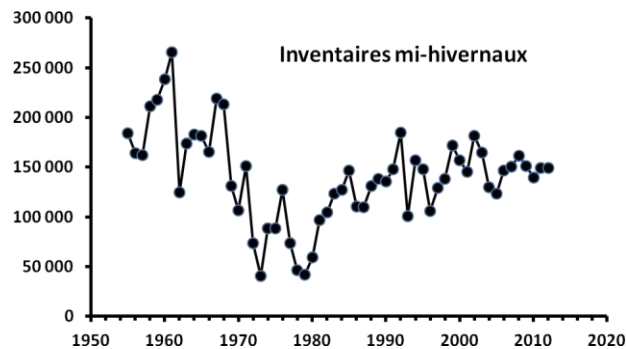
groups were counted. The populations had tripled, totalling over 12 500 individuals, i.e., 2650 in the south sector and 10 227 in the north sector. Brant had a clear preference for eelgrass beds (Benoit et al. 1996). At the time of the surveys, they were observed in other types of habitats, including open water areas, but only eelgrass beds were selected to any significant degree (Benoit et al. 1996). Aerial surveys conducted on June 4 and 6 in four coastal bays (Dead Duck Bay, Aquatuc Bay, Bay of Many Islands and Kakassituq Bay) revealed that Dead Duck Bay and Bay of Many Islands supported the largest numbers of Brant. On June 6, 1995, there was an average of 1187 individuals in each bay. Aquatuc Bay was by far the least heavily visited by Brant (Benoit et al. 1995–1996). The spring aerial surveys of 1994–1995 confirmed that the study sector between Walrus Point and Kakachischuan Point, which does not contain eelgrass beds, were not visited by Brant on spring migration (Benoit et al. 1996).

In 2005, a project was launched by the Canadian Wildlife Service, Quebec Region, as part of the Northern Ecosystem Initiative, to study species harvested by the Cree in the subsistence hunt. Hunters from the communities of Waskaganish, Wemindji and Mistissini were asked about the species harvested over two (Mistissini) or three years (Waskaganish and Wemindji). For each community, roughly one in three hunter-occupied households was interviewed about the species hunted by the members of the household over the course of the year. Brant accounted for less than 0.3% of the total number of birds harvested by the Cree of James Bay in 2006 and 2008. The most hunted species by far in each community and in each year surveyed was Canada Goose, which accounted for at least half the total take. In comparison, the number of Brant harvested in 2006 and 2008 was 239 and 49, respectively, for all communities interviewed, whereas the number of Canada Geese harvested for these two years was 49 961 and 48 293, respectively. The largest Brant harvest occurred at Wemindji, and the most Brant were harvested in 2005, with 148 birds out of a total harvest of 5 390 birds (CTA et al. 2009). According to surveys of the members of the Cree community of Wemindji, Brant are not a highly sought-after food species because they have been making the Cree sick for close to 20 years (Peloquin 2007; Sayles 2008).

Current population status

According to U.S. Fish and Wildlife Service data (Klimstra and Padding 2012), the Atlantic Brant population is relatively stable. In 2012, the total estimated population was 149 157 birds. In the past 10 years, the population has fluctuated between 123 000 and 181 000 birds. In 2011, the proportion of juveniles in the fall was 25% and the average number of juveniles per adult was 2.2 (see Figure 12).

A



B

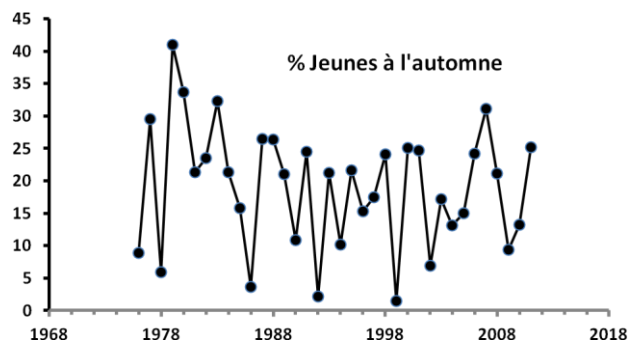


Figure 12: Atlantic Brant population trend in recent decades. A) Mid-winter Brant surveys conducted on the east coast of the United States. B) Proportion of juveniles relative to adults, counted during fall surveys on wintering grounds on the U.S. east coast (data from Klimstra and Padding 2012)

Translation key for Figure 12

Mid-winter surveys

% of juveniles in the fall

Although the overall Atlantic Brant population appears to be relatively stable, the James Bay Cree have observed a decline in recent years in the population that uses the east coast of James Bay.

In 2009, Cree hunters mentioned that Brant had become rare along the east coast of James Bay north of Rupert Bay in the spring (R. Cotter, pers. comm., from Castelli et al., in prep.). In interviews conducted in Chisasibi on February 25 and 26, 2013, the Cree complained about the significant decline in the number of waterfowl using James Bay during spring and fall migration since the construction of the La Grande hydroelectric project in 1975. With respect specifically to Brant, the Cree community members indicated that there were large numbers—as many Brant as Canada Geese—when eelgrass was abundant. They typically used the La Grande River sector, but since the construction of the La Grande complex, they have become rare at

Chisasibi. For example, a hunter related that four Brant were observed approximately 7 km north of Chisasibi a few years previously. It was the first time a young man from Chisasibi had ever seen Brant. In the Pointe Louis-XIV sector, a large group of Brant was observed in the spring roughly two years ago. The birds came from southwest of James Bay and continued their migration without stopping (Interview, Cree community of Chisasibi, 2013).

The number of Brant using the east coast of James Bay does not necessarily follow the same trend as the overall population counted in winter surveys. For example, a significant increase in the Canada Goose population was observed between 1996 and 2006, whereas a decline in catches was observed by Aboriginal hunters on the territory of Wemindji during the same period (Peloquin and Berkes 2009). The changes observed by Aboriginal hunters could be explained by changes in waterfowl behaviour or in hunting habits. The possibility that Brant have modified their migration route over the years in order to feed in other sectors following the eelgrass decline in James Bay cannot be ruled out. This possibility was clearly identified by the representatives of the Cree community, who believe that the geese have shifted their migration route eastward (McDonald et al. 1997; Federal Review Panel for the Eastmain-1-A and Rupert Diversion Project 2006; Interview, Cree community of Chisasibi, 2013). There are still gaps in the understanding of Brant migration routes.

Migration routes

A telemetry study was conducted in 2002–2003 to track Brant movements on their wintering, migration and breeding areas. Brant were captured along the northeast U.S. coast and fitted with either very high frequency (VHF) radio transmitters (59 birds) or platform transmitting terminal (PTT) transmitters (22 birds). The study results indicate that the Brant arrived at James Bay around May 25 and 28 and departed around June 12 and 15 (Castelli et al. in prep.). The end of the last week of June corresponded to peak spring migration on the northeast coast of James Bay (Benoit et al. 1996). During this study, the most heavily used sector in Quebec was Rupert Bay. The Brant tracked used the west coast of James Bay more heavily than the east coast (Castelli et al., in prep.). There are few eelgrass beds on the west coast of James Bay, but there are other feeding habitats, such as those used by Brant at their breeding grounds. In the telemetry studies, Brant were observed feeding in the salt marshes of Akimiski Island during spring migration (FOPO 2008; pers. comm. with Kenneth F. Abraham, 2013). Use of the east coast of James Bay by migrating Brant was well documented in the 1990s (Benoit et al. 1995, 1996; Reed et al. 1996).

Spring migration

The Atlantic Brant population on spring migration is more widely distributed in southern Quebec than in the fall (Reed 1995). There are two migration corridors: the eastern route (coastal route) and the western route.

The eastern route is northward along the New England coast through the Maritime Provinces, to the Gulf of St. Lawrence and the estuary, then likely past the Saguenay River before ending up in James Bay. The telemetry study conducted in 2002–2003 indicates that a minority of the birds tracked used the coastal route passing through the Maritime Provinces. With the extensive eelgrass declines that have occurred along the coastal route (Green and Short 2003), this route may have become less advantageous than the more direct inland route (western route), despite

the fact that the latter offers no coastal feeding areas before James Bay (Ward et al. 2005). However, Brant observations obtained during the Waterfowl Survey of the St. Lawrence Shoreline (WSHO) confirmed the presence of the species on the two shores of the lower estuary, in the Gulf and in the Gaspé Peninsula area. It was also observed on the Middle North Shore, in the areas of Sept-Îles and the Mingan Islands (Brousseau and Lepage 2013). The abundance of eelgrass along the estuary and Gulf of St. Lawrence partly explains why part of the population stops there in the spring.

The eastern route likely represents a shift in the route taken by Brant in the 1930s. That route consisted in a coastal route passing through the Maritime Provinces and then Sept-Îles Bay, which at the time was an important stopover due to its rich eelgrass beds. From that stopover, the Brant headed towards James Bay and Ungava Bay, before reaching their breeding grounds in Foxe Basin (Lewis 1937, from Castelli et al. in prep.). This northeastern route is no longer used (Castelli et al. in prep.), possibly because of the massive eelgrass decline along the coastal migration route and in the St. Lawrence estuary in the 1930s (Erskine 1988; Ward et al. 2005). However, it is interesting to note that birds have once again been observed in the vicinity of Sept-Îles (Reed et al. 1998 and Reed, pers. comm., 2012). Sampling work carried out in the Sept-Îles eelgrass bed between 2005 and 2010 indicate a decline in the number of flowering shoots relative to the number of vegetative shoots in recent years, which could suggest a reduction in stressors at this bed (Nellis et al. 2012).

The second corridor, the western corridor, originates in the states of New Jersey and New York, crosses inland to the confluence of the Ottawa and St. Lawrence rivers, crosses Lake Abitibi in some cases, and then heads directly for James Bay. As part of the telemetry study by Castelli et al. (in prep.), some birds stopped in the Lake Champlain, Montréal and Lake Saint-Pierre regions, and then headed over the Ottawa Valley to Abitibi-Témiscamingue and Radisson (Castelli et al., in prep.). Other birds made an almost non-stop flight between their wintering grounds and James Bay.

The telemetry study by Castelli et al. (in prep.) confirmed that the coast of James Bay is used during migration. Moreover, surveys conducted in Rupert Bay in May 2002 counted over 50 000 Brant (Tecsult Environnement inc. 2004). Owing to its geographic location, Rupert Bay is the first available feeding site for Brant after completing the first leg of their migration. From there, Brant moved to the northwest part of James Bay, and then headed north to their breeding grounds. The large majority of birds tracked in the telemetry study moved to the western shore of James Bay, where they remained until departing for their breeding grounds in the Foxe Basin region (Castelli et al. in prep.).

On June 7 and 8, 1995, the migrating Brant population moving along the northeast coast of James Bay was estimated at close to 12 700 birds, which is 8.5% of the total estimated population in January on the wintering grounds (Benoit et al. 1996, from Lemieux et al. 1999). However, in the 2002–2003 telemetry study, no use of the eastern shore of James Bay north of Rupert Bay was observed in the spring (Castelli et al. in prep.).

Fall migration

In the fall, Brant can spend up to a month along the coast of James Bay before continuing their migration to their wintering grounds. The west corridor is more heavily used in the fall. The species is uncommon in the St. Lawrence and eastern Quebec.

During the telemetry study by Castelli et al. (in prep.), the birds tracked ($n = 12$) during the fall migration arrived at Cape Henrietta Maria, the northernmost point on the west shore of James Bay, and then moved southeast to the area between Attawapiskat, Ontario, and Akimiski Island, Nunavut, before departing for the Atlantic Coast region. The birds that did not fly directly from James Bay to their wintering grounds made stopovers in Radisson, Abitibi-Témiscamingue, the Ottawa Valley and the Laurentians.

Observed changes in the Pacific Black Brant population

The distribution of Pacific Black Brant on their wintering grounds has shifted. Over the last 20 years, there has been a rise in sea-surface temperatures and sea level in the north Pacific. This has been associated with a decline in eelgrass abundance and distribution at the major Brant wintering areas along this coast. As a result, there has been a gradual shift in range of Black Brant wintering on the Pacific coast. There has been a decline in the number of Black Brant wintering in southern Mexico, where air and sea-surface temperatures already limit eelgrass growth (Ward et al. 2005), and an increase in Canada and the United States.

A significant increase in the number of Brant wintering in Alaska has also been observed, including a part of the Western High Arctic population, which has changed its traditional wintering grounds. The increased popularity of the wintering grounds in Alaska coincides with the warming trend in the North Pacific, which likely reduced the duration of ice cover along the coast of Alaska, giving Brant greater access to the eelgrass beds there. The number of Brant wintering in Alaska increased from fewer than 100 prior to the 1980s to 18 000 in 2001 (Ward et al. 2005).

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