Created and restored sedge marshes in the lower Fraser River and estuary: An evaluation of their functioning as fish habitat

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CREATED AND RESTORED SEDGE MARSHES IN THE LOWER FRASER RIVER AND ESTUARY: AN EVALUATION OF THEIR FUNCTIONING AS FISH HABITAT

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ABSTRACT

Levings, C.D. and D.J.H. Nishimura [ed.]. 1996. Created and restored sedge marshes in the lower Fraser River and estuary: An evaluation of their functioning as fish habitat. Can. Tech. Rep. Fish. Aquat. Sci. 2126: 143 p.

Ecological comparisons of transplanted, natural (reference) and disrupted (unvegetated) marsh sites on the lower Fraser River and estuary, British Columbia, were conducted between 1991 to 1993. The study examined vegetative biomass and cover, invertebrate abundance, fish abundance, fish residency and fish food, and submergence time for the three habitat types. Standing crop biomass at three transplant sites were within the range of values for reference sites but was much lower at an unstable site where sediment slumping had occurred. The percent cover of Lyngbyei's sedge (*Carex lyngbyei*) in eight transplant sites was <50% of that observed in adjacent reference sites when data were averaged over the study area; rushes (*Juncus* spp.) were more abundant in transplant sites. In all study reaches, abundance of invertebrates at transplant and reference sites was significantly higher than at disrupted sites. In several instances invertebrate abundance at transplant sites was greater than at reference sites. No significant difference ($p>0.05$) was observed among marsh sites when chum salmon (*Oncorhynchus keta*) and chinook salmon (*O. tshawytscha*) fry abundance were compared. However, chinook and sockeye (*O. nerka*) smolt catches were significantly different ($p<0.05$) among marsh sites and were usually higher at disrupted sites. In nine sites in the North Arm and Deas Slough area chum fry residency was examined. At one transplant site (DE1) marked chum fry were caught up to 48 hours after release. No fry were caught 1 h after release at a transplant site (DI1) and a disrupted site (DE4). At the remaining sites fry were caught up to 1 and 3 hours after release. At all sites, over 80% of the total number of food organisms examined in chum fry stomachs was harpacticoid copepods. Mean submergence time for reference marshes ranged from 33.2 to 50.7%, but for transplanted sites the value ranged from 26.4 to 60.1%. Our study shows that numerous factors need to be examined in determining if restored marshes will function as natural habitats. The development of a standardized set of reference criteria would assist in evaluating whether or not transplanted marshes are functioning as designed.

Keywords: transplanted vegetation, marsh, Fraser River estuary

RÉSUMÉ

Levings, C.D. and D.J.H. Nishimura [ed.]. 1996. Created and restored sedge marshes in the lower Fraser River and estuary: An evaluation of their functioning as fish habitat. Can. Tech. Rep. Fish. Aquat. Sci. 2126: 143 p.

On a effectué durant la période 1991-1993 des comparaisons écologiques entre des sites palustres transplantés, des sites naturels (de référence) et des sites perturbés (sans végétation) le long du cours inférieur et de l'estuaire du Fraser (Colombie-Britannique). L'étude a porté sur le couvert végétal et la biomasse de la végétation, l'abondance des invertébrés, l'abondance des poissons, le temps de séjour et la nourriture des poissons et la période de submersion des trois types d'habitats. À trois sites transplantés, la biomasse de la récolte sur pied était dans la fourchette de valeurs pour les sites de référence, mais beaucoup plus faible à un site instable où des sédiments s'étaient effondrés. Dans huit sites transplantés, la couverture de carex de Lyngbye (*Carex lyngbyei*) était, en moyenne pour toute la région étudiée, inférieure de 50 % à celle observée dans les sites de référence avoisinants; les joncs (*Juncus* spp.) étaient plus abondants dans les sites transplantés. Dans tous les tronçons étudiés, l'abondance des invertébrés aux sites transplantés et aux sites de référence était significativement plus élevée qu'aux sites perturbés. Dans plusieurs cas, l'abondance des invertébrés était plus importante aux sites transplantés qu'aux sites de référence. La comparaison de l'abondance d'alevins de saumon kéta (*Oncorhynchus keta*) et de saumon quinnat (*O. tshawytscha*) n'a révélé aucune différence significative ($p>0,05$) entre les sites. Toutefois, les captures de smolts quinnats et rouges (*O. nerka*) étaient signicativement différentes ($p<0,05$) d'un site à l'autre et étaient généralement plus élevées aux sites perturbés. À neuf sites de la région du bras North et du faux chenal Deas, on a examiné le temps de séjour des smolts kétas. À un site transplanté (DE1), des alevins de kéta étiquetés ont été rattrapés jusqu'à 48 heures après avoir été relâchés. Aucun alevin n'a été rattrapé 1 heure après le lâcher au site transplanté D11 ni au site perturbé DE4. Aux autres sites, des alevins ont été rattrapés jusqu'à 1 heure et 3 heures après avoir été relâchés. À tous les sites, plus de 80 % du nombre total d'organismes alimentaires trouvés dans les estomacs d'alevins de kéta étaient des harpacticoïdes. Le temps moyen de submersion pour les marais de référence variait de 33,2 à 50,7 %, mais de 26,4 à 60,1 % pour les sites transplantés. Notre étude montre que de nombreux facteurs doivent être examinés pour savoir si les marais restaurés peuvent jouer le rôle d'un habitat naturel. L'élaboration d'un ensemble normalisé de critères de référence aiderait à évaluer si les marais transplantés fonctionnent ou non comme prévu.

Mots-clés : végétation transplantée, marais, estuaire du Fraser.

CHAPTER 1

Introduction: Basic Design of a Study to Compare Plant, Invertebrate and Fish Ecology at Disrupted, Transplanted and Natural Marshes in the Lower Fraser River and Estuary

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INTRODUCTION

This document summarizes the results of a research program conducted under the auspices of the Fraser River Action Plan to evaluate the success of a subsample of fish habitat restoration and compensation projects in the lower Fraser River and estuary. More than 60 restoration, enhancement or compensation projects have been completed in the estuary since 1983 (Langer et al. 1994). The importance of estuarine sedge habitat, especially Lyngbyei's sedge (*Carex lyngbyei* L.), for growth and survival of juvenile salmonids in the estuary, has been appreciated for several years (e.g., Levy and Northcote 1982; Levings et al. 1991) and therefore most of the projects have focused on sedge marsh transplants. Langer et al. (1994) developed a "no net loss" ledger for marsh habitat in the estuary that considered all of the projects and concluded that net gain had been achieved. However, the performance of the transplanted marshes to produce detritus, support invertebrate fish food organisms, and maintain fish had not been evaluated. We realized that a complete functional evaluation would involve assessment of more general ecological parameters, such as hydrological capability and sediment conditioning. As pointed out by the National Research Council (NRC 1992), rehabilitation of a total ecosystem should consider those and several other key parameters.

In this document, we compared the performance of transplanted sedge marshes at restoration and compensation projects in the lower Fraser River and estuary. These data were necessary to determine if the projects were fulfilling the "no net loss" and "net gain" goals of the Department of Fisheries and Oceans Policy for Fish Habitat Management (DFO 1986). AN1 (Table 1) was a damaged site where various agencies collaborated to restore the fish habitat. The remainder of the study sites were locations where fish habitat was created to compensate for habitat loss due to industrial and residential development. A management record for all of the restoration and compensation projects in the Fraser River estuary is maintained at the Fraser River Estuary Management Program office (New Westminster, BC). In the following chapters, we have used the term "transplant site" to refer to the restoration and compensation areas, collectively.

The results of only a few estuarine transplant projects in our region have been published and none have focused on the Fraser River estuary. Long-term monitoring of the Miller Sands marsh creation project in the Columbia River estuary dealt with plants, invertebrates and fish (Newling and Landin 1985; Landin et al. 1989) but most reports that are available in the literature have usually dealt with separate components of the ecosystem. For example, Williams (1993) and Pomeroy et al. (1981) gave results of transplants for slough or Lyngbyei's sedge (*Carex lyngbyei*), but these studies did not include any sampling of invertebrates or fish. More recently, Levings and Macdonald (1991; Campbell River estuary) and Shreffler et al. (1990, 1992; Puyallup River estuary) considered fish and invertebrate use of restored habitat but not plants. A preliminary study by Whitehouse et al. (1993) showed that adult dipterans, known to be important in the diet of young salmon (e.g., Levings et al. 1991), were about as abundant in transplanted sedge marshes on the Fraser and Campbell River estuaries relative to natural marshes.

Evaluations were conducted between 1991 and 1993 on the North Arm, South Arm and Queens Reach in the lower Fraser River (Figure 1, Table 1). The results are presented in chapter form, with botanical, invertebrate and fisheries information presented consecutively. Each transplant site was matched by a natural or reference habitat. We have used the term "reference" in the following because, in some instances, sedge marshes in the Fraser River estuary have colonized sites affected by industrial development. As an example, marshes on the south side of the Albion Jetty developed on dredged sand. An unvegetated intertidal area was also sampled to contrast a disrupted area with a site that had not been transplanted. The marshes had been transplanted between two and six years ago. Sample triads of these three habitat types were within specific reaches of the river mentioned above. Sites were usually within a one or two kilometre reach. For the South Arm, a reference marsh on the mainstem, as well as one in Deas Slough where a compensation marsh was created, were both sampled.

In addition to the biological parameters, our study design included measurements of elevation, considered a key biophysical factor for the ecological function of the transplanted marshes. Submergence/emergence ratios affect plant survival because they directly affect the hydrology of the marsh (Hutchinson 1982; Williams 1993; Envirocon 1980). Invertebrates used as fish food that have more aquatic affiliations (e.g., copepods) are also usually more abundant in more submerged marshes compared to drier habitats where semi-terrestrial organisms such as Collembola are more dominant (Levings et al. 1995). For fish, submergence is perhaps the most critical parameter since it influences the availability of the entire habitat area to fish. Using these empirical data and an oceanographic model (Stronach 1995), we were able to compare differences in submergence between transplanted and reference marshes.

A summary and integration of results is given which provides possible biological and biophysical criteria for successful marsh transplants in the Fraser River estuary. Several documents have given general information on the engineering techniques and biological factors that can be important to create sedge marshes in estuaries and rivers (e.g., USCE 1988; Adams and Whyte 1990; NRC 1992; Kusler and Kentula 1989). Envirocon (1980) provided data specific to the Fraser River estuary, but dealt with an evaluation of conditions at nine sites chosen by harbour managers because the locations provided options for disposal of dredged sand. Most of the assessment of environmental conditions, especially submergence and currents, in the Envirocon (1980) project was done using a scaled physical model specifically built for the evaluations. However, recently developed mathematical models now allow these assessments to be completed on any given reach of the estuary and lower river. These contemporary models were used (Stronach 1995) to evaluate the subsample of marshes created since 1985 and to assist in criteria development.

Figures 5 to 12 provides an overview of the transplanted and disrupted habitat examined during this study, including examples of some of the methods used in marsh transplant projects. These photographs were obtained on two separate occasions (see figure caption for actual date): June 1991 which was during the field work; and June 1996 which was three to five years after the field work for the study was completed. During the photograph work we noted qualitative changes in the plant communities at some of the sites, reinforcing our recommendations that long term monitoring is required to document the decade-scale changes that are likely in both natural and transplanted habitats.

ACKNOWLEDGEMENTS

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Table 1. Location, code and description of marsh study sites in the Fraser River estuary, BC.

Name of Location	Site Code	Habitat Type	Description and Comments
NORTH ARM REACH			
Deering (north shore)	DI1	Compensation[a]	marsh planted 1989; protective boom; habitat loss on Deering Island
Deering (south shore)	DI2 (low)	Reference[b]	low bench of an eroded shore
Deering (south shore)	DI2 (high)	Reference	high bench of an eroded shore
Deering Island channel	DI4	Disrupted[c]	riprap revetment
McDonald Slough	MD1	Reference	mouth of natural channel
McDonald Slough	MD2	Restoration[d]	dendritic channel constructed 1990
McDonald Slough	MD3	Restoration	dendritic channel constructed 1990
Angus (north shore)	AN1	Restoration	marsh planted 1986; protective boom in disrepair
Angus (south shore)	AN2	Reference	
Richmond Island (downstream end)	AN3	Compensation	embayment created and marsh planted 1989; habitat loss on upstream end of Richmond Island
McDonald Beach (south shore)	AN4	Disrupted	unvegetated sandy beach; note coded as AN6 in Stronach (1995)
SOUTH ARM REACH			
Deas Slough	DE1	Compensation	marsh planted 1989; habitat loss was on mainstem est. 1 km upstream of DE2
Mainstem (north shore)	DE2	Reference	
Deas Slough	DE3	Reference	
BC Ferries dry-dock	DE4	Disrupted	unvegetated sand/silt beach
Deas Slough	DE5	Reference	sampled in 1992 for plant biomass only
Annacis Island (south shore)	AC1	Reference	
Patrick Bay (south shore)	AC2	Compensation	marsh planted 1985 in a created embayment; protective boom; habitat loss was in immediate vicinity
Seacadet Camp (south shore)	AC3	Disrupted	unvegetated sandy beach
Carter St. Park (north shore)	AC4	Compensation	marsh planted 1991; protective boom; planned for possible habitat loss est. 1 km downstream
QUEENS REACH			
Fletcher Challenge (north shore)	PM1	Compensation	habitat loss was est. 500 m downstream; natural recolonization after site preparation in 1991; protective boom
Port Mann (north shore)	PM2	Disrupted	unvegetated sand/gravel beach
Port Mann (north shore)	PM3	Reference	near mouth of Coquitlam River
CN Railway (south shore)	CN1	Compensation	marsh planted 1988; habitat loss was in immediate vicinity
CN Railway (south shore)	CN2	Reference	

Definition of Habitat Types:

[a] vegetation transplanted and/or habitat developed in compensation for fish habitat loss

[b] natural vegetated marsh site used as a reference habitat

[c] destruction/disruption of a natural marsh or a non-vegetated site to simulate a disrupted fish habitat

[d] fish habitat improvement (i.e., transplant of marsh or channelling to increase perimeter/area) of a natural marsh impacted by anthropogenic influences

Table 1 continued.

Site Code	Organisms Sampled			Location of Sampling Site	
	Plants	Invertebrates	Fish	Latitude/Longitude[1]	Northing/Easting[2] (s.d. in metres)
NORTH ARM REACH					
DI1	M,B	+	+	49°13'08.214"N 123°11'13.002"W[m]	5451813 (±3) 486386 (±4)
DI2 (low)	M,B	+	+	49°13'05.887"N 123°11'25.615"W[b]	5451741 (±3) 486131 (±1)
DI2 (high)	M,B	+	+	49°13'05.887"N 123°11'25.615"W[b]	5451741 (±3) 486131 (±1)
DI4	A	+	+	49°13'06.117"N 123°10'59.338"W[b]	5451747 (±2) 486663 (±2)
MD1	A	+[3]	-	49°12'48.283"N 123°11'29.308"W[b]	5451198 (±2) 486055 (±2)
MD2	A	+[3]	-	49°12'48.242"N 123°11'26.003"W[b]	5451196 (±3) 486122 (±1)
MD3	A	+[3]	-		
AN1	M,B	+	+	49°12'27.939"N 123°09'09.195"W[m]	5450563 (±4) 488888 (±2)
AN2	M,B	+	+	49°12'26.207"N 123°09'19.624"W[b]	5450510 (±3) 488677 (±2)
AN3	A	+	+	49°12'14.081"N 123°08'52.059"W[b]	5450134 (±3) 489234 (±2)
AN4	A	+	+	49°12'35.139"N 123°09'31.575"W[m]	5450786 (±3) 488436 (±2)
SOUTH ARM REACH					
DE1	M,B	+	+	49°07'20.162"N 123°03'32.489"W[b]	5441049 (±2) 495693 (±2)
DE2	M,B	+	+	49°08'12.538"N 123°03'32.990"W[m]	5442669 (±3) 495684 (±1)
DE3	M,B	+	+	49°06'58.570"N 123°04'02.854"W[b]	5440383 (±4) 495077 (±3)
DE4	A	+	+	49°07'19.318"N 123°05'08.551"W[b]	5441025 (±1) 493746 (±1)
DE5	B	-	-	49°07'03.981"N 123°03'36.354"W[b]	5440550 (±2) 495615 (±2)
AC1	M,B	+	+	49°10'14.847"N 122°57'40.062"W[s]	5446446 (±3) 502833 (±3)
AC2	M,B	+	+	49°10'29.452"N 122°57'10.599"W[s]	5446894 (±4) 503430 (±3)
AC3	A	+	+	49°11'03.886"N 122°56'00.139"W[m]	5447958 (±3) 504855 (±2)
AC4	M	+	+	49°10'59.177"N 122°55'22.020"W[m]	5447813 (±3) 504413 (±3)
QUEENS REACH					
PM1	M	+	+	49°13'21.747"N 122°50'20.759"W[s]	5452226 (±4) 511716 (±2)
PM2	A	+	+	49°13'18.191"N 122°49'37.615"W[m]	5452118 (±3) 512589 (±3)
PM3	M	+	+	49°13'29.354"N 122°48'39.793"W[s]	5452466 (±3) 513758 (±3)
CN1	M	+	+	49°12'41.837"N 122°46'16.103"W[s]	5451006 (±2) 516668 (±2)
CN2	M	+	+	49°12'41.582"N 122°46'08.516"W[s]	5450999 (±2) 516822 (±2)

M = mapped; B = biomass; A = absent; + = sampling done; - = no sampling done

[1] location determined using Global Positioning System (GPS) with differential correction
[b] measured from a boat along the sampling site
[m] measured near the middle of the marsh
[s] measured from a stake marking a sampling site within the marsh

[2] latitude/longitude converted to Universal Transverse Mercator (UTM)

[3] data on file at the West Vancouver Laboratory, Department of Fisheries and Oceans. Preliminary analyses are given in Stronach (1995).

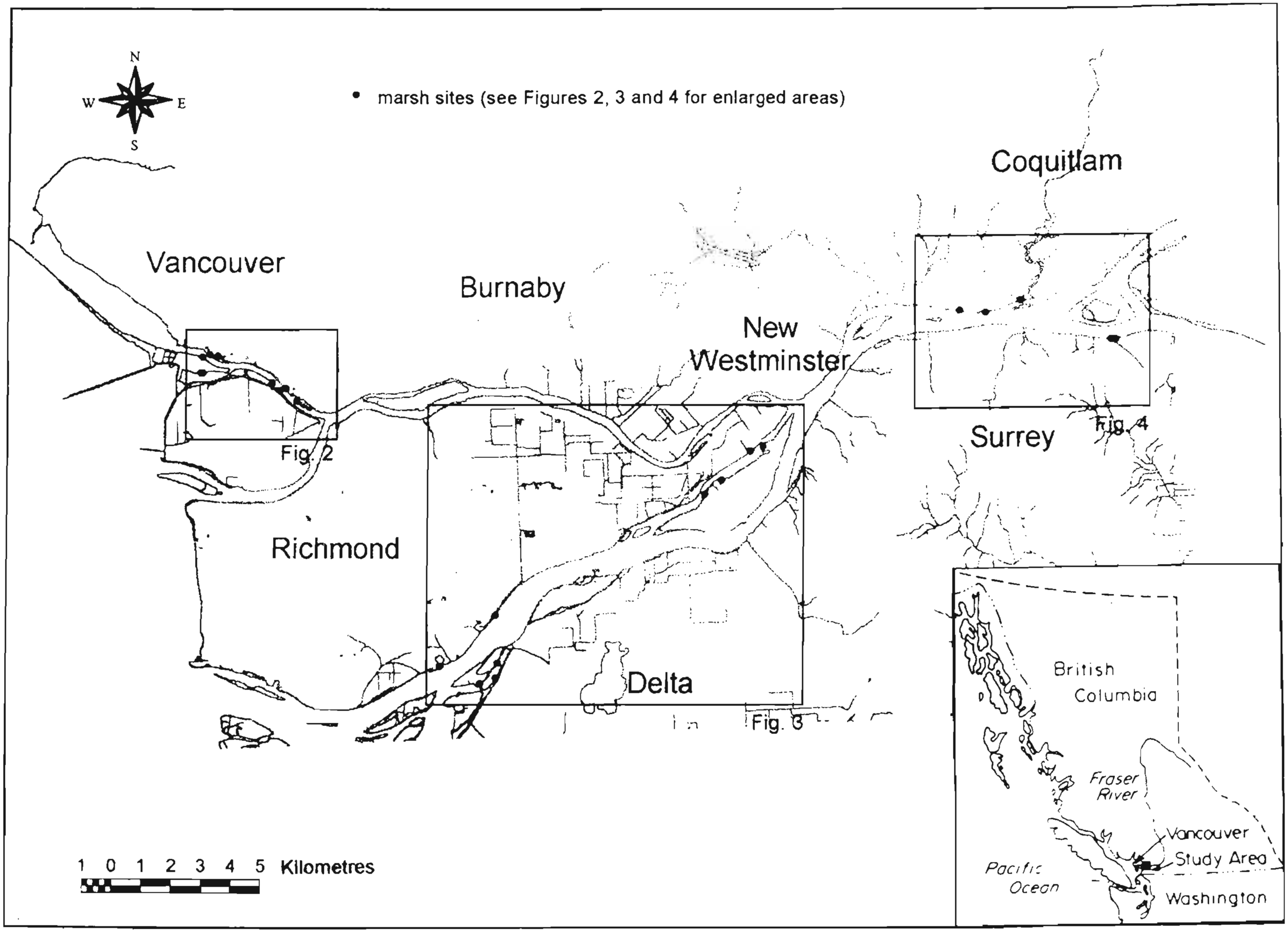

Figure 1. Location of marsh sites on the lower Fraser River and estuary, 1991 to 1993.

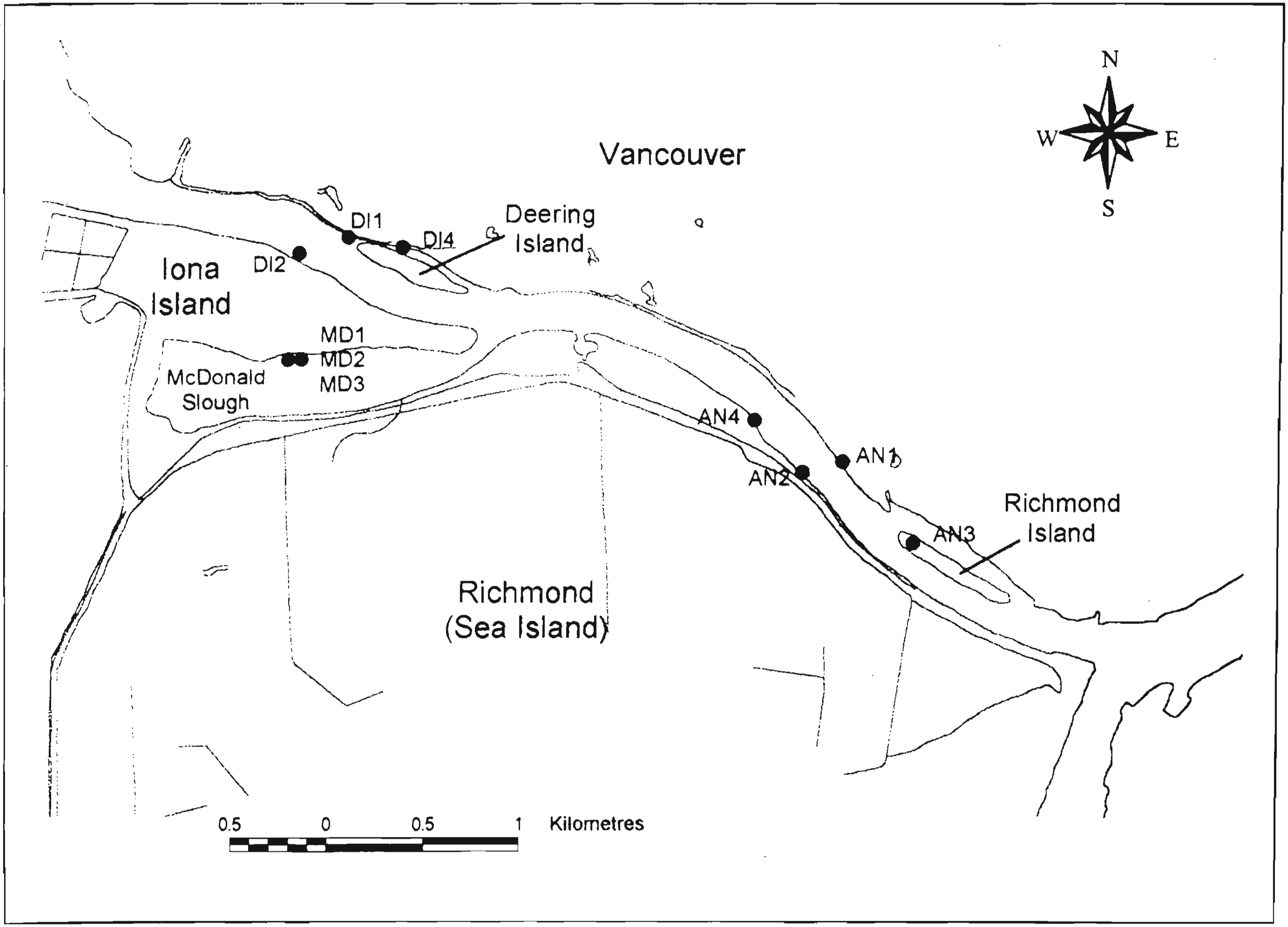

Figure 2. Location of study sites in the North Arm of the Fraser River estuary, 1991 to 1993.

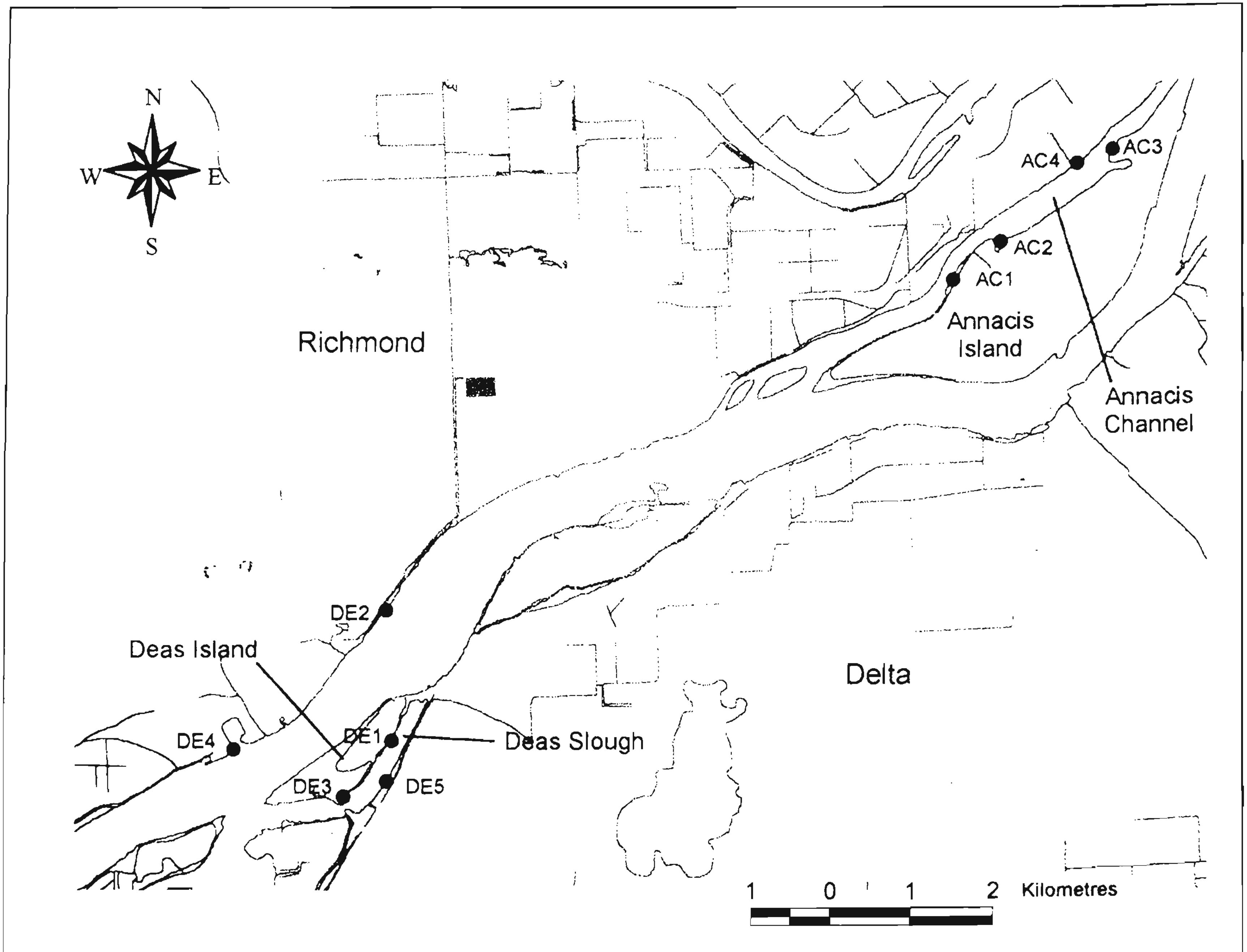

Figure 3. Location of study sites in the South Arm of the Fraser River estuary, 1991 to 1993.

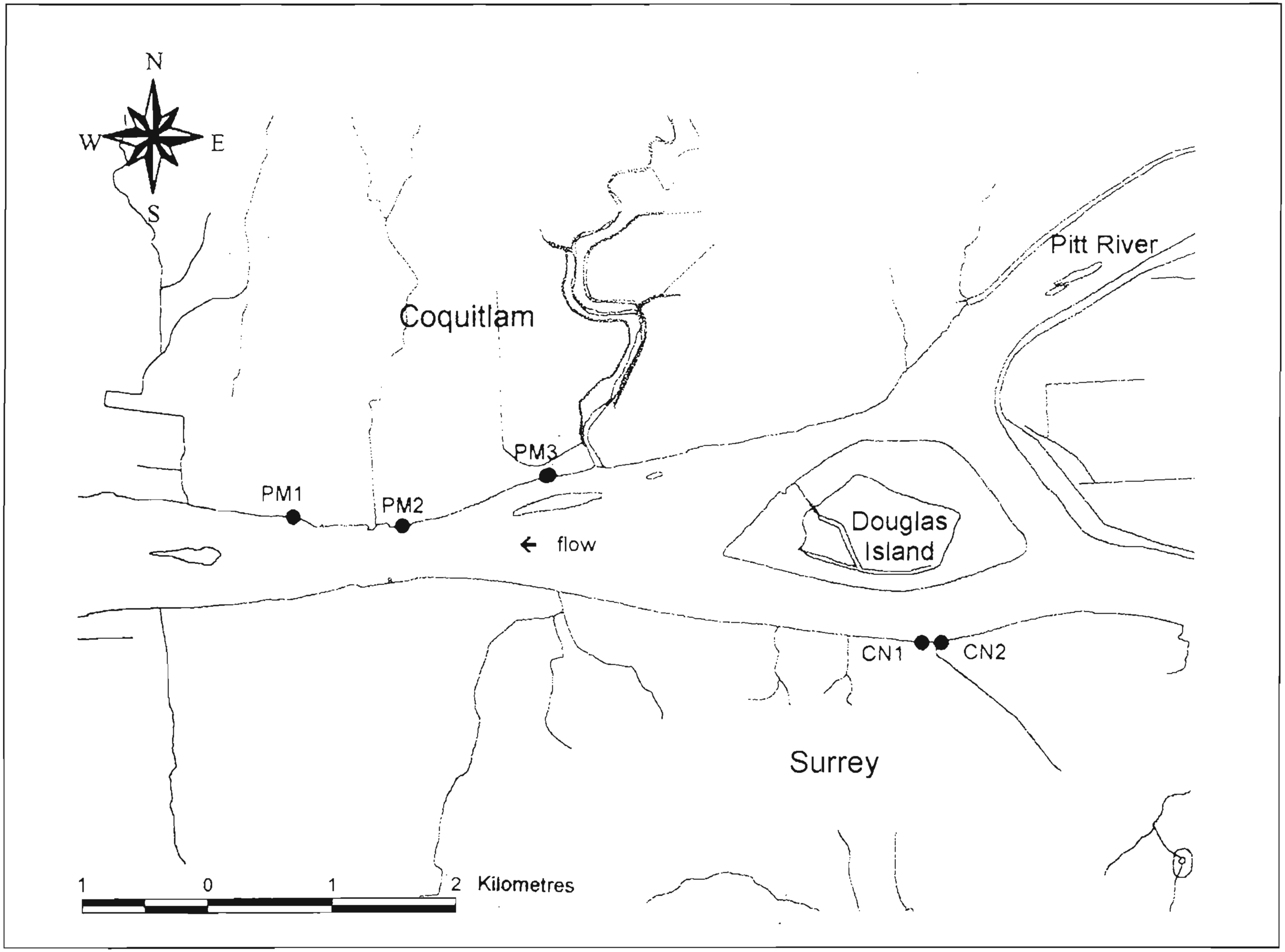

Figure 4. Location of study sites in Queens Reach of the Fraser River estuary, 1991 to 1993.

Figure 5. AN1 compensation site located in the North Arm (see Figure 2). Riprap placed along perimeter to prevent erosion of marsh and vegetation planted in 1986. Photographed 91-06-12.

Figure 6. DI1 compensation site in the North Arm (see Figure 2). Marsh planted in 1989. Photographed 91-06-12.

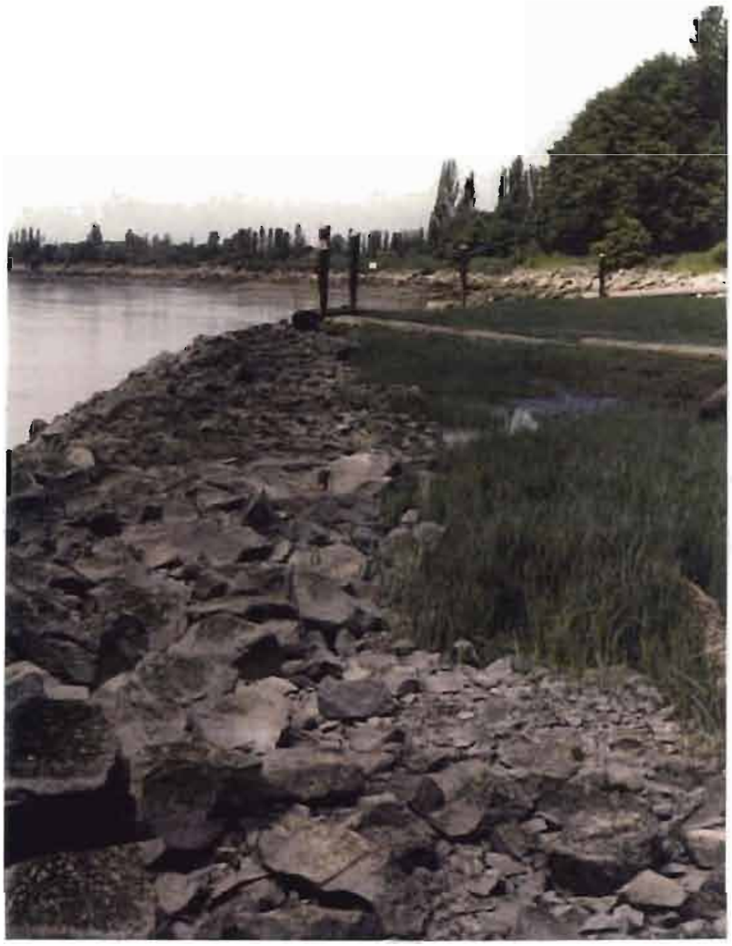

Figure 7.

DI1 compensation site located in the North Arm (see Figure 2). Shoreline view showing riprap used to prevent erosion of the marsh. Photographed 96-06-03.

Figure 8. DI1 compensation site in the North Arm (see Figure 2). Protective log boom prevents debris (e.g., driftwood) from destroying vegetation. Photographed 96-06-03.

Figure 9. AC4 compensation site in Annacis Channel of the South Arm (see Figure 3). Marsh planted in 1991. However, excessive debris in the marsh area demonstrates a failed log boom. Photographed 96-06-03.

Figure 10. AC2 compensation site in Annacis Channel of the South Arm (see Figure 3). An embayment was created and vegetation was transplanted in 1985. Photographed 91-06-12.

Figure 11. DI3 disrupted site at Deering Island in the North Arm (see Figure 2). Habitat loss due to placement of riprap revetment used to stabilize shores where residential development has occurred. Photographed 91-06-12.

Figure 12. AC3 disrupted site in Annacis Channel of the South Arm (see Figure 3). An unvegetated sandy beach simulated a disrupted site. Photographed 96-06-03.

CHAPTER 2

Changes in Fish Habitat in the Lower Fraser River Analyzed by Two Wetland Classification Systems[1]

by

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INTRODUCTION

This report provides data and descriptive information about historical losses of wetlands used by feeding and rearing juvenile salmon, in particular juvenile chinook (*Oncorhynchus tshawytscha*) and coho (*O. kisutch*) in the Fraser River floodplain (exclusive of the outer delta foreshore). Wetlands of the Fraser River floodplain contribute organic matter to the detritus-based food chain to which juvenile salmon are linked. Forested and non-forested wetlands provide habitat for a large variety of aquatic and terrestrial invertebrates which juvenile salmon prey upon.

Historical wetland losses were assessed on the basis of archival information available from surveyors' records (1859 to 1890; North and Teversham 1983), from aerial photographs (starting in 1938) and from present-day wetland inventory data. The main purpose of this study was to provide background information about the past and present extent of fish habitat in the Fraser River floodplain in three specific river reaches: the North Arm near Sea Island, the Queens Reach near the Coquitlam River (Port Mann) and the lower Harrison River. These areas have been studied since 1990 to determine the relationship between juvenile salmon and different types of marsh habitat: restored, disturbed and reference.

This report serves primarily as a data source and is accompanied by computer diskettes which contain digitized maps and dBASE records of historic floodplain vegetation and present-day wetlands (available from C.D. Levings).

METHODS

Historical Fish Habitat

Data Source

Data on historical fish habitat was derived from archival research and vegetation mapping undertaken by North and Teversham (1983). Their work was based on historical information from surveyors' notebooks (i.e., notes on vegetation types, soil conditions, presence of water, etc.), from which 26 vegetation types were interpreted, delineated and mapped at 1:50,000 scale, covering the floodplain of the lower Fraser River. For this project, we used the original manuscript maps of historical plant communities supplied by Margaret North, Geography Department, University of British Columbia (UBC). The manuscript maps consisted of seven 1:50,000 scale maps (NTS map sheets 92G/1, 92H/4, 92H/5, 92G/6, 92G/7, 92G/3 and 92G/2), which outline and identify each vegetation type. The maps were accompanied by a legend, which briefly described each vegetation type identified by a letter from "A" through "Z".

Classification of Fish Habitat: The 26 historical vegetation types were grouped on the basis of their likely use and availability for fish habitat (Table 1). This grouping was based on the degree of flooding, the occurrence of flood-tolerant plant species and the contiguity to the Fraser River.

Digitizing and Database Construction

The historical manuscript maps were digitized to show the historical shoreline and polygons of homogeneous vegetation types. Each polygon was identified with a unique alphanumeric code and linked to a dBASE record. The alphanumeric code was used to identify the vegetation type. Each data record was assigned a geographical coordinate (latitude and longitude) and had its area calculated in hectares.

Interpretation of Historical Aerial Photographs

Historical aerial photographs available at the UBC Geography Department were used to assess the nature of habitat change from 1938 to present-day.

Present-Day Fish Habitat

Data Source

Information on present-day fish habitat was derived from the Canadian Wildlife Service (CWS) "Wetlands of the Fraser Lowland" database (Ward et al. 1992). This information was available in dBASE format and each data record corresponded to a polygon object representing a specific wetland unit. The digitized wetland polygons were made available by the CWS along with a digitized base map of the Fraser Lowland (scale 1:50,000). The boundary of the Fraser Lowland (cf. Ward et al. 1992) encompassed a larger area than the study area boundary of the Fraser River floodplain (cf. North and Teversham 1983). This was because: 1) the delta front wetlands were excluded, as discussed later; and, 2) the Fraser Lowland corresponded to the area below 150 m elevation, whereas the Fraser floodplain corresponded to the alluvial deposits which extend from the river to the first significant break of slope (see Ward et al. 1992; North and Teversham 1983, for further details on their respective study boundaries).

Classification of Fish Habitat: The CWS database used the Canadian Wetland Classification System (CWCS) (Environment Canada 1987) for classifying wetland habitats. Gravel bars were included as an additional wetland category. Fish habitat was determined from this database by eliminating those wetland types which are generally not supplied with Fraser River floodwater (see Table 2) as well as wetlands outside the study area outlined by North and Teversham (1983). Further discussions on the selection of wetlands representing fish habitat are given below.

Marsh Transplant Sites

The CWS database included wetland sites that were restored, enhanced or created between 1983 and 1988. Transplant sites were identified and described for each of the three river reaches studied (i.e., lower end of the North Arm, Queens Reach near the Coquitlam River, lower Harrison River). This information was obtained from a habitat project database for the entire Fraser River Estuary Management Program (FREMP) area of the Fraser estuary (Kistritz, 1996).

Digitizing and Database Construction

Digitized polygon files and map layers provided by the CWS were translated into QUIKMap format. Each polygon representing a discrete wetland unit existed as a dBASE record and included geographic coordinates (latitude and longitude) and total area in hectares.

Assessment of Fish Habitat Changes

Comparison of Habitat Databases

Changes in fish habitat were determined by comparing the amount of historical with present-day wetland vegetation.

RESULTS AND DISCUSSION

Identification of Historical "Fish Habitat"

The identification of historical vegetation types representing fish habitat was based on the following considerations:

1. Frequency and Duration of Flooding. Any plant community which was noted (North and Teversham 1983) to be subject to an appreciable degree of flooding was included as fish habitat. For example, vegetation types included as fish habitat were described as having "...[annual exposure to river flooding]...[subject to flooding]...[within reach of occasional flood]..." (op. cit.).

2. Presence of Flood-tolerant Plant Species. Vegetation types containing flood-tolerant plant species (e.g., bulrush, cattail, saltgrass, hardhack, willow, alder, cottonwood, maple, cedar, hemlock, crabapple, spruce) were included as fish habitat.

3. Other Habitat. Plant communities not considered to be fish habitat included drier upstream sites that were not a true floodplain type, and those not supplied with Fraser

River floodwaters (e.g., bogs and seepage wetlands). Vegetation types not considered to be juvenile salmon habitat were as follows:

a) Bogs ("G" Bog, "I" Cranberry Marsh, "V" Mixed Coniferous, "Y" Pine Forest), which were considered to be hydrologically isolated from Fraser River floodwaters.

b) Areas not subjected to Fraser River flooding ("L" Alder Bottomland, "T" Cedar Swamp) because they were on elevated ridges or hydrologically isolated from Fraser River floodwaters.

c) Areas associated with drier sites ("Q" Disturbed Forest, "W" Mixed Coniferous, "U" Douglas Fir) that support flood-sensitive plant species such as Douglas Fir.

Shallow water areas and gravel bars were also not included as fish habitat because there was no reliable historical record on the extent of these habitat types. Shallow water areas and gravel bars associated with the Fraser River and its tributaries were not included in historical surveys. Larger sloughs, creeks and shallow water bodies were recorded by historical surveys, although we do not have a complete map of all such shallow water.

Sumas Lake, which was connected to both the Fraser and Chilliwack rivers, was one of the largest historical shallow water areas of considerable significance for rearing juvenile salmon. The lake, which was completely drained and diked by 1924, represented a shallow water habitat of approximately 3,600 ha (Siemens 1968).

Table 1 lists the 26 vegetation types associated with the historical manuscript maps and shows those types that were identified as being likely historical fish habitat.

Quantity of Historical Fish Habitat

Once the historical manuscript maps were digitized, a database file was constructed. The file contained a total of 792 records, each representing a specific vegetation type in a specific geographical location. A summary of this data file is provided in Table 1 showing the total area (ha) represented by each vegetation type category. The total area (Table 1) of all historical vegetation types that represented fish habitat was 65,359.8 ha.

Identification of Present-Day Fish Habitat

The identification of present-day fish habitat was based on the CWS wetlands inventory data with the exception of the following areas:

1. Areas Without Hydrological Connections to the Fraser River. Any wetland type which, by definition, was confined to a separate or terminal drainage system was not considered to be

fish habitat for juvenile salmon. The following habitat categories were thus excluded: Terminal Basin Marsh, Seepage Track Marsh, Basin Swamp, and all bog types.

2. All Shallow Water and Gravel Bar Features. The Shallow Water and Gravel Bar wetland categories were not considered in our comparative analysis of fish habitat because there was insufficient historical information on the presence of these features. However, they are obviously key categories of juvenile salmon habitat to be considered in any future studies. In 1968, the area of gravel used by spawning chum salmon alone was about 77 ha in the Fraser River and tributaries below Hope (Palmer 1972). Shallow water and unvegetated gravel bars were calculated by the CWS to cover 26,981 ha and 1,897.5 ha, respectively (Table 2). Some of the gravel areas were salmon spawning habitat, but most were unavailable because they were above water in the fall when the fish spawn. Gravel bars under water at the time of the CWS surveys (September) were not measured and therefore were not recorded in the data base.

3. Delta Front Marshes. Marshes along the delta front were excluded from our comparative analyses because the total extent of these marshes was not recorded by historical surveys. Therefore, prior to the first aerial photography in the 1930s, we have no reliable account of historical changes in the extent of the delta front wetlands. The following areas were excluded:

Unit No.	Location	Classification	Area (ha)
27	Boundary Bay	Coastal Marsh	150.5
22	Westham Island	Estuarine Marsh	746.2
21	Lulu Island	Estuarine Marsh	479.7
23	Brunswick Point	Estuarine Marsh	197.7
20	Sea & Iona Islands	Estuarine Marsh	126.3
24	Roberts Bank	Coastal Marsh	83.1
25	Tsawwassen	Coastal Marsh	1.2
28	Mud Bay	Estuarine Marsh	19.9
32	Musqueam Flats	Estuarine Marsh	17.4
33	North Arm Jetty	Estuarine Marsh	2.6
34	Musqueam Marsh	Estuarine Marsh	75.1
36	Iona Island North	Estuarine Marsh	5.6
TOTAL			**1,905.3**

4. Areas not within the Historical Study Area. Geographical areas in the present-day study that were not included in the historical study were excluded from the comparative assessment of fish habitat. The following table lists these wetlands by municipal location:

Municipality	Area (ha)
Central Fraser Valley uplands	321.3
Pitt Lake delta	163.5
North Vancouver City	1.9
North Vancouver District	5.9
Port Moody	6.3
Vancouver City	82.4
West Vancouver	14.0
TOTAL	**595.3**

Difference Between Historical and Present-Day Data

Two major differences were noteworthy between the present-day (Ward et al. 1992) and historical (North and Teversham 1983) vegetation categories and data.

1. The CWS classification system is based on the Canadian Wetland Classification System (CWCS) which was designed for general application of wetland vegetation systems across Canada. The historical plant community categories were much more local in context and covered more floodplain forest vegetation types.

2. Correspondence between historical and present-day forested wetlands was incomplete. The CWS wetland inventory data did not include floodplain forests because swamps were included as the only forested wetland type in the CWCS. Most areas on the Fraser River floodplain dominated by dense deciduous or coniferous trees or shrubs are not flooded frequently or long enough to be classified as true swamps. Floodplain forest or riparian woods would be a more appropriate classification for these areas. Some of these floodplain forests are associated with the banks and levees of the main river channel, sloughs and tributary streams. Other floodplain forest areas are associated with backwater areas and old abandoned channels distant from the present course of the Fraser River.

A summary table is provided to show the total area (ha) of present-day wetland types (Table 2). The total area of wetland categorized as fish habitat is 7,318.5 ha. This figure does not include shallow water and gravel bars because there are no comparable historical data for these habitats. A gross estimate of the total amount of historical fish habitat change in the entire study area of the Fraser River floodplain up to Hope would be as follows:

Present-day Wetland area = 7,319 ha
Historical Wetland area = 65,360 ha

Therefore, approximately 90% of all wetlands that were fish habitat have been lost, exclusive of the habitats presently located outside of the dikes on Sturgeon and Roberts Bank.

Most of this loss can be attributed to the substantial disappearance of the natural "grassland" and "grass with shrubs" vegetation type from the Fraser River floodplain. These grass-dominated vegetation types represented 54% of all the historical vegetation types. Grasslands included a number of different grass-dominated communities which were periodically flooded at high river stages. Early settlers were drawn to these "wet grass prairies" for the immediate grazing they offered, for their relatively "fertile" soils, and ultimately, the relative ease with which they could be diked and drained. Vast areas of grassland existed on Sea Island, Lulu Island, along the Serpentine and Nicomekl floodplain, and in the vicinity of Ladner, Sumas Lake, Fort Langley, Pitt Meadows, Matsqui and Chilliwack. Virtually all of these have been converted to agricultural lands or urban-industrial uses.

The estimated 90% loss of wetlands is imprecise because the historical data included vegetation types (mainly floodplain forests) which are not part of the present-day wetland inventory data. It is presently unknown how much floodplain forest remains, and how much of that is diked or undiked.

It is therefore recommended that the above figure estimating historical fish habitat changes for the entire study area should be used with caution and only after consideration of the above caveats. In particular, it should be recognized that there has been accretion of mudflats, and subsequent addition of marshes in the study area. According to Hutchinson et al. (1989) there has been significant addition of marshes.

Site-Specific Analyses: North Arm, Port Mann, Harrison River

A site-specific assessment of the features and attributes of historical and present-day fish habitat would provide much more precise estimates of historical changes. Such a site-specific assessment was undertaken for three river reaches: the North Arm near Sea Island, the Queens Reach near Port Mann, and the lower Harrison River.

North Arm

This study reach consisted of the North Arm from the middle of Iona Island to the east end of Mitchell Island, including Sea Island.

Historical Fish Habitat (1860 to 1890): Figure 1 provides a map of the historical wetlands that were considered to be fish habitat. The study reach supported 2,896 ha of wetland. Historically, Sea Island and the surrounding floodplain areas supported a network of sloughs and tidal channels.

Present-day Fish Habitat (1990): Figure 2 provides a map of the present-day wetlands considered to be fish habitat. The numbers represent the unit numbers of wetlands occurring between the border lines. Letters represent transplant areas (described below). Present-day fish habitat consists of 109.2 ha of wetland. The only significant slough remaining in this river reach is Bath Slough which is not accessible to juvenile salmon because it is controlled by flap gates and a drainage pump.

Marsh Transplant Sites (1986 to 1992): The North Arm study reach presently contains eight marsh transplant sites (A and C to I, Figure 2; B is a restoration site) which were the result of habitat management requirements authorized by the Fraser River Estuary Management Program (FREMP). All of these transplant marsh habitat sites (3.4 ha) were included in the present-day database.

Changes in Fish Habitat: 1860 to Present-day (1990): Ninety-six percent of the wetland area had been reduced in this particular reach. Changes in the river shoreline configuration were examined by superimposing the present-day and historical maps. There appeared to have been a net gain in river foreshore area compared to losses over time. However, from our examination of aerial photographs, it was apparent that most of this gain in river foreshore area was due to filling with dredged sand, at the expense of existing shallow water fish habitat.

1860 to 1938: Almost all of the net reduction in fish habitat occurred within this time period. These losses likely occurred even before the turn of the century. In 1861, Sea Island was one of the first delta lands to be farmed (Ross 1979). Diking, draining and farming essentially removed all of the historical fish habitat with the exception of the delta front marshes (excluded in this assessment) and parts of river islands and side-channels. In 1938, most of the side-channels associated with small river islands remained relatively intact, including Iona Island and McDonald Slough.

1938 to 1963: During this time period, most of the remaining river islands and side-channels were filled with sand. A causeway was built to Iona Island, on which a sewage treatment plant and jetty had been constructed. Much of the agricultural land was converted to urban and industrial use, including the development of an airport on Sea Island.

1963 to 1989: During this time period, some of the last remaining pockets of fish habitat were lost, primarily due to filling with sand for stockpiling dredged material and for industrial development.

Port Mann

This study reach included the wetland habitats of the Fraser River Queens Reach from the Brunette River to the Coquitlam River, including Douglas Island.

Historical Fish Habitat (1860 to 1890): Figure 3 provides a map of the historical wetlands, which consisted of 684 ha of wetland. Included as fish habitat is the "non-fish" vegetation type (w010) surrounding Douglas Island. Douglas Island shoreline cannot be considered part of a "drier upstream site" so it was also included in the present-day wetland inventory. The bog vegetation units v005, v006, i006, i007, i008 were included as fish habitat because of their close proximity to the Coquitlam River.

Present-day Fish Habitat (1990): Figure 4 shows the extent of present-day wetland habitat. There is currently a total of 126 ha of wetland remaining in this river reach.

Marsh Transplant Sites: In this river reach there were two locations where marsh transplants were conducted or sites prepared for natural colonization. Site "A" was a 0.17 ha bench prepared to allow natural colonization and Site "B" is a marsh transplant site of approximately 0.45 ha.

Changes in Fish Habitat: 1860 to Present-Day (1990): The largest change in the amount of river foreshore was apparent along the south shore of Douglas Island. This area appeared to have gained some shoreline habitat over the past century. Based on a comparison between historical and present-day maps, other shorelines in this river reach remained relatively unchanged.

A comparison between historical and present-day fish habitat showed an 82% reduction in wetland area.

1860 to 1938: Within this time period, the lower Coquitlam River was diked and large areas of the lower Coquitlam floodplain were converted to agriculture (e.g., Colony Farms). By 1938, pockets of industry (sawmills, fishing ports) were already present in the Brunette, Sapperton and Fraser Mills area.

1938 to 1963: During this time period, more pockets of industrial development occurred along the northern floodplain of Queens Reach. Urban development appeared on the northern edge of the Coquitlam floodplain.

1963 to 1989: During this time period, the entire remaining natural floodplain along the northern shoreline of Queens Reach was industrialized. What remained of the historical wetlands were the undiked portions of the Coquitlam floodplain (largely within the Coquitlam Indian Reserve) and all of Douglas Island.

Harrison River

This study reach included the floodplain area from the mouth of the Harrison River to Morris Creek.

Historical Fish Habitat (1860 to 1890): Historical fish habitat was confined mainly to the areas at the mouth of the Harrison River, the Chehalis River and Morris Creek (Figure 5). Other areas were too steeply sloped to support fish habitat. The total amount of historical wetland was 1,247 ha.

Present-day Fish Habitat (1990): Figure 6 shows the present-day fish habitat which consists of 321 ha of wetland. Habitat areas shown in Figure 6 and indicated as having no marsh and swamp areas were classified as shallow water habitat (No. 388) or gravel bars (No. 388) in the CWS inventory. Gravel bars between the mouth of the Chehalis River and Weaver Creek are important chum spawning habitat. In 1968, 14.60 ha of spawning habitat were measured in this area (Palmer 1972).

Marsh Transplant and Created Sites: There were two adjacent habitat restoration sites ("A" and "B") located in the Chehalis delta (Figure 6). Approximately 0.1 ha of foreshore was planted with vegetation in the spring of 1992 to compensate for alienated rearing habitat. In 1979, 1980 and 1986, spawning habitat for chum salmon was created† by placing gravel over native substrate near the mouth of the Chehalis River, creating 1.5 ha of fish habitat. Finally, in 1965, 1.86 ha of sockeye and chum spawning habitat were created using spawning channels at the headwaters of Weaver Creek.

Changes in Fish Habitat: 1860 to Present-Day (1990): On the basis of a comparison between historical and present-day shorelines, the most noticeable changes were apparent in mid-

† development of habitat in advance of the Policy for the Management of Fish Habitat (DFO 1986)

channel islands (apparently gained habitat) and wetland habitat associated with small bays which appeared to have been lost. Both of these "apparent" changes may be erroneous. Small mid-channel islands may not have been noted in historical surveys. Therefore, to what extent of these are new habitat remains open to question. Widely fluctuating seasonal and annual water levels in this area cause exposure of parts of shoreline in dry periods and total flooding during wet periods. The observed location of certain shoreline areas may have depended on the flood conditions present when surveys were undertaken.

Historical losses in fish habitat were restricted to the floodplain area around Harrison Mills. This area was diked and converted to farmland, cutting off Bateson and Duncan sloughs from the Harrison River. These sloughs are presently controlled by pumps. Aerial photography older than 1963 was not available for this area, so it was difficult to determine the time sequence of development activities in this river reach.

There was an apparent 75% loss in wetland habitat, exclusive of gains in gravel bar habitat because of spawning channel creation. Present-day floodplain forests are not mapped and this might result in an overestimate in the historical loss of forested wetlands.

Implications of Historical Fish Habitat Changes of Specific Wetland Types

Non-Forested Wetlands

Utilization of marshes in the lower Fraser River by juvenile chinook and coho had been shown to occur primarily during the March to August downstream migration period. Marshes in the estuary, as well as those along the river mainstem, provide important rearing habitat (Levy and Northcote 1982). Chinook and chum fry utilizing lotic habitats in the mainstem likely do not reside in a fixed location, but rear as they gradually migrate downstream. Thus, the lower Fraser represents a rearing "corridor," continuously offering feeding opportunities to the juvenile fish. Therefore, any loss of riverine marsh is considered to be a loss of potential fish rearing habitat. Chinook and coho smolts tend to occupy deeper water habitats and do not rear directly in the marsh habitats. However, they are closely linked to marshes through their feeding behaviour. During the spring, when smolts are present in the lower Fraser, water levels rise and insect emergence increases with rising spring temperatures. These discharge pulses, particularly in the tidal zone below the Harrison River, flood backwaters, side-channels, lateral marshes and river margins, flush insects of all life stages into the riverine drift. Salmon fry established along the river margins are ready to take advantage of the increasingly available food source, and smolts occupying the slightly deeper water are ready to take advantage of the insect drift as well as salmonid fry. Therefore, although smolts do not physically occupy the river margins as do fry, these habitats are important for food production and availability to all young salmon.

Forested Wetlands

Forested wetlands can be placed into two categories: 1) those that are associated with river channels, creek mouths, sloughs or with water flowing into them; and, 2) backwater areas that are recharged during spring freshet or only in years of above average flooding.

Changes in forested wetlands in the lower reaches of small tributaries could have implications to rearing chinook fry during the spring downstream migration and to coho for rearing and overwintering. As previously mentioned, chinook fry will immigrate into the lower reaches of lower Fraser tributaries to utilize rearing habitat (Levings et al. 1995). Forested wetlands in the lower reaches are likely rich insect production areas that probably afford chinook fry a substantial feeding and rearing opportunity during their downstream migration.

Forested wetlands in the lower reaches of tributaries are also important to coho stocks from these tributaries. Juvenile coho have been known to migrate substantial distances (40 to 50 km) between summer rearing habitat and overwintering habitat (e.g., Fedorenko and Cook 1982). They will move from smaller tributaries to larger rivers in the summer and back into smaller tributaries for winter residency. Skeesick (1970) and Bustard and Narver (1975) both found good survival of coho that demonstrated this dispersal behaviour. It is likely that such fish would encounter and utilize preferred rearing areas. This may be the case in forested wetlands in the lower reaches of many lower Fraser tributaries such as the Chilliwack, Coquitlam and Harrison rivers.

Forested backwater areas not associated with tributaries would be important primarily during the spring downstream migration period as food production areas. As the Fraser River rises and these areas become inundated, insects are washed into the river which benefit both coho and chinook at all juvenile life stages.

As mentioned earlier, aquatic as well as terrestrial insects are important food for salmonids, especially chinook fry. Shoreline areas in backwater wetlands would provide good opportunities for feeding on this prey. Forested wetlands, both backwater and those associated with creeks and rivers, would also be important to other opportunistically feeding salmonid species, such as anadromous cutthroat trout (*Oncorhynchus clarki*) and Dolly Varden (*Salvelinus malma*).

Whether or not the historical loss of forested and non-forested wetlands (that primarily contributed to fish as food production areas) has had a significant effect on the total abundance of preferred food items is unknown and requires further work. For example, considering the wide distribution of chironomids (Whitehouse et al. 1993), the historical loss of habitats that were primarily important for chironomid production may not have had a substantial impact on overall fish production. However, research is required to determine if salmonid preferences for chironomids are species-specific, in which case it would be important to examine habitat loss in relation to species-specific utilization.

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Table 1. Historical vegetation types and their respective areas (ha). Classified by North and Teversham (1983). Vegetation types designated fish habitat are shown in italics.

Map Code	Vegetation type	Dominant Plants	Total Area (ha)	Fish habitat (ha)
A	*Marshes*	*bulrush, sedge, cattail*	822.3	822.3
B	*Saltgrass Prairie*	*saltgrass*	945.1	945.1
C	*Grassland*	*grasses*	17,691.5	17,691.5
D	*Burnt Forest*	*grass, hardhack*	792.5	792.5
E	*Grass with Shrubs*	*willow, hardhack, crabapple*	17,837.6	17,837.6
F	Moss With Trees	sphagnum, scattered pine, hemlock, spruce	939.8	*
K	*Willow Scrub*		1,279.3	1,279.3
H	*Crabapple Scrub*	*exclusive of ridges along Lulu and Westham Islands*	705	705
J	*Mixed Scrub*	*willow, rose, crabapple, hardhack*	4,826.7	4,826.7
G	Bog	labrador tea, cranberry, salal, pine	4,773.1	*
I	Cranberry Marsh	cranberry, pine	4,154.1	*
M	*Mixed Woodland*	*alder, willow, crabapple, hardhack*	2,680.9	2,680.9
L	Alder Bottomland	alder	398.2	*
N	*Mixed Woodland*	*cottonwood, alder, willow, crabapple*	6,238.5	6,238.5
O	*Mixed Woodland*	*cottonwood, maple, hazel, willow, aspen*	978	978
R	*Mixed Forest*	*cottonwood, cedar, alder, maple*	1,416.8	1,416.8
P	*Disturbed Forest*	*(fire or logging) cedar, cottonwood, alder, etc.*	1,716.8	1,716.8
Q	Disturbed Forest	Douglas fir, cedar, alder, birch, maple	8,585.8	*
S	*Cedar Forest*	*cedar, cottonwood, alder*	2,628.0	2,628.0
W	Mixed Coniferous	drier upstream sites	9,029.6	*
X	Mixed Coniferous	*spruce, cedar, hemlock, alder, willow, crabapple*	3,488.7	3,488.7
T	Cedar Swamp	cedar, alder, willow, crabapple, hardhack, skunk cabbage	1,198.2	*
Z	*Spruce Forest*	*spruce, willow, crabapple, alder, briars, vine maple*	1,312.1	1,312.1
U	Douglas Fir	Douglas fir, hawthorn, Oregon grape	103.3	*
V	Mixed Coniferous	(on peat) cedar, spruce, pine, hemlock, labrador tea, etc.	1,088.4	*
Y	Pine Forest	pine, birch, moss	603.1	*
		Total	**96,233.4**	**65,359.8**

* Vegetation type not included in comparative analyses

Table 2. Present-day wetland types (cf. Ward et al. 1992). Fish habitat are shown in italics.

Wetland Classification	Relevance to Comparative Analysis with Historical Data	Present Area (ha)
1. MARSH		
Estuarine	*some occurrence within historical study area*	2,445.2
Coastal	*no occurrence within historical study area*	235.2
Tidal Freshwater	*well represented in historical study area*	539.7
Floodplain	*well represented in historical study area*	1,146.9
Stream	*well represented in historical study area*	969.6
Active Delta	*well represented in historical study area*	268.2
Terminal Basin	hydrologically isolated from Fraser R. floodwaters	54.4
Shallow Basin	*well represented in historical study area*	118.9
Seepage Track	hydrologically isolated from Fraser R. floodwaters	34.4
Shore		299.2
SUBTOTAL OF FISH HABITAT FOR MARSH		6,022.9
2. SWAMP		
Stream	*well represented in historical study area*	178.3
Basin	hydrologically isolated from Fraser R. floodwaters	12.3
Floodplain	*well represented in historical study area*	1,246.5
SUBTOTAL OF FISH HABITAT FOR SWAMP		1,424.8
3. BOG		
Domed	hydrologically isolated from Fraser R. floodwaters	1,496.7
Shore	hydrologically isolated from Fraser R. floodwaters	207.3
Basin	hydrologically isolated from Fraser R. floodwaters	17.7
Flat	hydrologically isolated from Fraser R. floodwaters	160.8
SUBTOTAL OF FISH HABITAT FOR BOG		0
4. FEN		
Stream	*included in the swamp category*	1,189.4
Shore	*included in the swamp category*	1,182.0
SUBTOTAL OF FISH HABITAT FOR FEN		2,371.4

Table 2 continued.

Wetland Classification	Relevance to Comparative Analysis with Historical Data	Present Area (ha)
5. SHALLOW WATER		
Stream	not comparable to historical information	2,983.2
Oxbow	not comparable to historical information	697.4
Delta	not comparable to historical information	388.2
Terminal Basin	not comparable to historical information	0.6
Shallow Basin	not comparable to historical information	461.3
Kettle	not comparable to historical information	31.5
Shore	not comparable to historical information	294.4
Estuarine	not comparable to historical information	14,487.8
Tidal	not comparable to historical information	7,600.0
Non-tidal	not comparable to historical information	36.6
SUBTOTAL OF FISH HABITAT FOR SHALLOW WATER		0
6. GRAVEL BARS (unvegetated)	not comparable to historical information	1,897.5
SUBTOTAL OF FISH HABITAT FOR GRAVEL BARS		0
	SUBTOTAL OF ALL FISH HABITAT	9,819.1
	LESS WETLANDS OUTSIDE STUDY AREA	-595.3
	LESS DELTA FRONT MARSHES	-1,905.3
	TOTAL FISH HABITAT	**7,318.5**

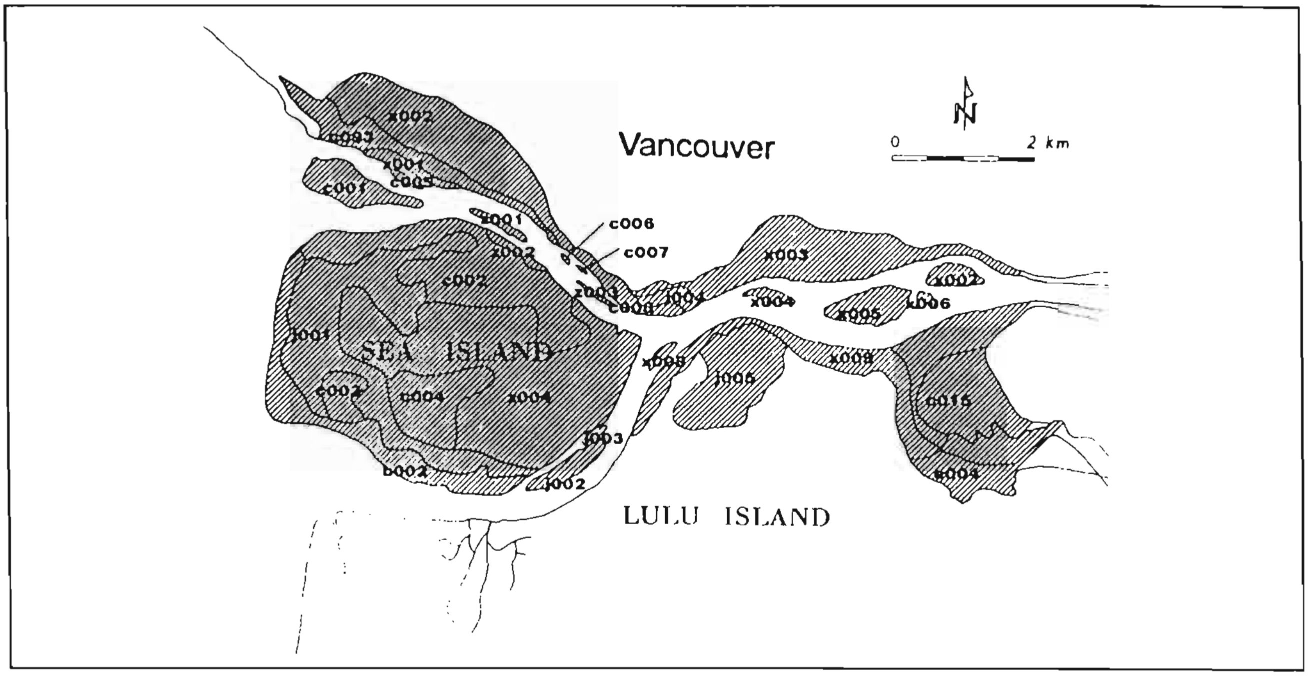

Figure 1. Historical fish habitat (2,896 ha) of North Arm River Reach. Alphanumeric designations refer to vegetation types (see Table 1).

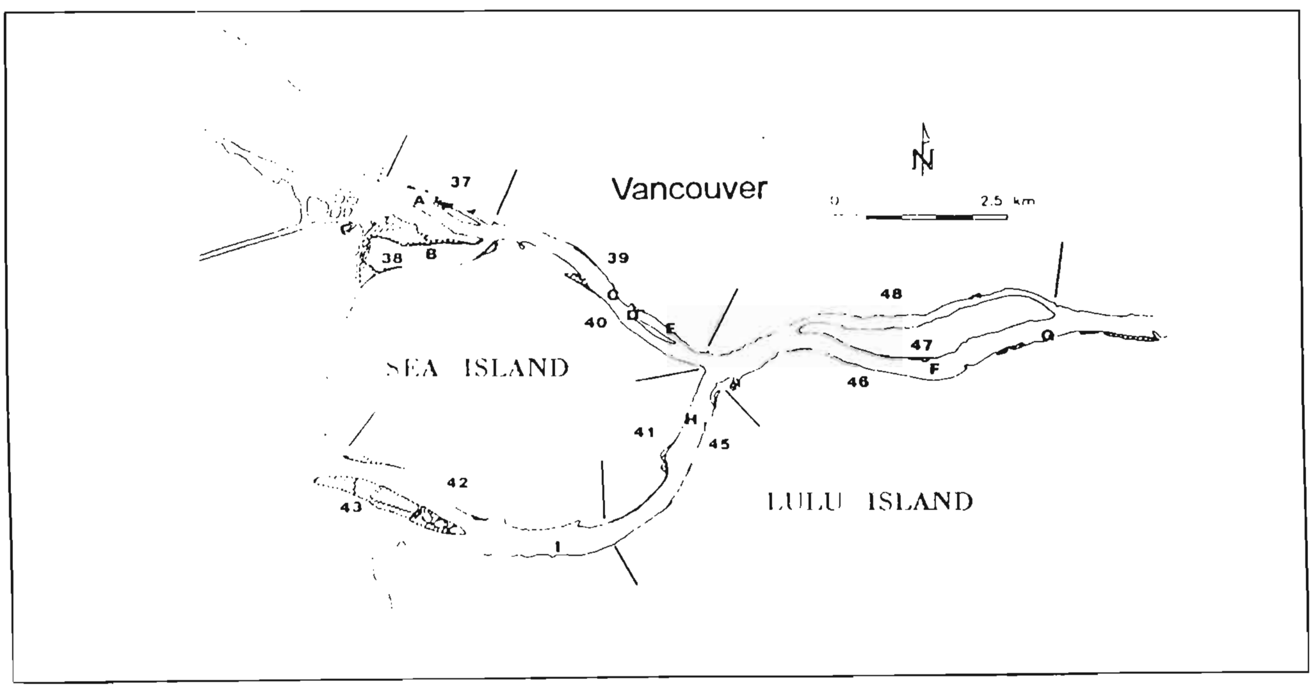

Figure 2. Present-day wetlands (109.2 ha) of North Arm River Reach. Numbers are wetland units occurring between line segments. Except for B (restoration), letters show the location of marsh transplant sites.

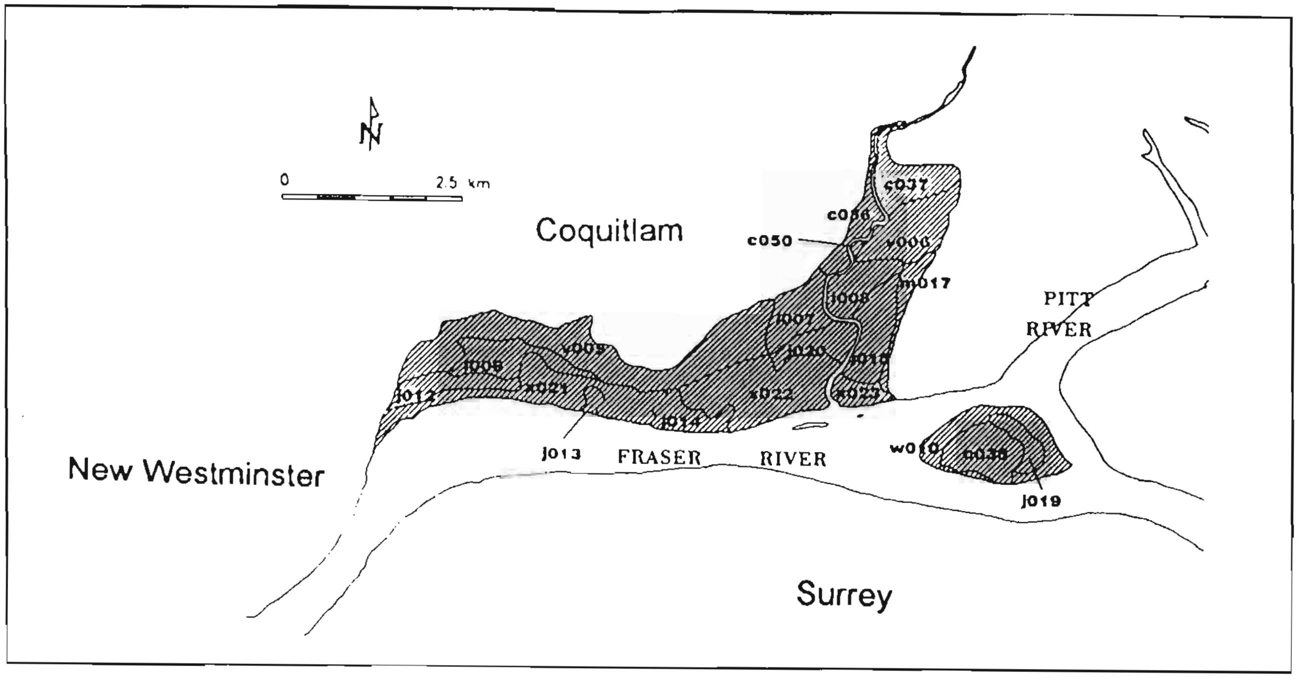

Figure 3. Historical fish habitat (684 ha) of Queens Reach. Alphanumeric designations refer to vegetation types (see Table 1).

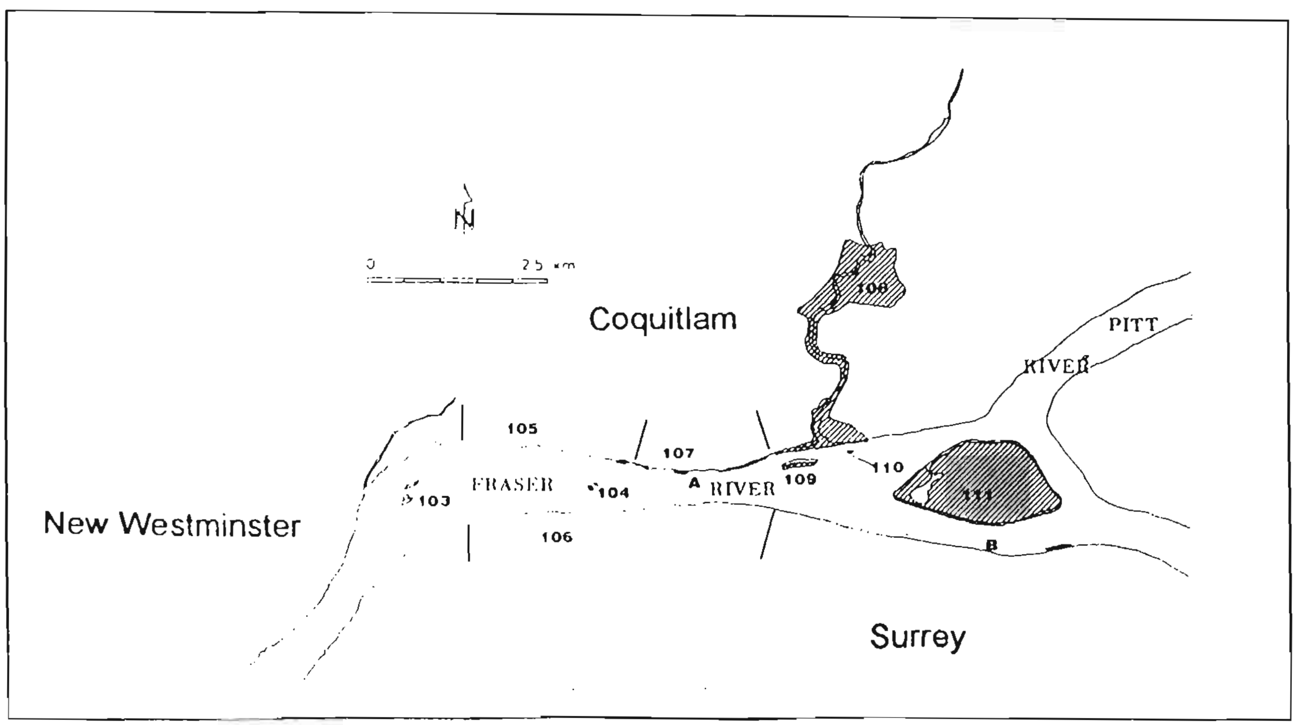

Figure 4. Present-day wetlands (126 ha) of Queens Reach. Numbers are wetland units occurring between line segments. Letters show the location of marsh transplant sites.

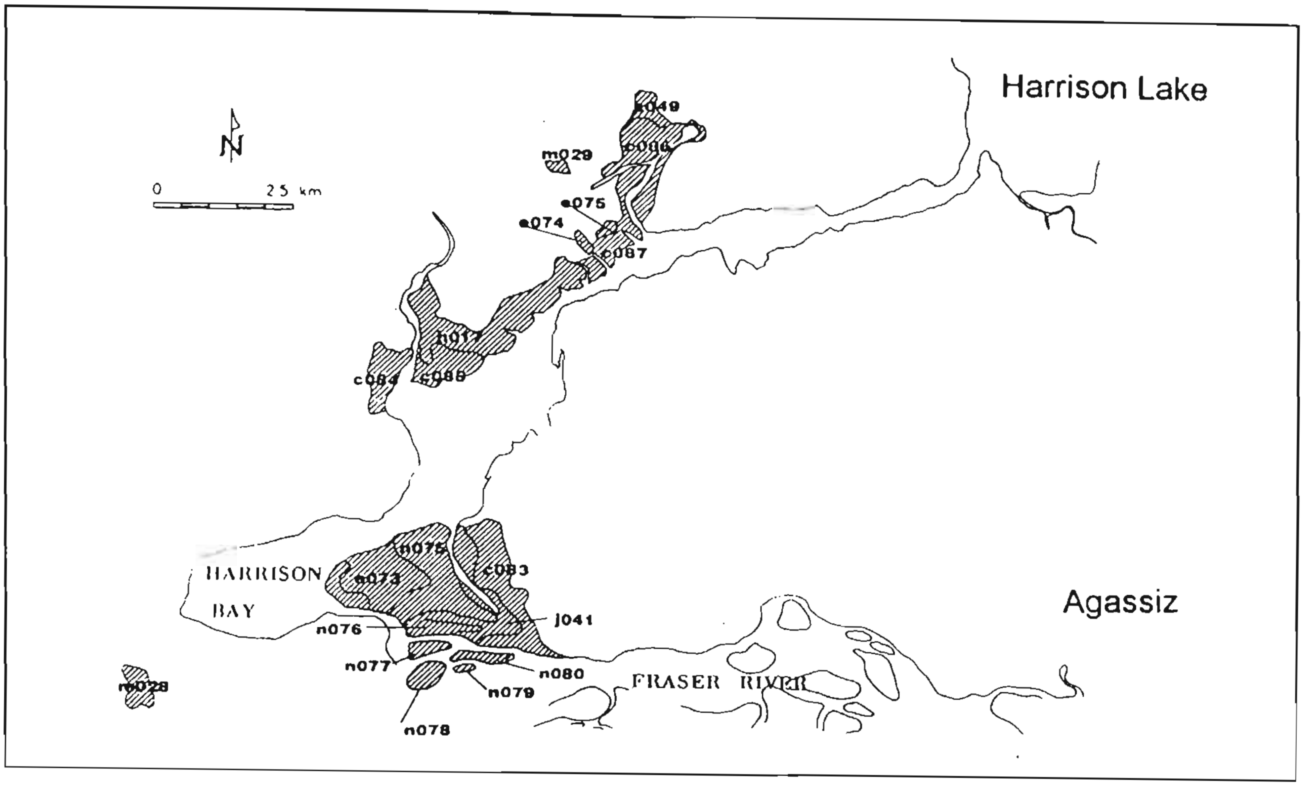

Figure 5. Historical fish habitat (1,247 ha) of the lower Harrison River reach. Alphanumeric designations refer to vegetation types (see Table 1).

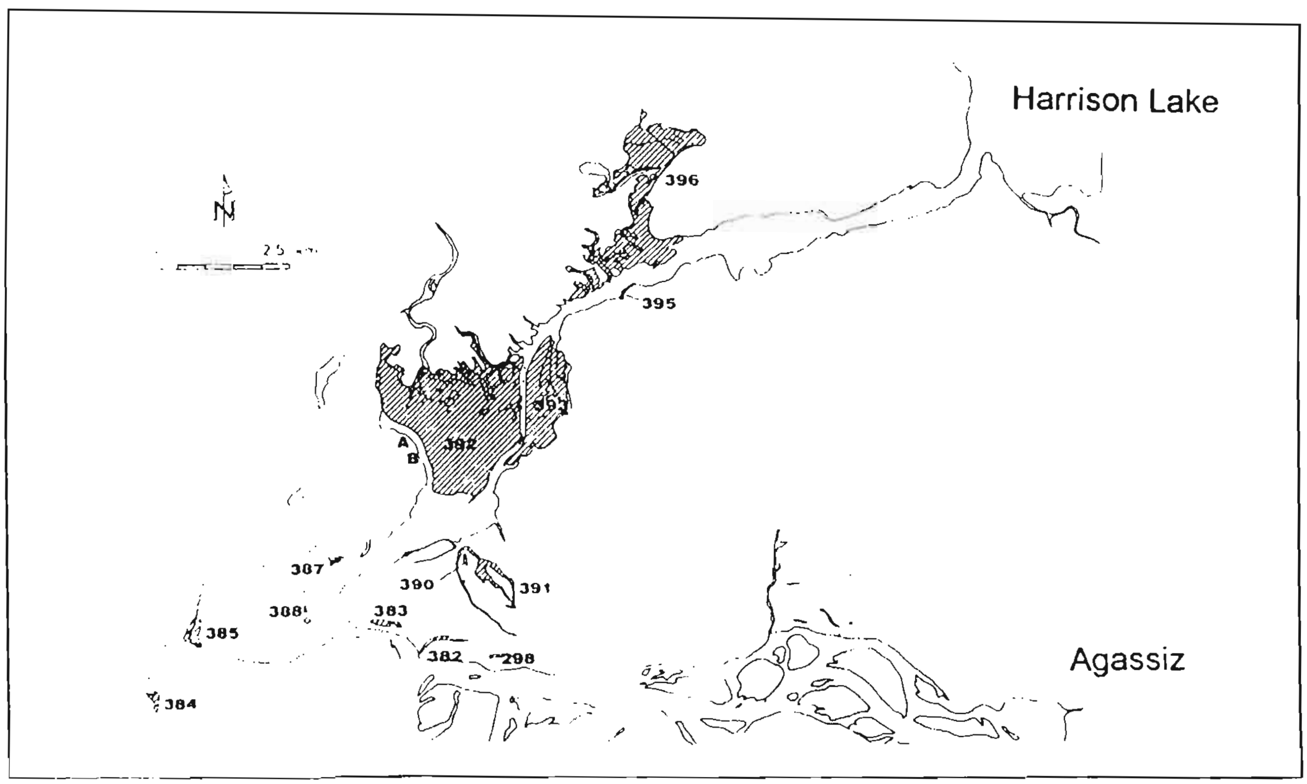

Figure 6. Present-day wetlands (321 ha) of the Lower Harrison River Reach. Numbers are wetland units occurring between line segments. Letters show the location of habitat restoration sites

CHAPTER 3

A Report Concerning the Provision

of "Baseline Maps and Quantitative Data

on Species Composition and Cover at Marshes

Developed at the Fraser River Estuary"[2]

by

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2 This section contains excerpts from the original report: Moody, A.I. 1993. A report concerning the provision of "Baseline maps and quantitative data on species composition and cover at marshes developed at the Fraser River Estuary". Prepared for Department of Fisheries and Oceans, West Vancouver Laboratory, by AIM Ecological Consultants Ltd., 25 p. On file in the West Vancouver Laboratory Library, Department of Fisheries and Oceans.

INTRODUCTION

AIM Ecological Consultants Ltd. was commissioned to provide baseline maps and quantitative data on species composition and cover at marshes developed in the Fraser River estuary. The objectives of the study were:

- to provide maps of developed marshes and nearby reference marshes for the establishment of baseline conditions for future fisheries habitat studies;
- to identify and quantify all plant species within the study areas;
- to identify, for the various vegetation communities:
  - elevations
  - boundaries; and,
- to demarcate sites using semi-permanent markers to allow future relocation.

The sites selected for study were as follows (see also Chapter 1, present document):

1. North Arm: AN1, AN2; and DI1, DI2
2. South Arm: DE1, DE2, DE3; and AC1, AC2, AC4
3. Queens Reach: PM1, PM3; and CN1, CN2

Sites on Queens Reach and AC4 in Annacis Channel of the South Arm were surveyed in August 1992. All other marshes were surveyed between August 15 and September 30, 1991.

APPROACH AND METHODOLOGY

At the outset of the study, field visits were undertaken with the scientific authority to confirm the study areas and their appropriate reference sites. Subsequently, permanent transects were established in each of the study areas. Each site contained one main transect, which was situated midway between the riparian zone and the river. Cross-transects were located perpendicular to the main transect, and crossed the entire width of the marsh from the riparian zone to the river edge. Surveying was conducted by professional surveyors (Arnold and Associates) with transect locations and elevations tied-in to nearby benchmarks (identified on the plans for each site). These benchmarks were identified according to the regional coordinate system and registered in the respective local government offices. Where possible, the centre-point of each plot was surveyed for elevation information. This was not feasible in all situations due to time constraints and the limited tidal window available for the surveying.

Quantification

Due to the variety of techniques which have been used, and because of the characteristics of the particular sites studied, we chose a technique which appeared commonly in a variety of studies throughout BC. This technique also seemed to offer the greatest advantages for long-term monitoring:

- Transects were established across the intertidal gradient at subjectively chosen sites, to represent all of the vegetation types mapped. Permanent locations were established by steel posts driven to within 15 cm of the substrate at the start and end points of the main transects, which were aligned along the shoreline.

- 1 m^2 quadrats were established at 5 m intervals. The centre point of the plot was marked by a wooden stake.

- Plant cover was described by Braun-Blanquet floristic association system (Table 1) as described in Shimwell (1971).

RESULTS

Mapping

The most recent colour aerial photographs available for this study of the lower Fraser River area were taken on September 3, 1986 (flight lines BCC533-535 and BCC589). Enlargements (46 x 46 cm) of these photos (resulting scale 1:6,000) were provided by the Habitat Protection Unit of Fisheries and Oceans Canada for our mapping purposes. More recent photographs (April 1989) had been taken too early in the growing season to be of use for vegetation mapping.

We had proposed to digitize the photos for computer mapping. However, site and photographic limitations prevented this approach. Photographic coverage was only available for the reference sites since most of the developed marshes had not been created at the time of the photography. Furthermore, four of the reference sites were situated on the south side of the river and shadows from the treed riparian fringe obscured a considerable portion of the narrow marsh band. The timing of the available photography (too early or too late in the growing season) resulted in the same concerns as identified for the stem-height and photo-point establishment above. Enlargement of the photographs had also resulted in a lack of image clarity and difficulties with interpretation. In most cases, due to the lack of suitable photography, total mapping coverage was not feasible with any degree of accuracy. As a consequence, the marsh mapping was quite subjective; based on photos when possible but largely on first-hand field experience and plot results, which were extrapolated between transects.

No attempt was made to classify community types or to apply ordination techniques based on the plot results largely because intent of the vegetation mapping was not for floristic analysis but rather to provide a generalized map of each study site which could be used for the associated fisheries studies. Field results were used to identify vegetation types according to the main species found in the plot. Where one or more species were relatively uniformly distributed, with a minimum Braun-Blanquet rating of 2, the vegetation type was named for the main component(s), for example, *Juncus balticus*. Where a single species dominated the plot (with a minimum Braun-Blanquet rating of 2), but included components of other species, the resulting vegetation type was referred to as a Mix, with the dominant species identified, for example, *Scirpus validus* Mix would have consisted of predominantly *Scirpus validus*, but may have included small amounts of *Eleocharis palustris*, *Carex lyngbyei* or other species. A number of the mixes could be considered transition zones between major species assemblages. A total of 24 vegetation groups (Table 2) were identified for all of the study sites; however, each site contained only 3 to 8 groups.

Graphic representation of the above information was achieved by overlaying the vegetation types on the elevation survey base maps (Figures 1-14). The maps were reduced in order to fit within the report, consequently they do not have a standard scale from site to site. Original full scale survey maps are on file at the West Vancouver Laboratory of DFO.

Survey

Geodetic elevation survey results (derived from closest recorded benchmarks) were presented separately on 14 plans at a scale of 1:500. Elevations ranged between a high of just over 1.9 m to a low of -0.83 m (Table 3). In general, the predominant vegetation distribution was between 0.2 and 1.0 m. A dramatic shift in marsh elevations occurred between the lower river marshes and those situated around the Port Mann area (Figure 15). The highest mean elevations were found at the CN1 transplant site (1.38 ± 0.27 m) where the range of surveyed elevations was from 0.95 to 1.94 m. The lowest mean elevation was at DE3 reference site (0.45 ± 0.19 m) with a range of -0.36 to 0.9 m.

Substantial variability was encountered in the distribution of vegetation types both between sites and within sites. Although it was outside the terms of reference for this study, we undertook a limited analysis of some vegetation groups and the elevation range across which they occurred at the various study areas (Table 4). General descriptions of the sites and their vegetation types are provided in the following section. A complete list of all the plot results is given in the original contract report (Moody 1993).

<u>AN1 Transplant</u>

The created marsh at Angus Lands (Fraser River Park) was, on average, higher in elevation (0.79 m vs. 0.7 m) and had greater variability (s.d. 0.31 vs. 0.25) than did AN2 reference. Substrates at this site were variable, from a heavy clay at the east end of the main transect, switching to sand at the higher elevation sites and then to an eroded sand/rubble base near the west end of the transect. It is likely that substrate type had as much influence as elevation in determining the growth of marsh species in this site. At this site, *Carex lyngbyei* thrived at the eastern end of the site in clay soils at a mean elevation of 0.9 m (range 0.68 to 1.07 m). *Juncus balticus* occurred over a broader range than *C. Lyngbyei*, from 0.12 to 1.26 m in elevation (mean 0.83 m). In sandy substrates, growth of all species was impoverished and chlorotic.

AN2 Reference

At this site, *Carex lyngbyei* and *Scirpus americanus*, or *S. validus*, dominated the lower elevations of the site, up to approximately 0.6 m. Above 0.6 m, *Carex lyngbyei* was found in association with *Juncus balticus*, and at approximately 0.8 m, *Potentilla pacifica* became a major element in the *Carex/Juncus* mix.

DI1 Transplant

The created marsh at Deering Island was on average lower in elevation (0.63 vs. 0.78 m) but had less variability (s.d. 0.15 vs. 0.39) than did DI2 reference. At this site, *Scirpus validus* dominated at the lower elevations at the eastern end of the site (0.2 to 0.7 m) whereas *Carex lyngbyei* occurred at the western end of the site at approximately 0.5 to 1 m in elevation. The original plantings of sedge plugs were still in evidence and in numerous cases rhizomes were exposed above the sediment surface.

DI2 (high) and DI2 (low) Reference

Virtually pure *Carex lyngbyei* was found in elevations from 0.4 to 0.9 m while the *Carex/Juncus* mix occurred from 0.6 to 1.09 m (low site). Above the (approximately) 1 m elevation (high site) various grass species were found in association with *Juncus balticus*, whereas *Carex lyngbyei* faded from the association. This area was very dissected and in a number of instances showed signs of marsh clumps, one to several square metres in size having been undercut and dropped to a lower level. These clumps had in some cases continued to grow where the remainder of the bench was situated at a higher level. The erosion had resulted in deep incisions into the marsh and the formation of multiple benches of growth.

AC2 Transplant

The created marsh at Patrick Bay was on average lower in elevation (0.5 vs. 0.7 m) and had a greater variability (s.d. 0.47 vs. 0.27) than did the nearby AC1 reference site. The vegetation at this site was highly variable with numerous species found in a single plot. The marsh was distributed in a narrow band around the perimeter of the bay. Small patches of *Carex lyngbyei* were found in elevations from 0.4 to 1.2 m but nowhere was the species found as a dominant element.

AC1 Reference

This marsh formed a relatively uniform shelf, up to 40 m wide, along the northern shore of Annacis Island. It consisted predominantly of *Carex lyngbyei* (at elevations 0.5 to 1.1 m) with

Eleocharis palustris as a significant element at lower elevations (0.06 to 0.6 m) and *Juncus supiniformis* and *Equisetum fluviatile* (among others) found in association above 0.8 m. This area was quite uniform and exhibited a relatively gentle, undisturbed slope from the riparian zone down to the lower edge of the vegetated zone.

DE2 Reference

This marsh consisted of a relatively high elevation (mean 0.85 m), narrow bench along the Richmond shore of the Gravesend Reach of the Fraser River. The vegetation consisted predominantly of a mix of various species (mostly *Carex lyngbyei, Juncus supiniformis, J. balticus* and various species of grass). Pure stands of *Carex lyngbyei* were not common in this area.

DE1 Transplant

This site was a very narrow platform, fronted by riprap, along the northern perimeter of Deas Slough. The elevations were very uniform with a mean of 0.73 m, ranging from 0.5 to 1.0 m, and less variability than at either reference site (DE2 and DE3). The vegetation was also very uniform, dominated primarily by *Juncus supiniformis*.

DE3 Reference

DE3 reference was situated immediately west of the DE1 transplant site. Elevations in this area were much lower (mean of 0.45 m) than those of the transplant site (mean of 0.73 m) and had greater variability than that of the transplant site (s.d. of 0.26 vs. 0.12). The vegetation in this area ranged from *Scirpus validus* situated in a band at the forefront of the marsh, to a *Carex lyngbyei* mix and pure *C. lyngbyei* pockets in the central portions of the marsh.

AC4 Transplant

AC4 consisted of a constructed bench approximately 30 m wide, which sloped gradually from the natural riparian zone toward a riprap berm. The created marsh was on average higher in elevation (0.82 vs. 0.67 m) but had less variability (s.d. 0.22 vs. 0.27) than did the nearby AC1 reference. The lowest elevations were found at the western end of the site. The upper elevations were dominated by a mix of *Carex lyngbyei, Scirpus cyperinus, Phalaris arundinacea, Juncus supiniformis* and *J. effusus*. Lower elevations were dominated by *Juncus* species which tapered off to a zone of *Ranunculus* and *Isoetes* at the very lowest elevations. The lower elevation sites showed extensive waterfowl use of the 1991 plantings.

PM1 Transplant

This site was prepared for transplanting of sedges, however, vegetation was established naturally. The marsh at site PM1 was on average higher in elevation (1.35 vs. 1.12 m) and had greater variability (s.d. 0.23 vs. 0.15 m) than did the PM3 reference site. Many areas showed significant erosion and limited vegetative cover. The site was heavily incised by wave action to the point where some survey points were lost due to slumping in the short period of time between the establishment of the transects and the survey. The area was partially unvegetated, and partially colonized by *Scirpus* and *Juncus* species. A band of *Phalaris arundinacea* was situated adjacent to the riparian zone.

PM3 Reference

The PM3 reference site consisted of a narrow strip of marsh, upstream of site PM1. It was situated on the north bank of the Fraser River between the mouth of the Coquitlam River and the Port Mann bridge. The riparian zone produced heavy growth in this area, resulting in substantial overhanging tree and shrub growth in the upper marsh zone. The marsh was quite diverse with a mixture of *Carex lyngbyei*, *Juncus* spp. and various *Scirpus* species.

CN1 Transplant

The created marsh at the CN1 transplant site was marginally higher in elevation 1.38 m vs. 1.33 m) and had greater variability (s.d. 0.27 vs. 0.15) than did the CN2 reference site. The standing crop of the transplant marsh was among the greatest observed during this survey with *Typha latifolia* over 2 m in height, *Juncus effusus* and *Scirpus cyperinus* both well over 1 m in height. A dense patch of *Lythrum salicaria* and *Salix* spp. was located in a small pocket of high marsh.

CN2 Reference

The CN2 reference site consisted of a narrow strip of marsh, immediately upstream of the CN1 transplant site. It was very uniform in elevation and consisted largely of *Phalaris arundinacea* mixes. Sedges were only found in limited distribution at the forefront of the marsh.

REFERENCES

Moody, A.I. 1993. A report concerning the provision of "Baseline maps and quantitative data on species composition and cover at marshes developed at the Fraser River Estuary". Prepared for Department of Fisheries ad Oceans, West Vancouver Laboratory, by AIM Ecological Consultants Ltd. 25 p.

Shimwell, D.W. 1971. The Description and Classification of Vegetation. University of Washington Press, Seattle, Washington. 322 p.

Table 1. The Braun-Blanquet cover abundance scale (Shimwell 1971).

% Cover	Braun-Blanquet
>75	5
50 to 75	4
25 to 50	3
5 to 25	2
<1 to 5	1
few, small cover*	0.1 (+,r)

* To facilitate data summation in the spreadsheet program (Microsoft Excel), two categories of the Braun-Blanquet Cover-Abundance Scale which are usually assigned non-numerical values (+,r) were combined into a numerical value of 0.1. These categories were rated in the Braun-Blanquet Cover-Abundance Scale as follows:

\+ few, with small cover

r solitary, with small cover

Table 2. Summary of vegetation types identified for the surveyed marshes.

Agrostis Mix
Carex aquatilis
Carex (Juncus) Mix
Carex lyngbyei
Carex (Eleocharis) Mix
Eleocharis (Carex) Mix
Eleocharis palustris
Graminae/Equisetum Mix
Isoetes/Ranunculus Mix
Juncus balticus
Juncus/Carex Mix
Juncus effusus
Juncus/Graminae Mix
Juncus spp.
Lythrum salicaria
Phalaris Mix
Phalaris/ Juncus Mix
Phalaris/Typha Mix
Phalaris/Equisetum Mix
Phalaris/Graminae/Lythrum Mix
Scirpus spp.
Scirpus validus Mix
Scirpus/Carex Mix
Typha Mix
Unvegetated

Table 3. Elevations of surveyed lower Fraser River marshes.

Site	Mean elevation (m)	Standard deviation (s.d.)	Maximum (max.)	Minimum (min.)	Number of stations for elevation measurements
AN1 Transplant	0.79	0.31	1.26	0.21	34
AN2 Reference	0.70	0.25	1.08	0.04	25
DI1 Transplant	0.63	0.15	0.88	0.32	21
DI2 Reference	0.78	0.39	1.39	-0.18	47
AC2 Transplant	0.50	0.43	1.26	-0.83	70
AC1 Reference	0.67	0.27	1.17	0.14	80
AC4 Transplant	0.82	0.22	1.18	0.47	35
DE2 Reference	0.85	0.25	1.44	0.33	30
DE1 Transplant	0.73	0.12	1.06	0.50	57
DE3 Reference	0.45	0.26	0.90	-0.36	77
PM1 Transplant [1]	1.35	0.23	1.89	0.64	32
PM3 Reference	1.12	0.15	1.48	0.76	56
CN1 Transplant	1.38	0.27	1.94	0.95	29
CN2 Reference	1.33	0.15	1.67	0.95	72

[1] This site was prepared for transplanting of sedges, however, vegetation was established naturally.

Table 4. Elevation range (m) of plant groups according to site.

Site		*Carex lyngbyei* Mix	*Carex lyngbyei*	*Carex/* Graminae	*Juncus balticus*	Graminae	*Typha latifolia* Mix	*Juncus* spp.
AN1 Transplant	mean	0.38	0.94	0.83	0.84	-	-	-
	s.d.	-	0.14	-	0.32	-	-	-
	min.	0.38	0.68	0.83	0.12	-	-	-
	max.	0.38	1.1	0.83	1.26	-	-	-
AN2 Reference	mean	0.41	0.60	0.91	0.88	-	-	-
	s.d.	0.53	0.24	0.18	0.19	-	-	-
	min.	0	0.06	0.74	0.46	-	-	-
	max.	1	1.08	1.10	1.18	-	-	-
DI1 Transplant	mean	0.49	0.68	-	-	-	-	0.88
	s.d.	0.20	0.16	-	-	-	-	-
	min.	0.10	0.35	-	-	-	-	0.88
	max.	0.77	0.9	-	-	-	-	0.88
DI2 Reference	mean	-	0.62	0.92	0.86	0.31	-	-
	s.d.	-	0.72	-	0.42	1.60	-	-
	min.	-	-0.85	0.92	-0.13	-0.84	-	-
	max.	-	1.41	0.92	1.49	1.50	-	-
AC2 Transplant	mean	0.19	-	0.72	1.04	1.15	-	0.69
	s.d.	-	-	0.76	0.14	0.13	-	0.28
	min.	0.19	-	0.18	0.81	1.03	-	0.17
	max.	0.19	-	1.26	1.20	1.29	-	1.26
AC1 Reference	mean	0.63	0.59	0.80	0.97	0.63	-	0.97
	s.d.	0.30	0.27	0.14	-	-	-	0.12
	min.	0.05	0.16	0.52	-	-	-	0.80
	max.	1.19	1.09	1.08	-	-	-	1.17
AC4 Transplant	mean		1.02					0.77
	s.d.		0.06					0.21
	min.		0.97					0.47
	max.		1.06					1.18
DE2 Reference	mean	0.90	0.81	0.80	0.91	1.02	-	0.82
	s.d	0.06	0.15	1.51	0.32	0.14	-	0.26
	min.	0.83	0.67	0.62	0.34	0.92	-	0.33
	max.	0.96	1	1.06	1.44	1.12	-	1.35
DE1 Transplant	mean	-	-	-	-	-	-	0.73
	s.d.	-	-	-	-	-	-	0.12
	min.	-	-	-	-	-	-	0.03
	max.	-	-	-	-	-	-	0.71
DE3 Reference	mean	0.47	0.40	0.54	-	-	0.38	0.92
	s.d.	0.29	0.29	0.27	-	-	-	0.29
	min.	-0.36	-0.01	-0.29	-	-	0.38	0.71
	max.	1	0.91	0.84	-	-	0.38	1.12
CN1 Transplant	mean	-	-	-	-	-	1.29	1.14
	s.d.	-	-	-	-	-	0.20	0.34
	min.	-	-	-	-	-	0.96	0.95
	max.	-	-	-	-	-	1.69	1.86
CN2 Reference	mean	-	1.2	-	-	1.38	1.27	1.03
	s.d.	-	0.23	-	-	0.13	0.15	0.1
	min.	-	1.03	-	-	1.17	0.97	0.95
	max.	-	1.36	-	-	1.67	1.56	1.15
PM3 Reference	mean	-	1.08	-	-	-	-	1.08
	s.d.	-	0.05	-	-	-	-	0.1
	min.	-	0.99	-	-	-	-	0.88
	max.	-	1.16	-	-	-	-	1.29
PM1 Transplant[1]	mean	-	-	-	-	-	-	1.35
	s.d.	-	-	-	-	-	-	0.09
	min.	-	-	-	-	-	-	1.17
	max.	-	-	-	-	-	-	1.49

[1] This site was prepared for transplanting of sedges, however, vegetation was established naturally.

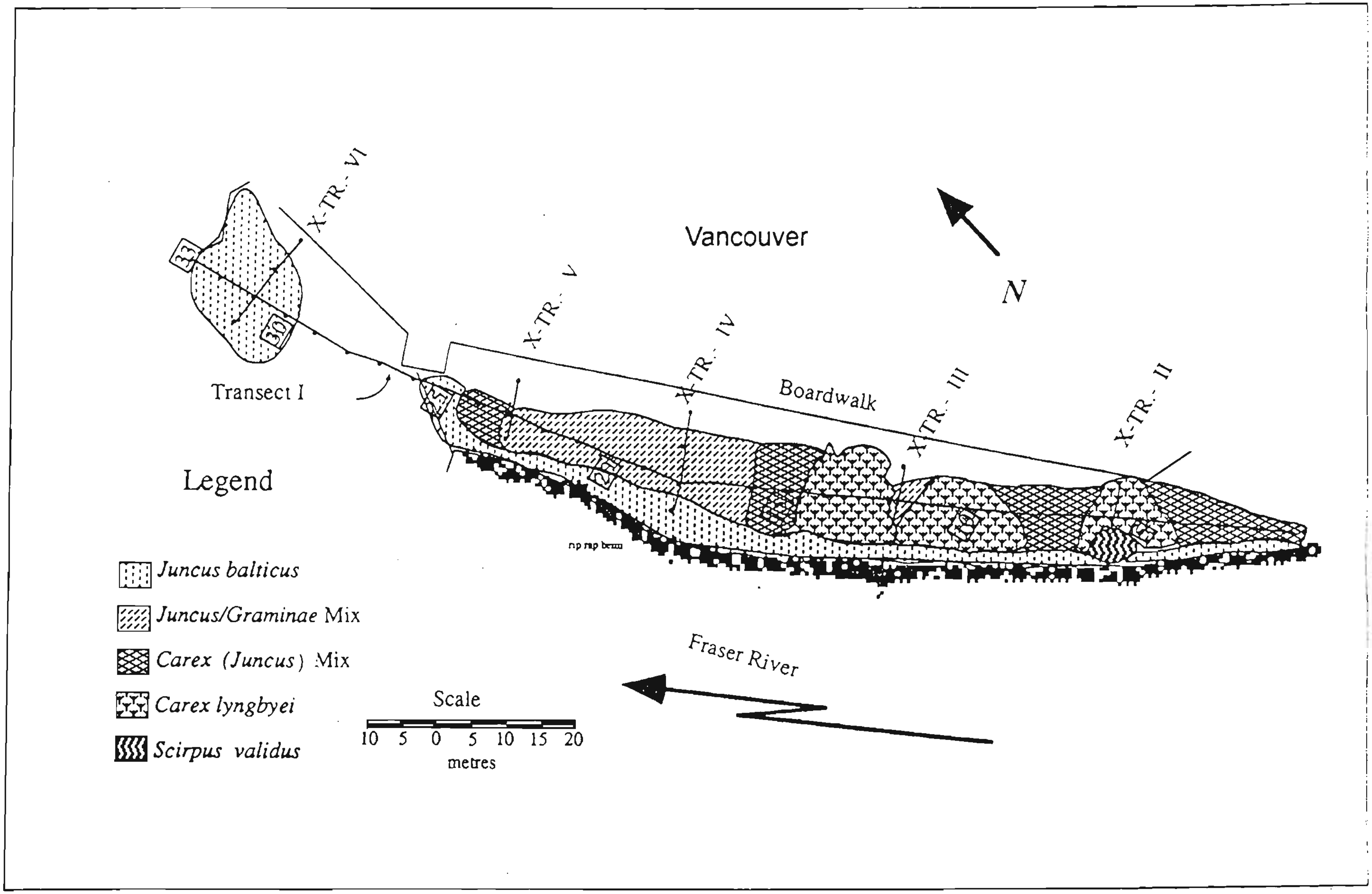

Figure 1. Generalized vegetation map for site AN1 (see Table 1, Chapter 1 for geo-reference)

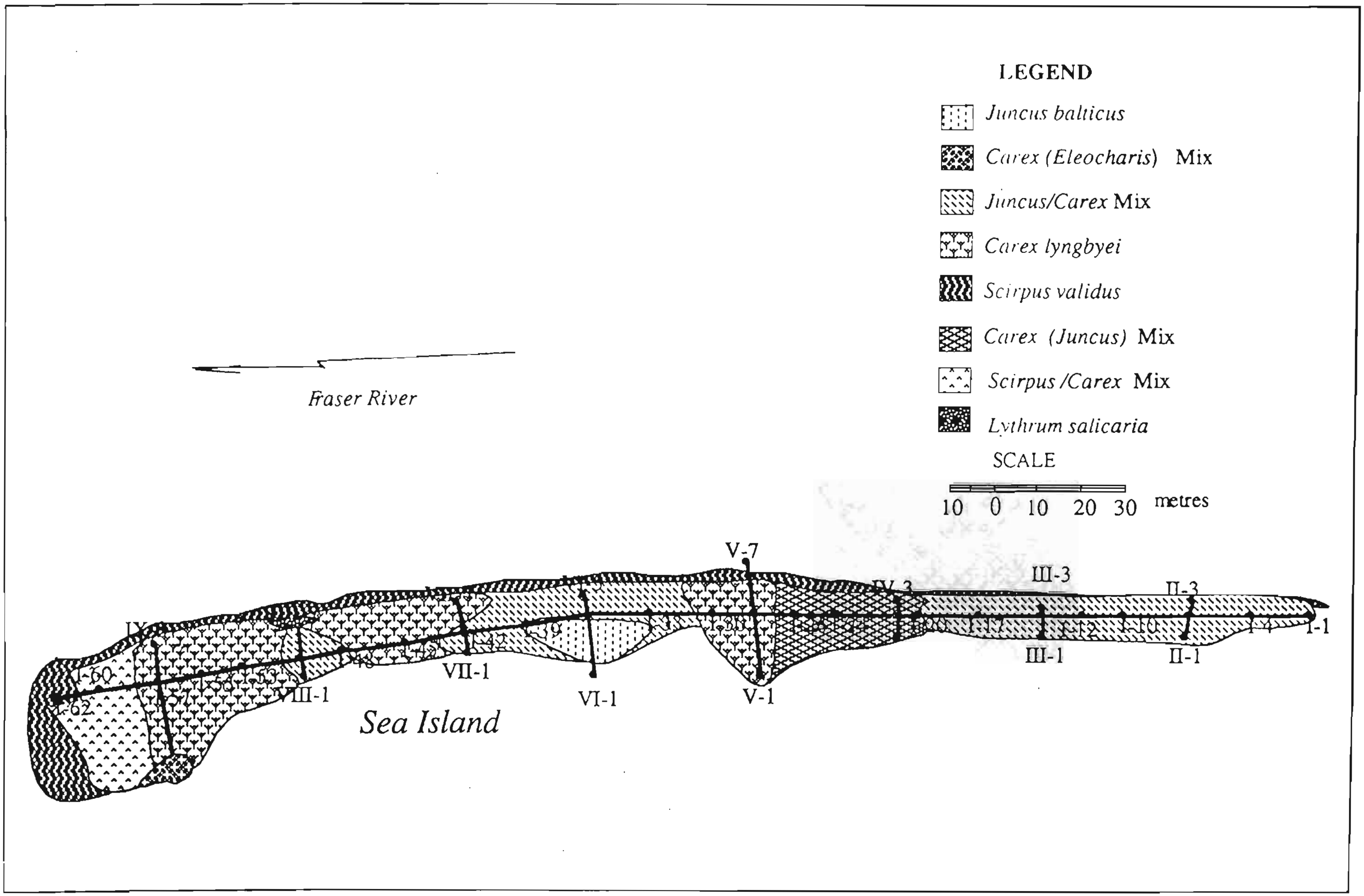

Figure 2. Generalized vegetation map for site AN2 (see Table 1, Chapter 1 for geo-reference).

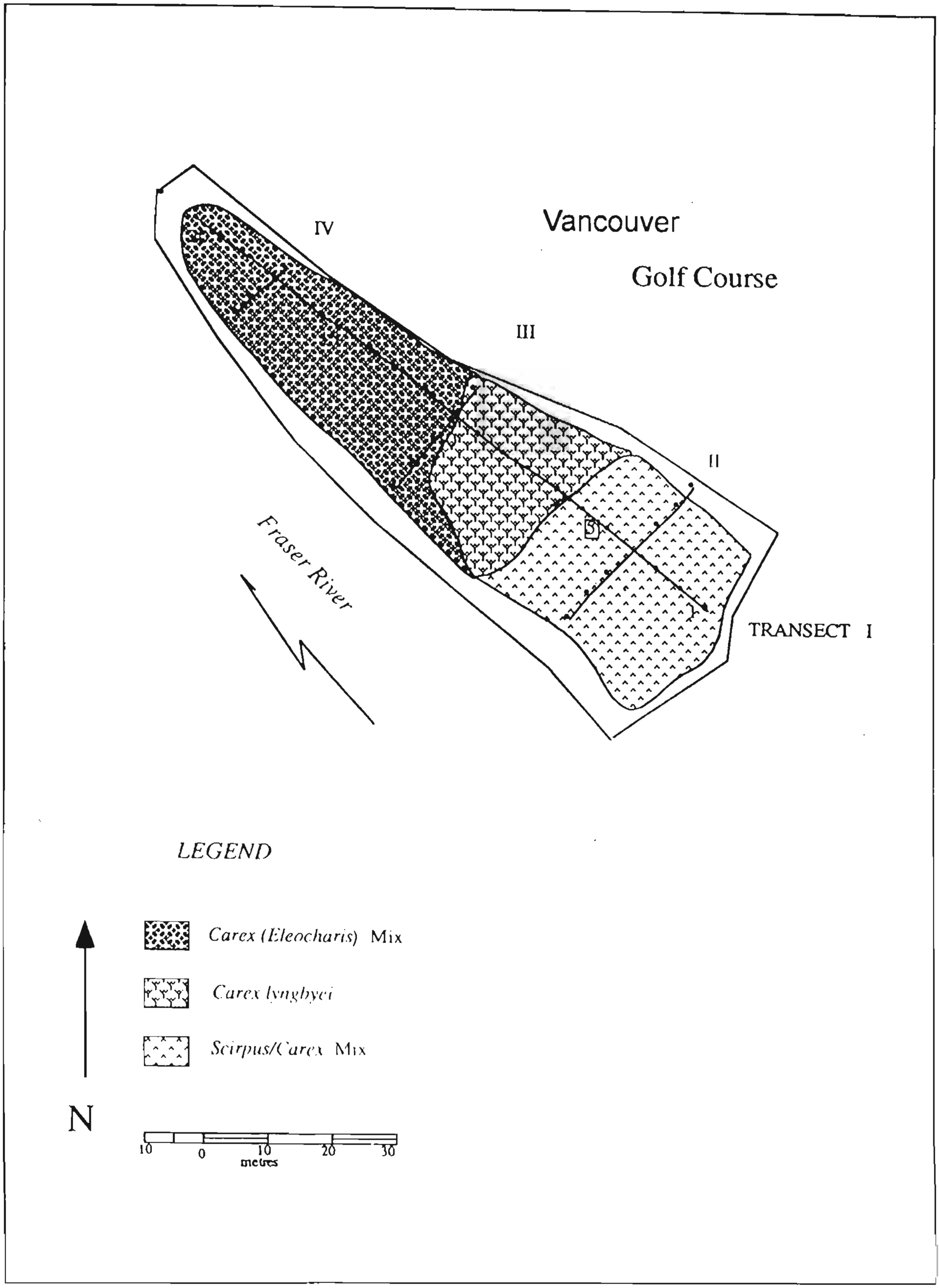

Figure 3. Generalized vegetation map for site DI1 (see Table 1, Chapter 1 for geo-reference).

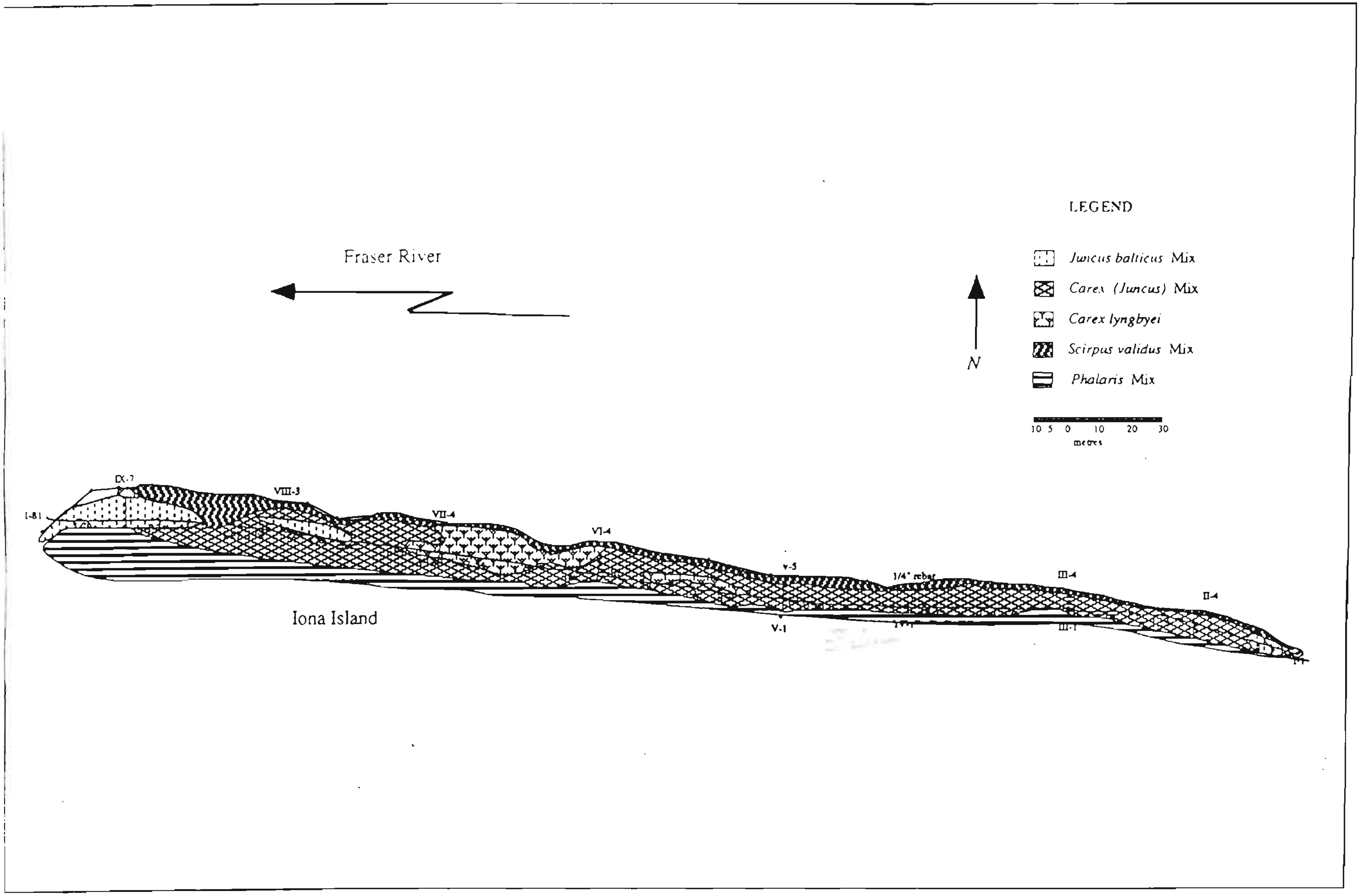

Figure 4. Generalized vegetation map for site DI2 (see Table 1, Chapter 1 for geo-reference)

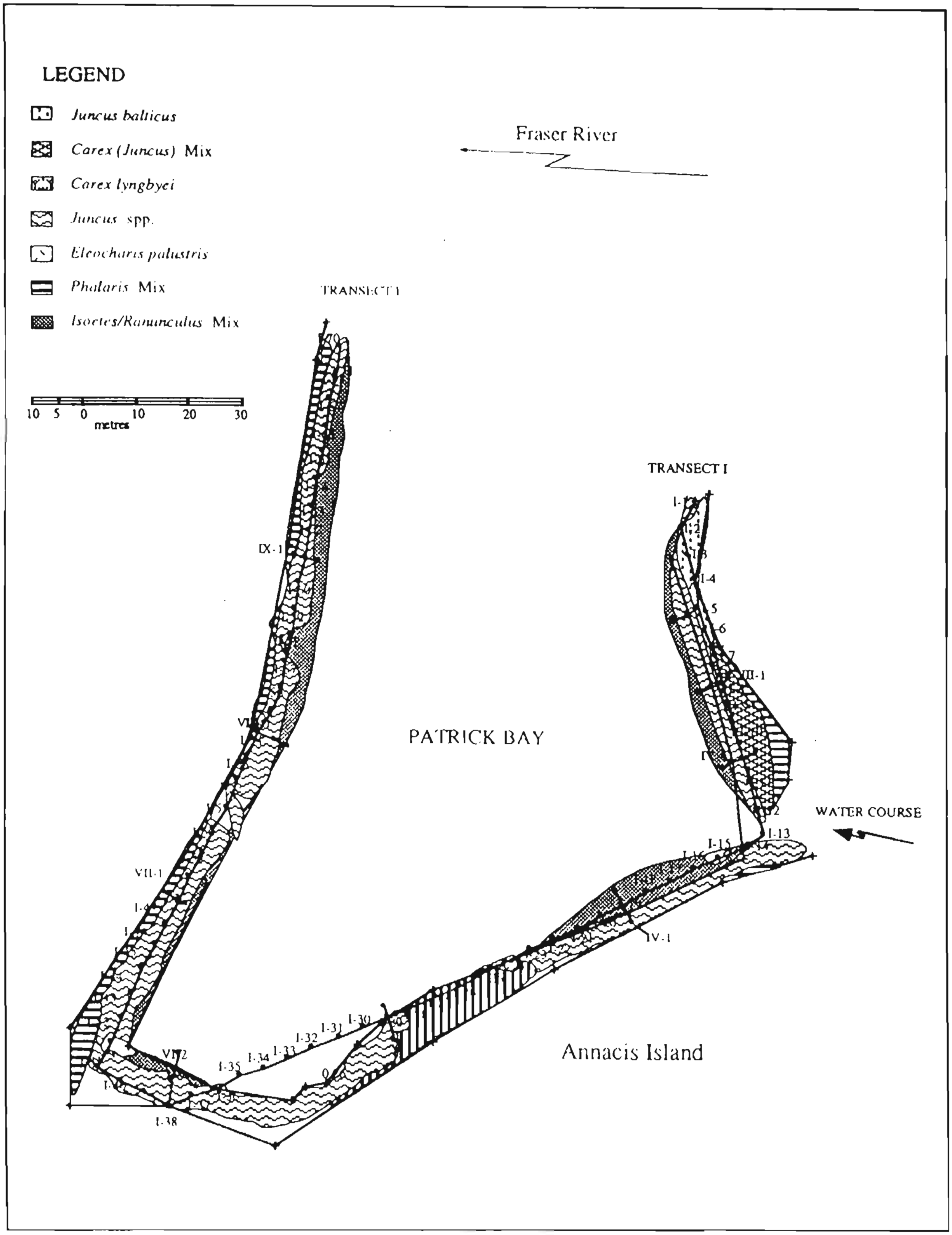

Figure 5. Generalized vegetation map for site AC2 (see Table 1, Chapter 1 for geo-reference).

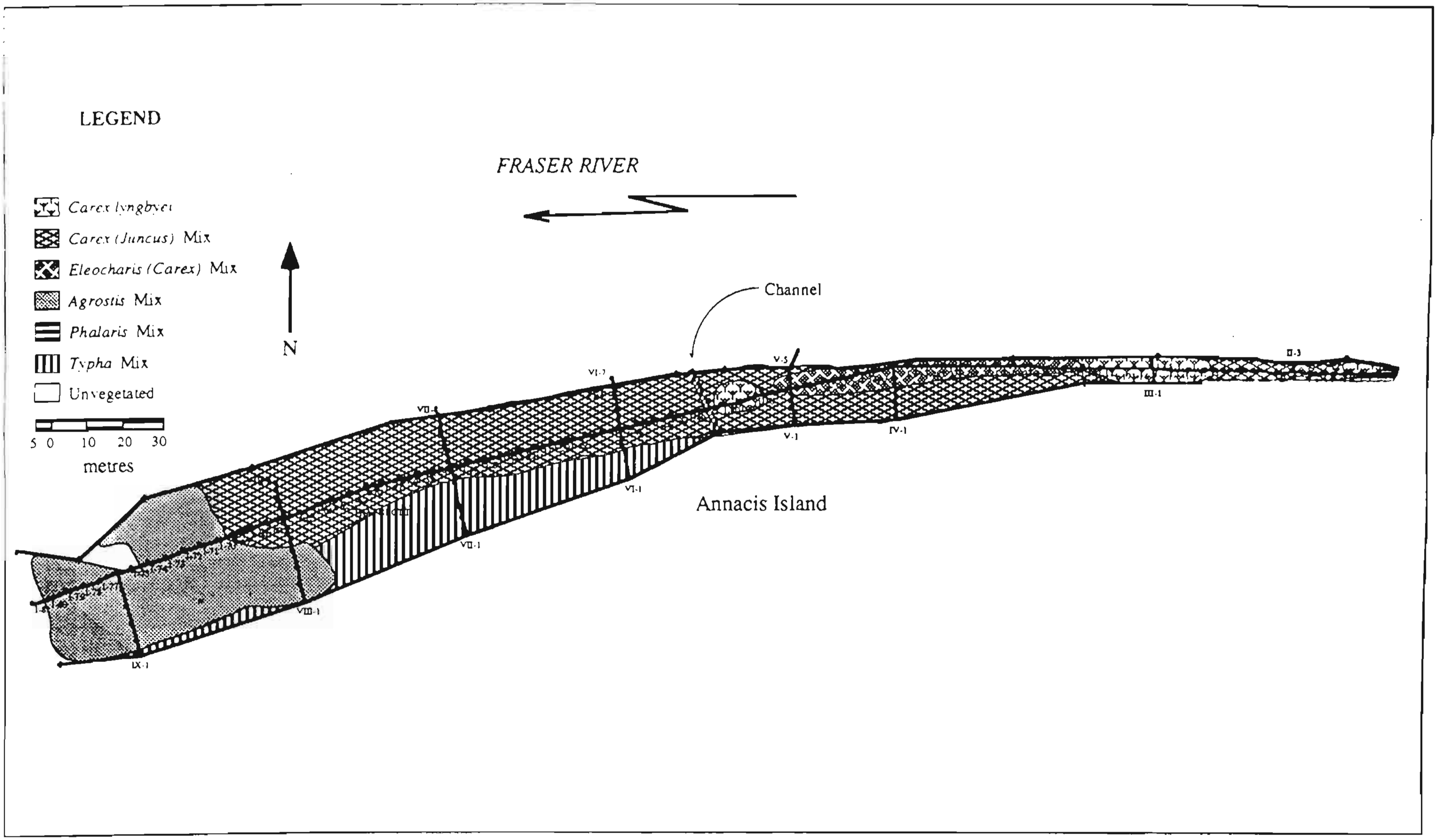

Figure 6. Generalized vegetation map for site AC1 (see Table 1, Chapter 1 for geo-reference).

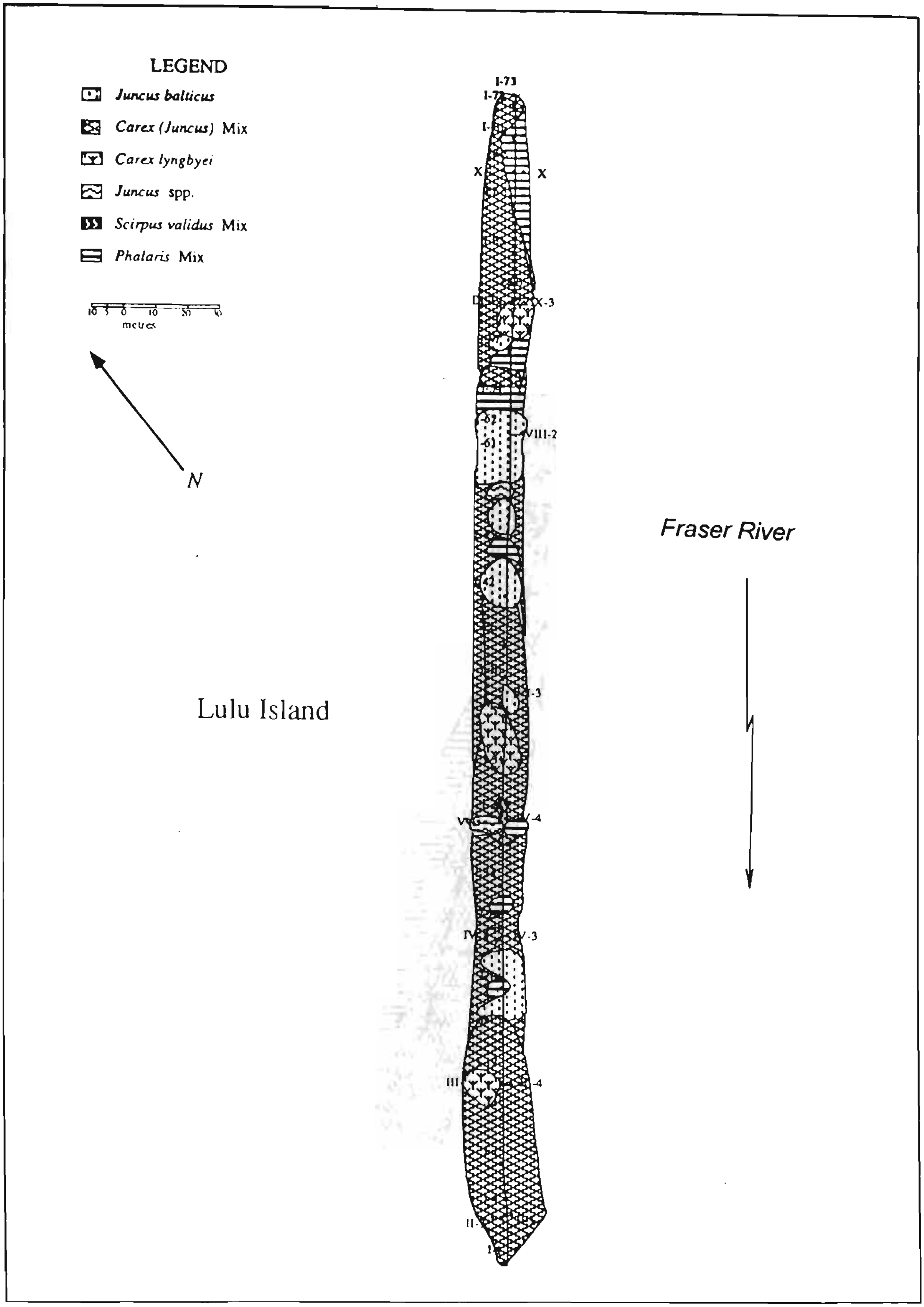

Figure 7 Generalized vegetation map for site DE2 (see Table 1, Chapter 1 for geo-reference).

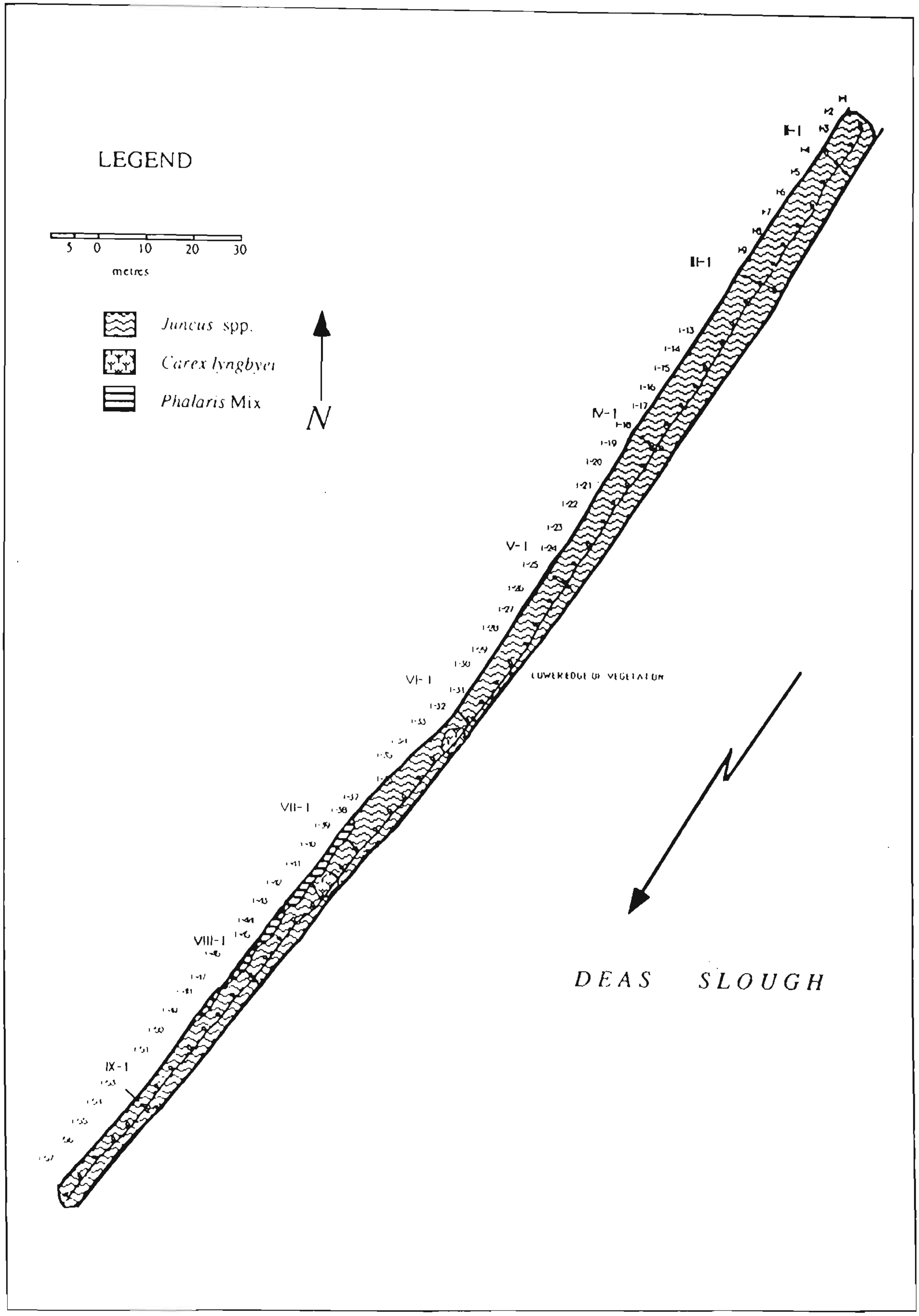

Figure 8 Generalized vegetation map for site DE1 (see Table 1, Chapter 1 for geo-reference)

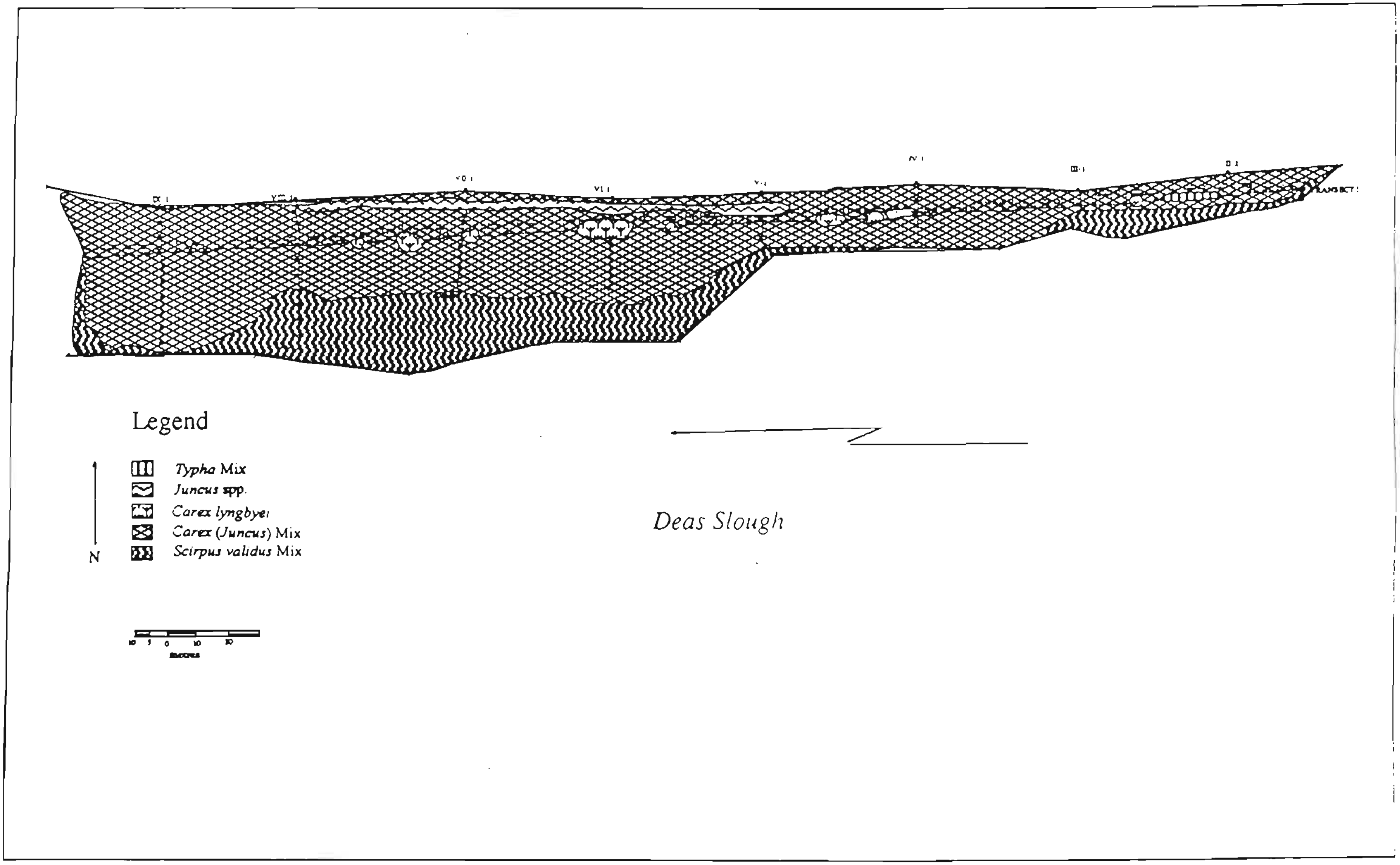

Figure 9. Generalized vegetation map for site DE3 (see Table 1, Chapter 1 for geo-reference)

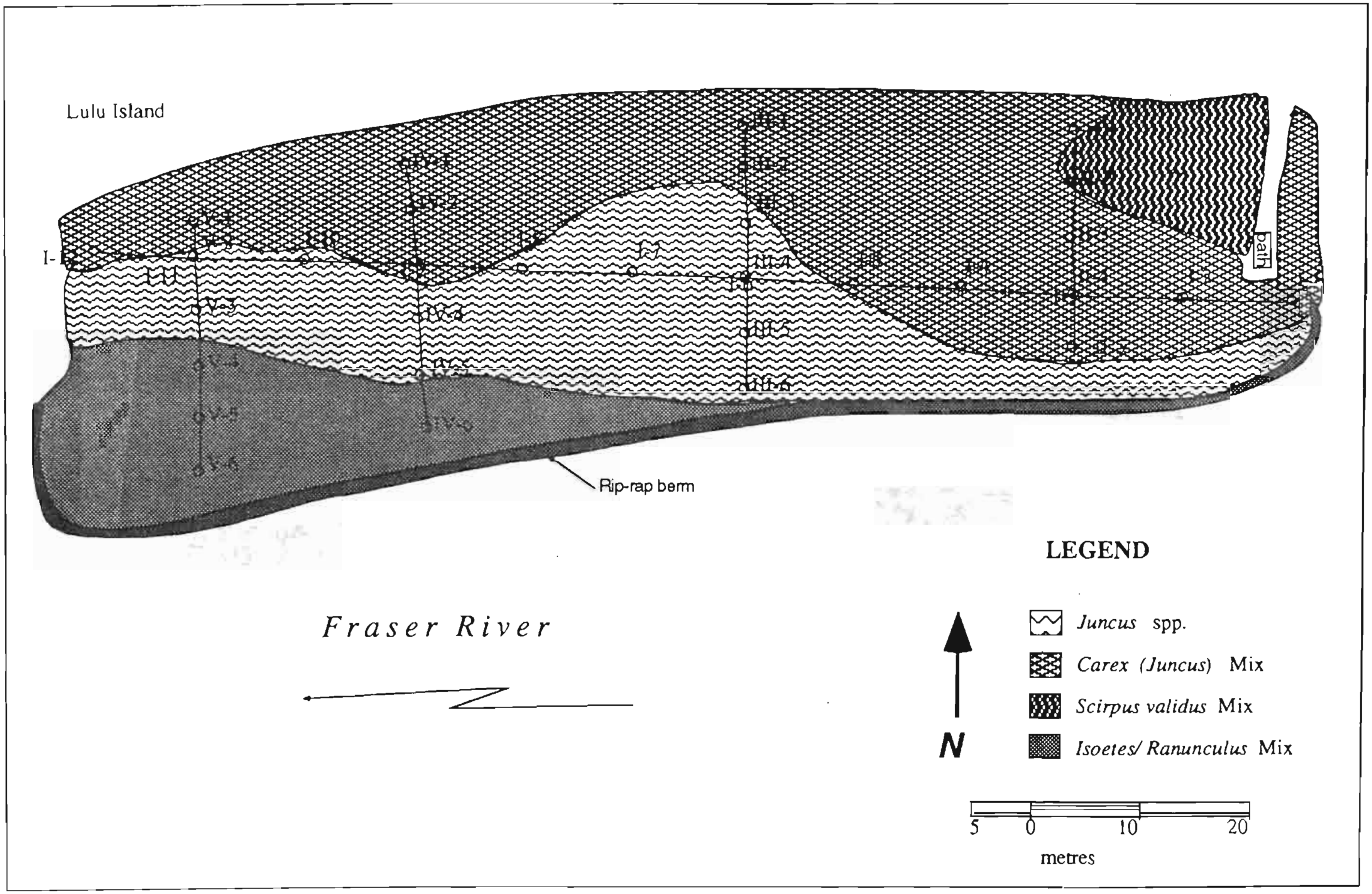

Figure 10. Generalized vegetation map for site AC4 (see Table 1, Chapter 1 for geo-reference).

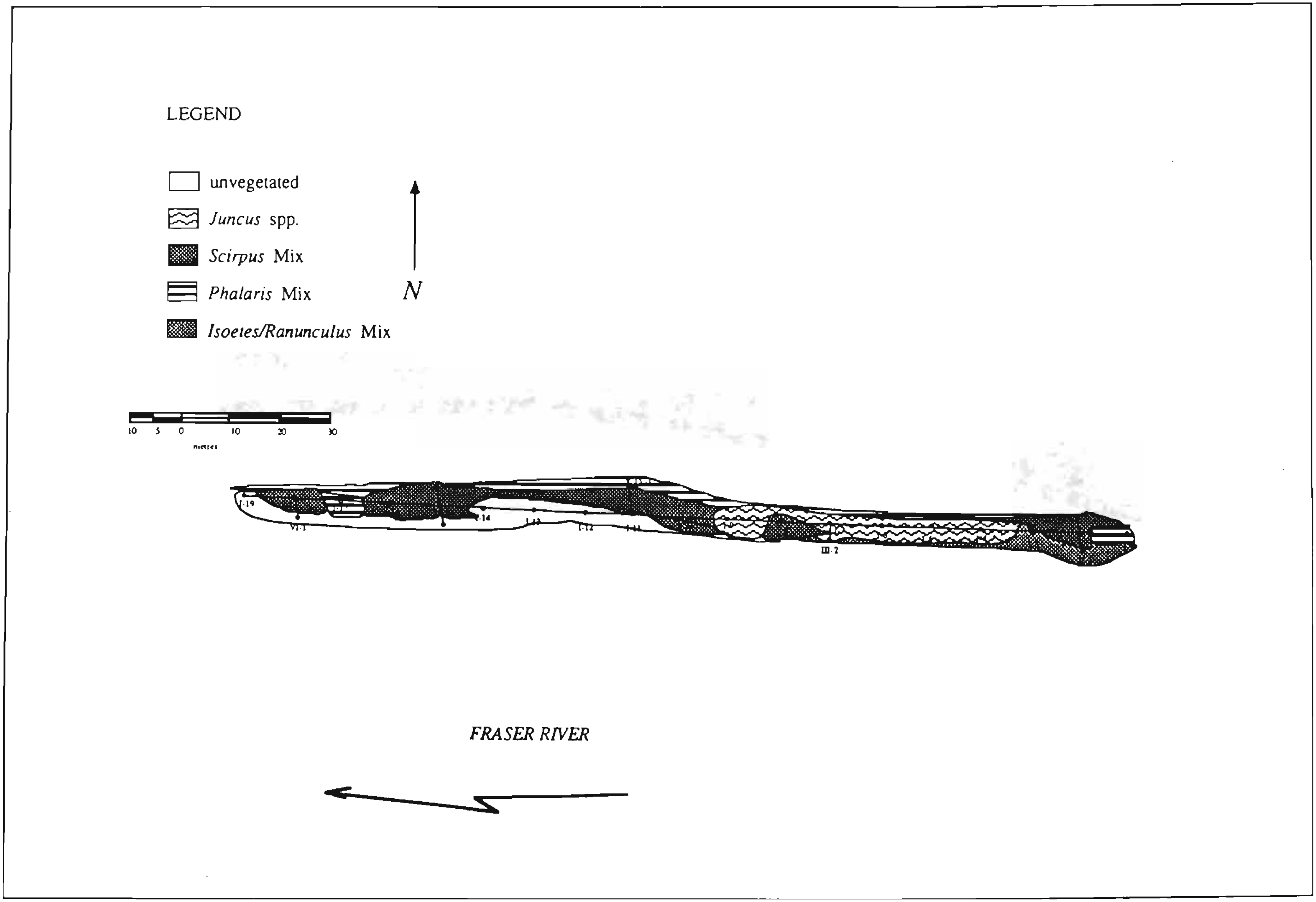

Figure 11. Generalized vegetation map for site PM1 (see Table 1, Chapter 1 for geo-reference).

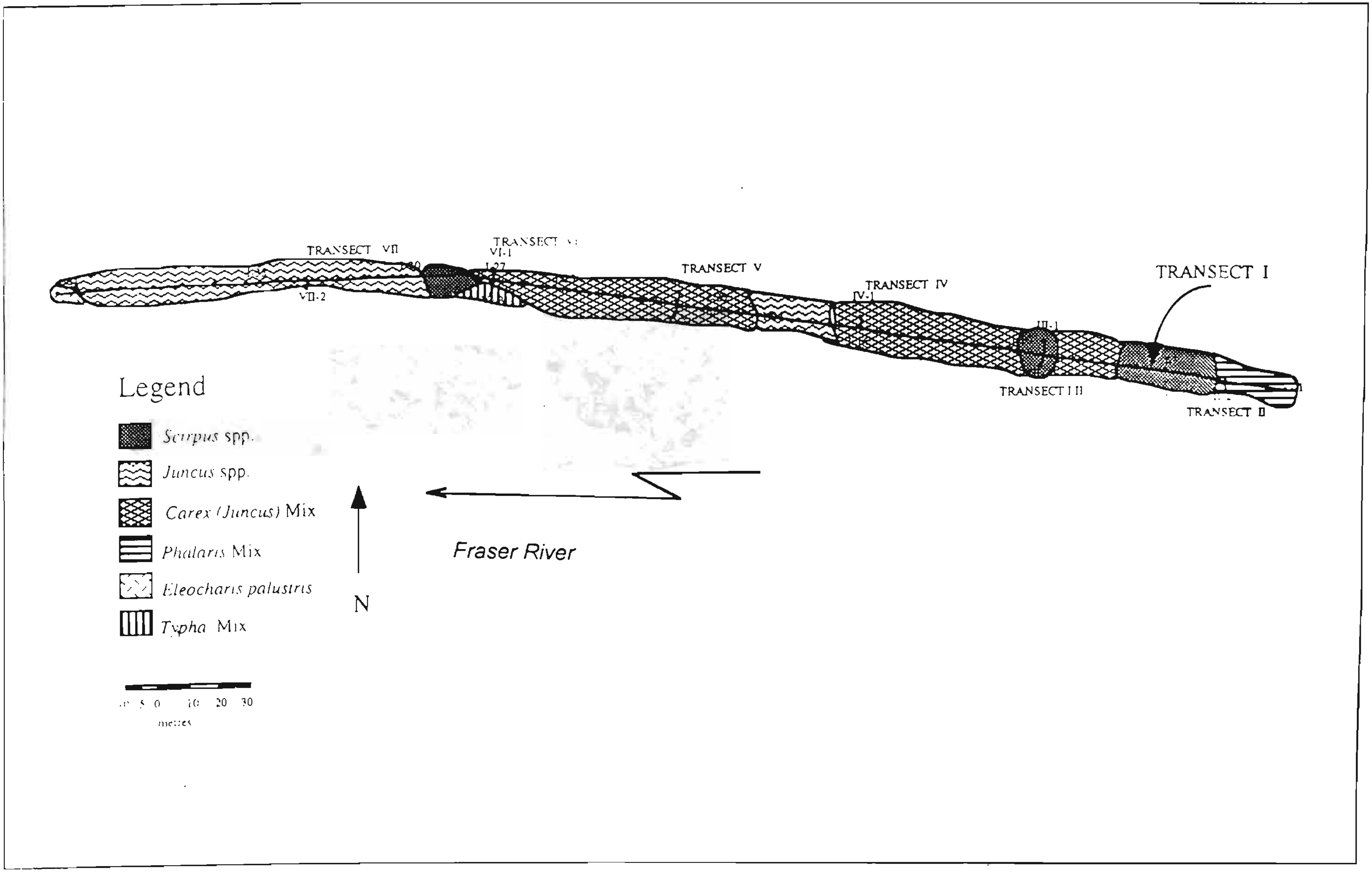

Figure 12. Generalized vegetation map for site PM3 (see Table 1, Chapter 1 for geo-reference).

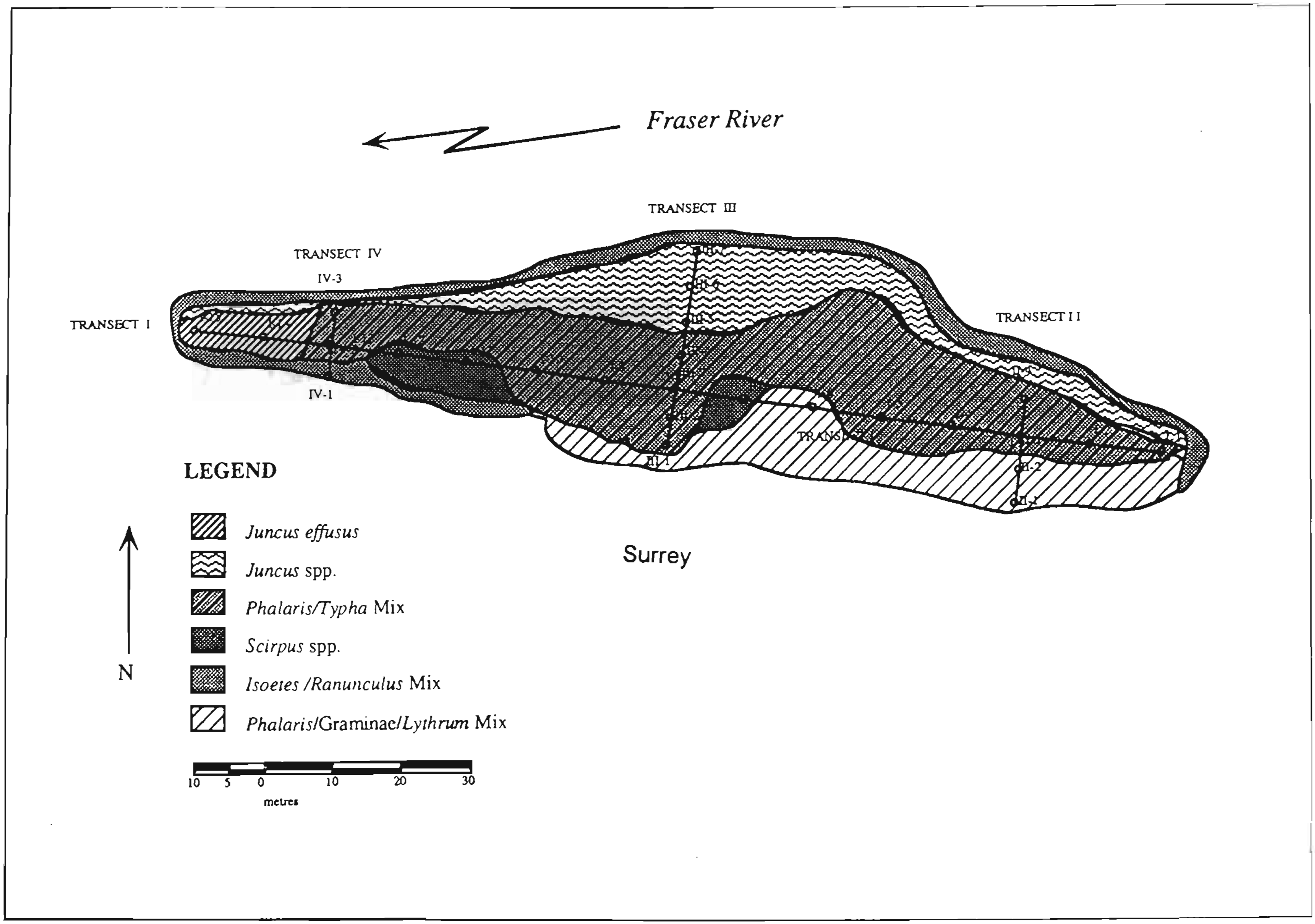

Figure 13. Generalized vegetation map for site CN1 (see Table 1, Chapter 1 for geo-reference).

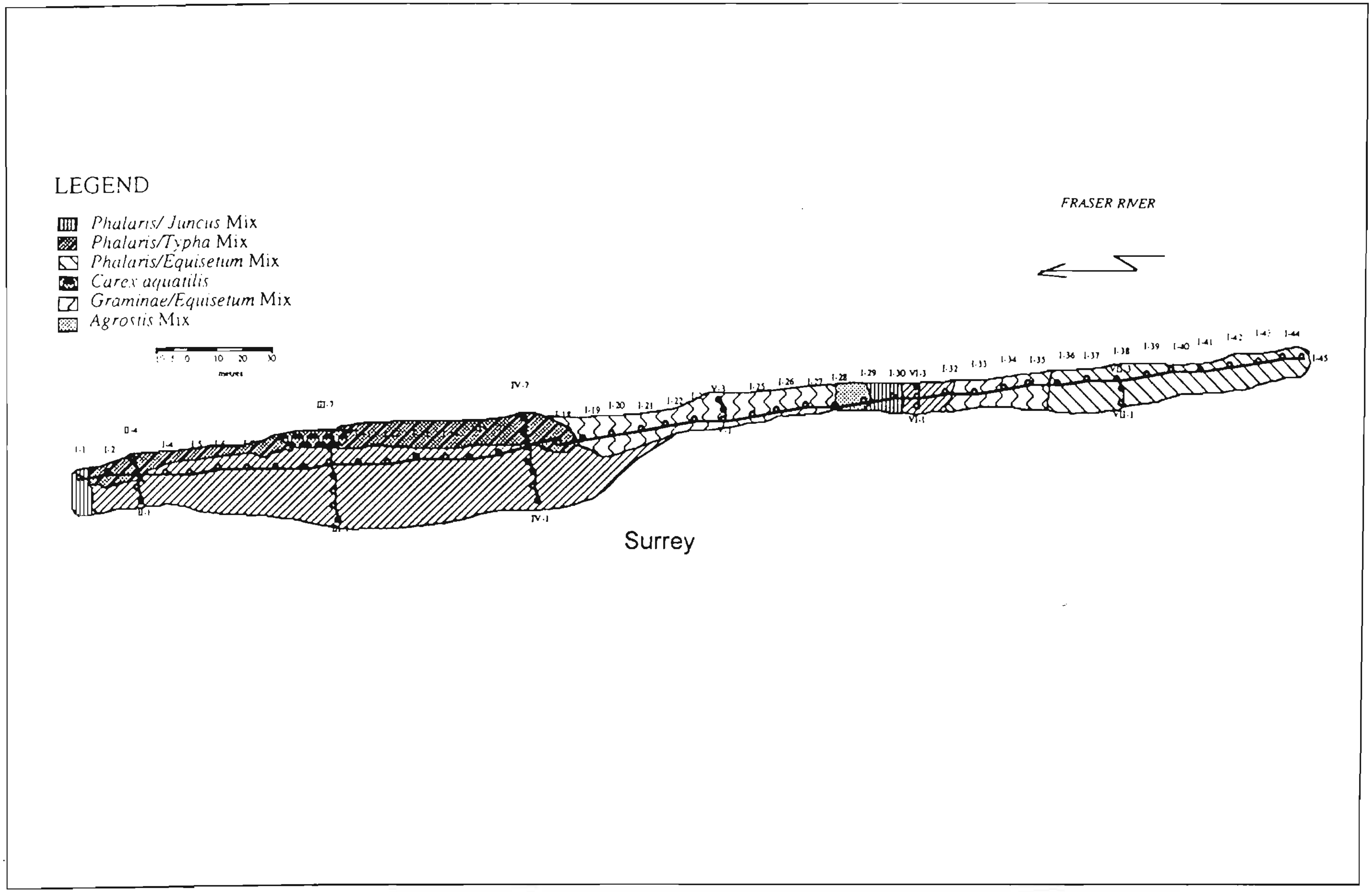

Figure 14. Generalized vegetation map for site CN2 (see Table 1, Chapter 1 for geo-reference).

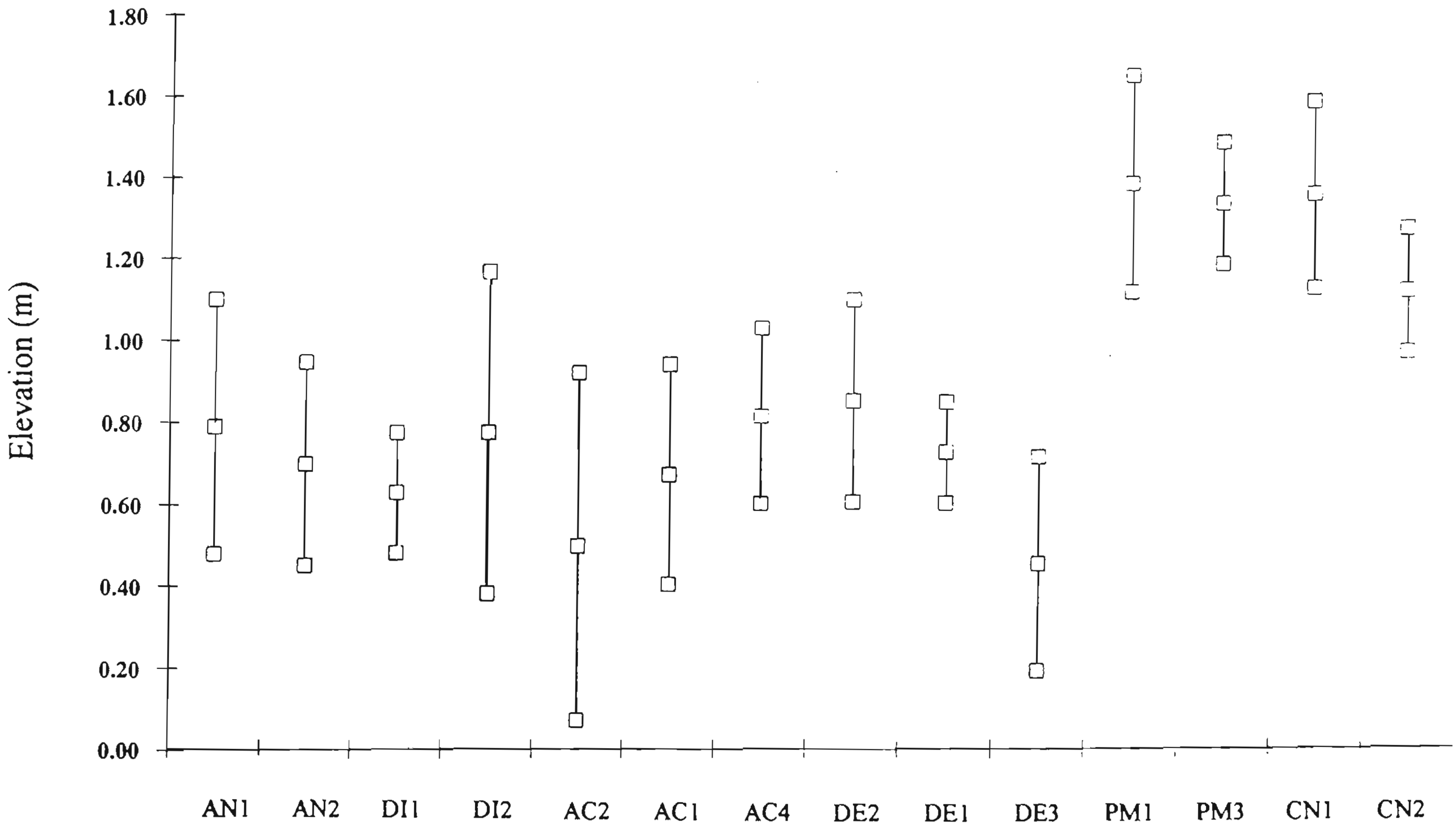

Figure 15. Geodetic elevations (mean ± s.d.) of surveyed marsh sites in the lower Fraser River estuary.

CHAPTER 4

Fish Food Invertebrates from Transplanted, Unvegetated and Natural Sedge Marshes[3]

by

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[3] This section contains excerpts from and additions to the original report: Stronach, J.A. 1995. An analysis of plant and benthic invertebrate communities in marshes on the lower Fraser River. Prepared for Department of Fisheries and Oceans, West Vancouver Laboratory, by Seaconsult Marine Research Ltd. March 1995. 77 p. On file in the West Vancouver Laboratory Library, Department of Fisheries and Oceans.

INTRODUCTION

This report presents data on invertebrates potentially available as fish food organisms in disrupted, transplanted and reference sedge marshes in the lower Fraser River. There are only a few comprehensive papers where the colonization by invertebrates of transplanted marshes have been documented (e.g., Wilcox 1986; Landin et al. 1989; Levings and Macdonald 1991; Peck et al. 1994; Batzer and Resh 1992). Some data on chironomid abundance at a transplant marsh on the North Arm have been presented (Whitehouse et al. 1993) but this is the only report from the lower Fraser River or estuary.

Within a given size range, all macroinvertebrates are potentially available as fish food for the various fish communities in the estuary. Invertebrates consumed by juvenile salmonids and other fish rearing in the lower Fraser River and estuary have been examined by numerous authors (e.g., salmonids: Levy et al. 1979, Levings et al. 1991; Birtwell et al. 1987; Levy et al. 1982; Levings et al. 1995; non-salmonids: Northcote et al. 1978; Levy et al. 1979; Birtwell et al. 1993; Archipelago Marine 1994). In general, the fish are opportunistic although obviously invertebrate taxa consumed are a function of the fish species' life style and adaptations to their primary habitat. For example, benthic or demersal species, such as starry flounder, would be expected to be more consistent users of truly benthic organisms, such as oligochaete worms, whereas surface-feeding fish, such as chum fry, prey on drift invertebrates (e.g., chironomid larvae, pupae and adults).

The sites selected for study were as follows (see also Chapter 1, present document):

1. North Arm: AN1, AN2, AN3, AN4; and DI1, DI2, DI4
2. South Arm: DE1, DE2, DE3, DE4; and AC1, AC2, AC3, AC4
3. Queens Reach: PM1, PM2, PM3; and CN1, CN2

MATERIALS AND METHODS

Invertebrates were collected from March to August 1992 during monthly surveys at 20 sampling sites (Table 1, Chapter 1, present document). Data from sites characterized by transplanted vegetation were analyzed and compared with reference marshes and disrupted (non-vegetated) sites.

Sampling for invertebrate fish food organisms was conducted using an epibenthic sled with a 10 x 10 cm mouth opening and fitted with a 44 µm net (Sibert et al. 1977). The net was pushed through the water when depth was between 30 and 50 cm, although this varied with topography of the particular site sampled. The sled samples were obtained between particular coordinates on the vegetation sampling grid as described by Moody (1993; Chapter 3, present document). There was some variation in this pattern because of local topography (e.g., small ditches and depressions). Three replicate sled samples of 5-10 m length were obtained from each site.

Samples were fixed in the field with 10% formalin mixed with rose bengal. In the laboratory, the samples were then preserved with 40% ethanol. Animals were sorted from plant material and sediment using a Wild M5 stereomicroscope and counted. Identifications were to the level of family or higher taxa (Borror and DeLong 1964; Borror et al. 1981; Borror and White 1970; Chu 1949; McAlpine et al. 1981; Merritt and Cummins 1984) and are listed in Table 1.

Invertebrate Abundance and Community Structure

The average number of invertebrate taxa and the mean density (number·m^{-2}) of invertebrates were calculated from all samples collected at each site. The mean density was also calculated for selected invertebrates grouped according to the second column in Table 1. The seasonal variation of these groups was determined. The relative abundance within each of the 13 taxonomic categories of Table 1 was used to determine quantitative measures of community structure, comparing both transplanted and disrupted sites to their respective reference site. These comparisons were based on a mean of the ratios of invertebrate densities for each taxonomic category, and a correlation coefficient between the list of invertebrate densities for each comparison pair.

For the above statistical calculation, as well as the principal component analysis, the average invertebrate density for each particular taxonomic group had to be determined. This was achieved by averaging either over a number of replicates or over the entire season. This average was calculated by averaging the logarithm of the number of animals per square metre for each invertebrate taxon (Figure 1). Numerically, this calculation may be expressed in terms of the average of N numbers as:

$$x = \frac{1}{N}\sum \log_{10}(a_n + 1) \qquad (1)$$

where: N = number of samples, either number of replicates during a monthly survey, or total over the season;
n = indexing variable; and,
a_n = number of animals per square metre

Note that adding unity to each a_n preserves a valid number for the logarithm in the event that any particular value of a_n is zero. Equation (1) assures that no one sample will dominate the average. This can occur with quantities, such as invertebrate densities, for which the standard deviation is typically two or three times as large as the arithmetic average. The seasonally-averaged mean invertebrate densities (e.g., mean ratio of the relative number of invertebrates) were then used to compare the data from transplanted and disrupted sites to their respective reference site.

Principal Component Analysis

A principal component analysis on the abundance data was conducted to search for systematic patterns in the variability about the mean for each reference-transplant-disrupted grouping. To reduce the sensitivity of the community structure to the one or two most abundant invertebrates, all densities (D) were transformed to $\log_{10}(D+1)$. Then replicates were averaged according to Equation (1) and the seasonally-averaged mean invertebrate density for each taxon as defined by Equation (1) was subtracted. These transformed data were then input to the principal component analysis.

In the principal component analysis, the correlation matrix for species-species pairs was first computed, averaging over all stations in both reference and transplant sites. The eigenvectors and eigenvalues of this matrix were then determined. The eigenvectors were the principal components, and were merely weighted species lists describing a specific grouping of organisms according to their relative abundance. The eigenvalue corresponding to each eigenvector represented the relative amount of variance explained by that eigenvector. For instance, if an eigenvector explained 90% of the variance, then one could describe all but 10% of the total species composition of each station as the mean invertebrate density, averaged over the entire estuary for the period March through August 1992, plus a station-specific scaling factor times the first eigenvector species list, E1.

RESULTS

Number of Taxa

Thirteen taxa of invertebrates were identified and enumerated. Table 1 lists these categories and four groups (A, B, C, D) to which the major taxa were assigned for the analysis of seasonal variability. The invertebrate database also contained vegetation cover data for *Carex*, *Juncus* and *Scirpus* (expressed as a Braun-Blanquet index) based on vegetation distribution maps (Moody 1993; Chapter 3, present document).

The average number of taxa from the study sites ranged from six (DI4 disrupted) to 10 (DE3 reference), and was about the same for each reach of the river. At four of the five disrupted areas, invertebrate counts were lower than at the comparable reference and transplanted marshes. At AN1, there was no difference between the disrupted site and other habitats. The number of taxa at the reference and corresponding transplant sites was about the same. There was approximately one less taxon at each of the disrupted sites compared to its reference site.

Number of Animals

The mean invertebrate density and standard error were calculated for selected species at each reference, transplanted and disrupted marsh site. Bar graphs of mean taxon abundance were constructed by reach and are shown in Figures 1 to 11. The standard error, superimposed as a vertical line terminated by a small horizontal bar (±1 SEM), was typically large for most of the invertebrate species and reflected the natural variability of the animals. Over all the sample sites, Harpacticoids, Ostracods, Copepod nauplii, Chironomid larvae and Oligochaetes were the most abundant invertebrates. Bar graphs showing the monthly mean invertebrate composition at all sites have been presented elsewhere (Stronach 1995; Appendix H).

In all cases, the mean invertebrate density at transplant and reference sites was equal to or greater than the number observed at the comparable disrupted site (Figure 1). Based only on the mean density of invertebrates, AN1, AN3, DI1, AC4, DE1 and PM1 were successful transplant marshes because they supported more invertebrates than their corresponding reference sites. Conversely, AC2 and CN1 transplant sites were less successful.

Similar results were observed when the following selected invertebrates were examined: Ostracods, Copepod nauplii, Chironomid larvae, Acarina and Ceratopogonid larvae (Figures 3, 4, 5, 6 and 10). Except for sites AN1, AN2 and DI2, the mean density of Harpacticoid copepods was also greater at the transplant and reference sites compared to the disrupted site (Figure 2). The same was found for Collembola, with the exception of site DI4 (Figure 7).

For Cladocerans and Cyclopoid copepods (Figures 8 and 9), there was either no difference in densities or the abundance at the disrupted site was slightly higher when compared to the transplant and reference sites. The only invertebrate taxon with higher mean density at disrupted sites relative to its respective transplant and reference sites was Calanoid copepods (Figure 11).

Statistical Analyses of Invertebrate Communities at Transplanted, Reference and Disrupted Sites

Two parameters were calculated in order to summarize the relationship between disrupted, transplanted, and reference marsh sites. The mean ratio represented the relative number of invertebrates at the three habitat types. A value greater than 1.0 suggested that the transplant or disrupted site showed a higher density of invertebrates relative to the reference site. The coefficient correlation was a measure of the species-to-species variability in the community structure of the three sites. If the correlation was near 1.0, then the habitats had similar structure, whereas values near 0.0 showed that the invertebrate species composition was considerably different. In the analyses, the invertebrate data from site AN3, a constructed embayment, was excluded. Several hundred square metres of vegetation were planted in 1989 at this site. However, in 1993, when the invertebrate sampling was completed, sedge plants were found only around the perimeter of the embayment.

Table 2 provides a summary of the invertebrate structure in terms of mean ratios and correlation coefficients for each marsh comparison. Appendix I in Stronach (1995) gives data on vegetation associated with particular invertebrate taxa. These relationships are discussed below, first for the entire estuary, and then for individual site pairs.

All Study Sites

A comparison of invertebrate abundance at transplant and reference marshes for the entire data set from the estuary was initially examined. The mean ratio of the seasonally-averaged densities at the reference and transplant sites was approximately 1.00, indicating roughly the same abundance (Table 2). The correlation coefficient between the ordered taxon lists was 0.97 which indicated a high degree of similarity in the community structure.

Similarly, a comparison between disrupted and reference marshes for the entire estuary was examined. Compared to the reference marshes, the disrupted sites had quite different population structures. The mean ratio of 0.61 indicated fewer animals at the disrupted sites, and the correlation coefficient of 0.57 suggested dissimilar community structure (Table 2). Indeed, the disrupted sites were considerably lower in all taxa except Copepod nauplii, Cladocerans, Cyclopoids and Calanoids.

North Arm

Invertebrate densities at DI2 reference and DI1 transplant were quite similar. DI4 (disrupted) showed considerably lower abundance than the reference site, but the correlation coefficient was relatively high (Table 2). Vegetation at the reference site was composed of *Juncus* and *Carex*, and vegetation at the transplant site was composed of *Carex* and *Scirpus*.

AN2 (reference) and AN1 (transplant) showed similar mean abundance, and the distribution of animals into the various taxa was similar at the two sites (mean ratio = 1.00, correlation = 0.95) (Table 2). These values were nearly identical to those for the combined estuary. AN4 (disrupted) showed lower numbers of animals, and the community structure differed substantially from the reference and transplant sites. The reference site supported *Carex, Juncus* and *Scirpus*, with *Carex* the dominant species. The transplant site supported *Juncus* and *Carex*, with *Juncus* the dominant species, and no *Scirpus*.

South Arm

There were two reference sites in this reach, a slough site (DE3), and a mainstem site (DE2), providing two comparisons (Table 2). Considering first the combination with a slough reference site, the mean ratios and correlations were roughly the same for DE3,1,4 as for the estuary as a whole. The reference site showed a larger abundance of *Carex*, and the transplant site was characterized by more *Juncus*, while *Scirpus* was found only at the reference site.

However, comparing the mainstem reference site (DE2,1,4) revealed that DE2 (reference) marsh showed considerably fewer invertebrates than the transplant site DE1, and only slightly more than the disrupted site (ratio = 0.94). Since vegetation at these reference and transplant sites were almost identical, some other factor must explain the magnitude of the mean density differences. Clearly, the flow and flushing regimes differ between a mainstem and a slough environment which may in turn affect invertebrate communities.

Two three-way comparisons were made with Annacis Channel marshes (Table 2): AC1,2,3 and AC1,4,3. The latter comparison was similar to the overall estuary: the ratio of mean numbers at the transplant site and reference site was 1.02, and there was a relatively high correlation coefficient (0.85) between the two distributions. The AC1,2,3 comparison indicated fewer invertebrates at AC1 (transplant) site where the mean ratio dropped to 0.65 between transplant and reference sites, and the correlation coefficient was also slightly lower (0.75).

The mean ratio for the disrupted site (AC3) was 0.39, the lowest value obtained, and the correlation coefficient was weak (0.51). Because of the thresholding used to eliminate rare taxa, the correlation coefficients between reference and disrupted sites differed in the AC1,2,3 and in the AC1,4,3 comparisons, even though the same data were used in the analysis.

Vegetation at the reference site was primarily *Carex*. At AC2 (transplant), the vegetation was mainly *Juncus* with a little *Carex*, while AC4 (transplant) supported *Juncus* in abundance and significantly more *Carex* than AC2. Increased vegetation cover at AC4 may have contributed to the higher invertebrate population relative to AC2.

<u>Queens Reach</u>

At the Port Mann locations, the invertebrate mean density was greater at the PM1 (transplant) than at the PM2 (reference), and the correlation between the two data sets was strong (Table 2). The mean ratio and the correlation coefficient for the disrupted site were about the same as for the estuary as a whole. PM3 (reference) supported three classes of vegetation (*Scirpus*, *Carex*, *Juncus*), whereas the transplant site supported primarily *Juncus*, with a lesser amount of *Scirpus*, and no *Carex*.

The CN location consisted of one reference (CN2) and one transplant (CN1) site. Both the mean number of invertebrates and the distribution of taxa, as indicated by the correlation coefficient, were quite similar at these two stations. The transplant site was characterized by limited amounts of *Juncus* and *Scirpus*. However, the reference site showed significant stands of *Equisetum fluviatile*, and both reference and transplant sites contained *Phalaris arundinacea*.

Seasonal Trends

The seasonal signal in the data for individual taxa was not readily discernible compared to the overall variability. However, when the data were aggregated into four major groups (A, B, C, D) a month-to-month variation in invertebrate density was evident (Stronach 1995). Harpacticoid copepods exhibited a peak of abundance in April at the reference and transplanted sites followed by a decline as summer progressed. The disrupted sites showed a similar, but weaker, seasonal trend that peaked in May-June. Copepod nauplii exhibited a strong seasonal signal only at the transplanted sites, with peak density in March that was an order of magnitude higher than at the reference marshes, and both achieved peak densities in May-June. Chironomid and Ceratopogonid larvae density peaked in April at the reference sites, but abundance did not fluctuate markedly at the transplanted sites. The disrupted sites were always characterized by much lower numbers of these taxa. Collembola and Acarina abundance increased at transplanted sites over the summer. At the reference sites, density peaked in March and August with values in the intervening months stable at about 50% of the spring peak. Numbers of these two taxa at the disrupted sites were much lower, except in August when the density was about the same as the reference and transplanted marshes in May, June and July.

Principal Component Analysis

Figure 12 gives an example plot of the eigenvector composition for seasonally-averaged invertebrate data, presented as bar graphs of the first two eigenvectors (E1 and E2) for reference/transplanted pairs CN1 and CN2. An example of the scatter plots of E1 versus E2 amplitudes for each quadrat, plotted separately for quadrats from transplant and reference marshes, is given in Figure 13. Similar plots for invertebrate data for particular reaches, vegetation data, and invertebrate data combined with vegetation, are given in Stronach (1995).

Combined Estuary Data

The structure of the first two eigenvectors of the combined estuary data for invertebrates explained 68% of the variance in the invertebrate composition of the various stations, E1 explaining 53% and E2 explaining 15%. The first eigenvector (Figure 14) had a similar structure to the seasonally-averaged mean density data (Figure 15). For instance, the largest seven entries in the seasonally-averaged mean list and in E1 differed in one element only: Cladocerans were found in the mean data plot and Ceratopogonid larvae appeared in the display of E1. Thus, the first eigenvector accounted for variations in density from site to site without significant change in community structure. Some of this variation could be seasonal. For instance, the dominant taxon in E1, Chironomid larvae, tended to be present in greater abundance in April and May, with reduced abundance in later surveys (Figure 16).

In E2 (Figure 17), taxa from groups D (Collembola and Acarina) and C (Ceratopogonid larvae and Chironomid larvae) comprised most of the positive weights while groups A (Harpacticoids) and B (Copepod nauplii, Cyclopoids, etc.) were important members with negative weights. This eigenvector was thus related to differences in community structure among the various stations. Because each of these groups revealed month-to-month trends (Stronach 1995), seasonal variability was likely an important parameter for interpretation of the E2 structure.

Using the eigenvector structure defined by all invertebrate data from the estuary, the amplitudes of E1 and E2 were determined for each month, with all stations considered collectively. The E1 amplitudes (Figure 18) indicated a significant difference between reference/transplanted marshes and disrupted sites: most of the time the reference and transplanted sites showed positive amplitudes, whereas the disrupted sites always had negative values. Since E1 was similar in structure to the mean density (see above), this diagram illustrates that at all times over the estuary as a whole, the aggregate of the disrupted sites supported fewer animals than either the reference or transplanted sites. For E2 (Figure 19), a reasonably strong seasonal signal was present for all three types of sites, and reflected the seasonal change in community structure. For instance, in March, April and May the density of taxa in groups A and B (harpacticoid copepods and copepod nauplii) which had negative E2 weights was much higher than in the subsequent three months (Stronach 1995). Members of group D (Collembola and Acarina) which had large positive weights in the E2 structure, tended to increase from relatively low densities (negative E2 amplitude) prior to June to maximum densities in August (Stronach 1995).

Based on these two eigenvectors, one can probably assume that at least 53% of the variance was tied up with the difference between less productive disrupted sites and the more productive reference and transplanted sites. At least 15% was associated with seasonal changes.

The scatter plots of eigenvector amplitude have been given in detail elsewhere (Stronach 1995) are an alternate expression of the patterns noted in the monthly eigenvector amplitudes. The reference sites were about equally distributed, positively and negatively, with respect to E2, and they had mainly positive values of E1. The transplanted sites were similar, although the E2 range was somewhat larger. The disrupted sites were completely different, with almost all points lying in the negative E1 plane. The E2 distribution for disrupted sites was similar to the other two marsh types, although there was a slight bias to negative values.

DISCUSSION

Determining whether or not a transplanted marsh adequately compensates for habitat loss is a complex issue, since the quality of the habitat is difficult to evaluate in quantitative terms. Our report describes procedures which were investigated as possible ways to provide quantitative guidance in setting the parameters for a transplanted site, and in evaluating the ecological performance of them.

Invertebrate Statistics

The seven most abundant invertebrate taxa in the sample data set were: Harpacticoids, Ostracods, Copepod nauplii, Chironomid larvae, Oligochaetes, Cladocerans and Acarina. The first three taxa mentioned usually had the highest seasonally-averaged mean densities at each reference or transplanted marsh site compared to the latter three.

By grouping the major invertebrate taxa into four groups, seasonal patterns were found that revealed differences among the groups and among the three habitat types. Generally, invertebrate density was much lower at the disrupted sites, and their spring peak density was delayed one or two months with respect to the reference and transplanted marshes.

Community Structure

Characterization of relative invertebrate community structure between reference and transplanted marshes was accomplished by calculating the mean density ratio and the inter-site correlation coefficient among the individual taxa using their mean densities.

The benthic invertebrate data indicated that all transplanted marshes except AC2 supported higher invertebrate densities than the corresponding reference sites. Again with the possible exception of AC2, the invertebrate community structures were fairly well correlated between reference and transplanted sites. All unvegetated disrupted sites supported lower invertebrate densities. DE2 reference marsh supported significantly lower invertebrate densities than other marshes. Reported densities were a little higher there than at the unvegetated, DE4 disrupted site.

Although we initially hypothesized that the transplanted sites were not adequate for invertebrates, the abundance data obtained by epibenthic sled indicated that the transplanted sites were as good or better habitats than the reference sites. Other information, such as food value and provenance of the detritus arising from transplanted and reference marshes, are also required to determine their ecological significance.

Principal Component Analysis

Principal component analysis of invertebrate density was conducted in order to determine the causes for the observed variability between stations. Preliminary interpretations of the first two principal components, E1 and E2 structures, were advanced.

For invertebrates, the eigenvector structure for E1 was similar to the mean density structure; hence, the principal mode of variation was a general fluctuation in density without major change in community composition. For the estuary as a whole, the first eigenvector explained 53% of the variance in the invertebrate samples. The monthly D1 amplitudes discriminated clearly between reference/transplanted marshes and disrupted sites. In addition, the reference and transplanted marshes exhibited a seasonal trend in the E1 amplitude with a peak in early spring.

The second eigenvector, which explained a further 15% of the variance, appeared to discriminate the four major taxonomic groups which each displayed distinct seasonal trends. The monthly E2 amplitudes divided the data into two three-month time blocks: March-May and June-August. Variation among the reference, transplant and disrupted sites played a lesser role.

Based on data examined in this study, the largest factors in determining invertebrate density were a) whether or not the marsh was vegetated; and, b) the time of year.

The combined analysis of vegetation and invertebrates did not change the invertebrate structure of either the first eigenvector or the monthly E1 amplitude in any important way (Stronach 1995). Usually the dominant plant species was associated with the first four or five invertebrates in E1, and the dominant invertebrates commonly included Harpacticoids, Ostracods, Oligochaetes and Chironomid larvae. Variability in the dominant vegetation species was associated with changes in Harpacticoids, Oligochaetes and the other dominant taxa in the E1 structure. With only one exception, transplanted marshes supported higher densities of invertebrates than paired reference marshes, and commonly did so when *Juncus* was the dominant vegetation. AC2 was the only transplanted marsh with substantially lower invertebrate densities than the reference marsh.

It appears that, with the available data, the principal component analysis of invertebrates was more successful than an analysis of the vegetation community structure in explaining observed variability among the marshes. Although an unvegetated habitat apparently cannot support high invertebrate densities, as measured with the present techniques, the abundance of vegetative cover otherwise does not correlate with benthic invertebrate density. Other factors are required to explain the vegetation variance, and might include the physical and chemical analysis of the sediment and water column.

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Table 1. Invertebrate taxa identified in the lower Fraser River marshes, March to August, 1992.

Taxon	Group
Harpacticoids	A
Ostracods	*
Copepod nauplii	B
Chironomid larvae	C
Oligochaetes	*
Acarina	D
Collembola	D
Cladocerans	B
Cyclopoids	B
Ceratopogonid larvae	C
Polychaetes	*
Calanoids	B
Other Malacostracans	*

* indicates data for this taxa were not subjected to detailed statistical analyses because they were not considered important fish food organisms or were present in low numbers. Mean abundance data for all taxa are given elsewhere (Stronach 1995) and the raw data are on file at the West Vancouver Laboratory, Department of Fisheries and Oceans

Table 2. Mean ratio of seasonally-averaged invertebrate density and correlation coefficient between densities for invertebrate taxa, calculated between altered and natural marsh sites in the Fraser estuary 1992.

Reach	Transplant:Reference			Disrupted:Reference		
	Sites	Ratio[1]	Correlation[2]	Sites	Ratio[1]	Correlation[2]
All	Estuary (all sites)	1.00	0.97	Estuary (all sites)	0.61	0.54
North Arm	DI1: DI2	1.04	0.94	DI4: DI2	0.57	0.80
	AN1: AN2	1.00	0.95	AN4: AN2	0.77	0.59
South Arm	DE1:DE3	1.03	0.85	DE4:DE3	0.51	0.43
	DE1:DE2	1.88	0.92	DE4:DE2	0.94	0.70
	AC2:AC1	0.65	0.75	AC3:AC1	0.39	0.43
	AC4:AC1	1.02	0.85	AC4:AC1	0.39	0.51
Queens	PM1:PM3	1.22	0.94	PM2:PM3	0.68	0.45
	CN1:CN2	1.02	0.96	-	-	-

[1] measure of similarity in invertebrate abundance
[2] degree of similarity in the community structure

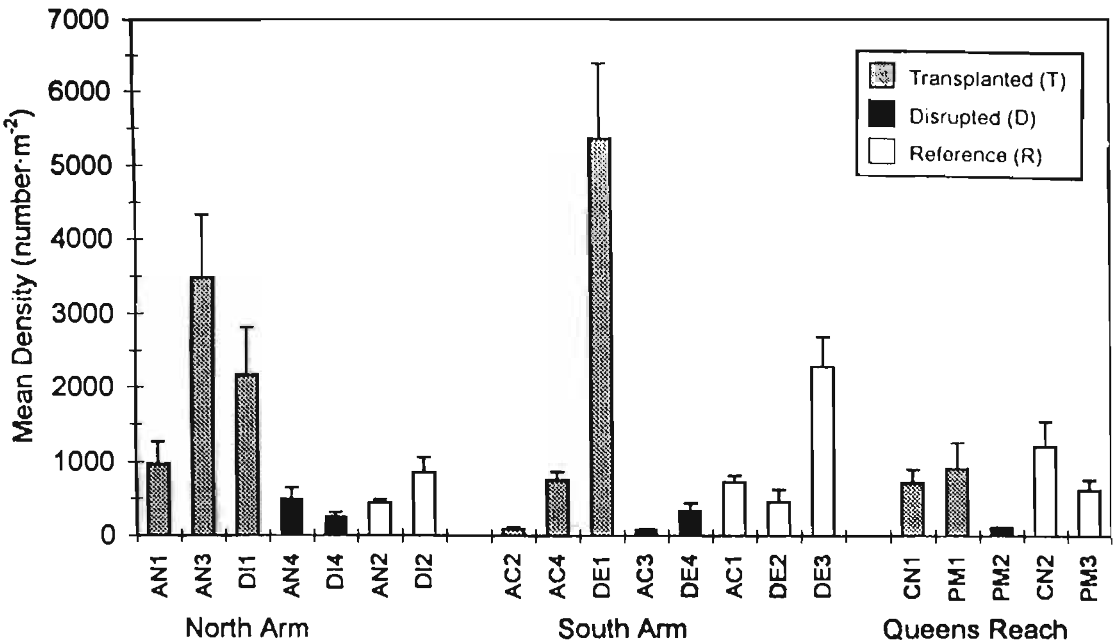


Figure 1. Mean density (number·m^{-2} ± SEM) of invertebrates (all taxa combined) at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

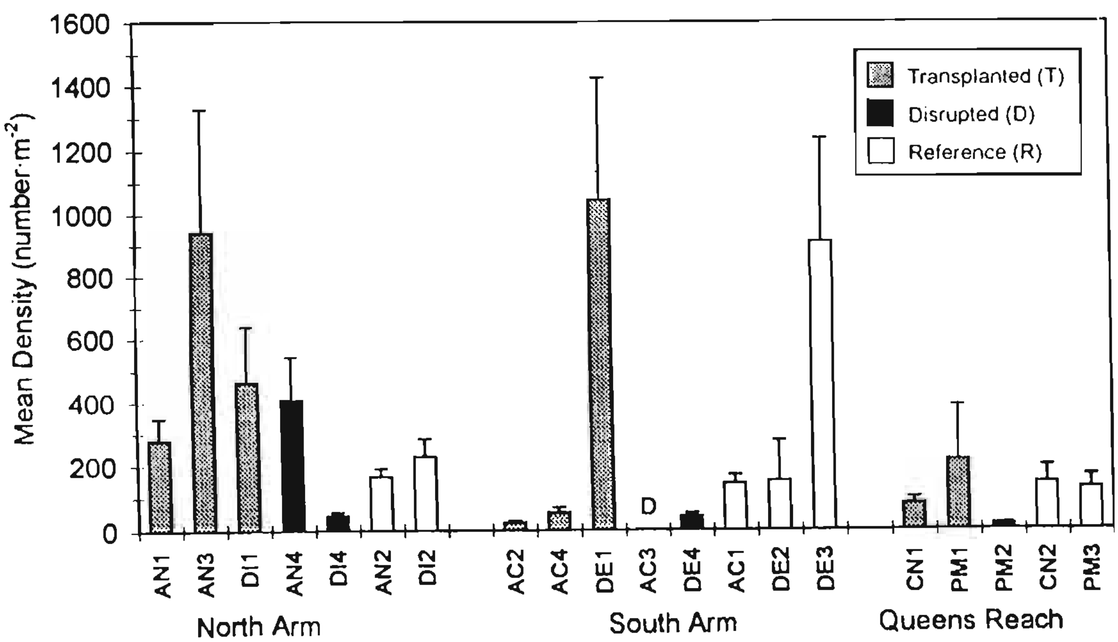


Figure 2. Mean density (number·m^{-2} ± SEM) of harpacticoid copepods at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

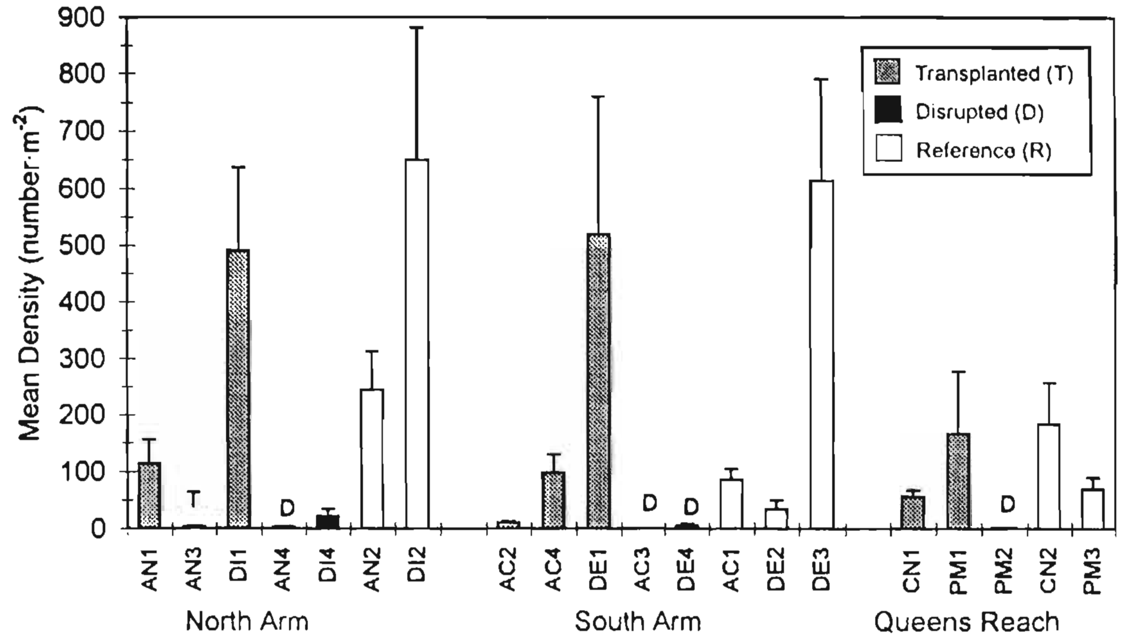


Figure 3. Mean density (number·m^{-2} ± SEM) of ostracods at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

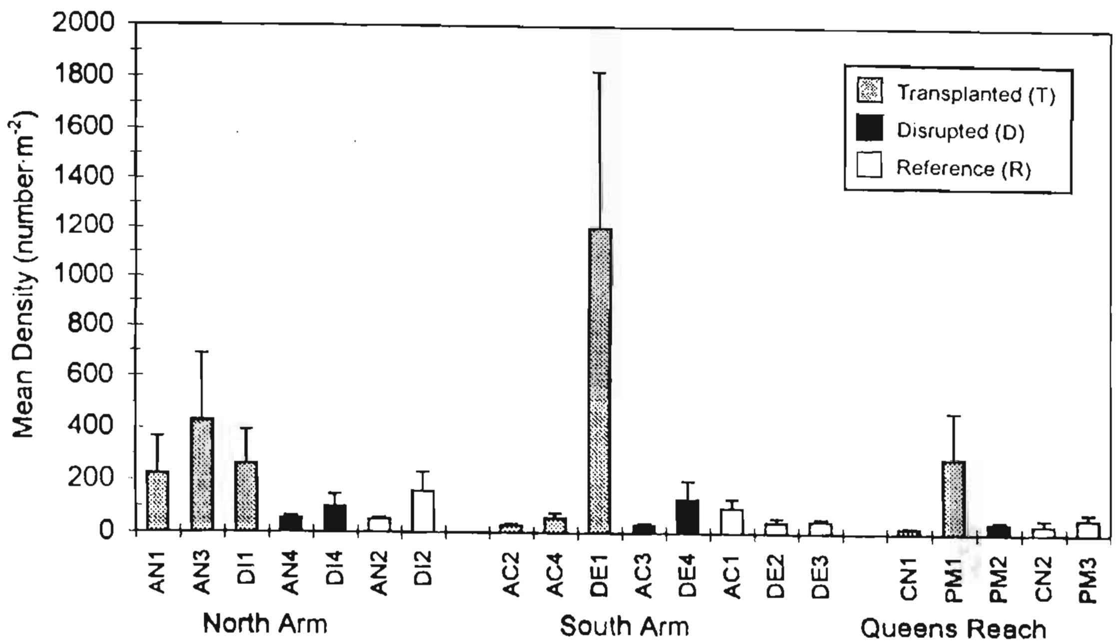


Figure 4. Mean density (number·m^{-2} ± SEM) of copepod nauplii at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

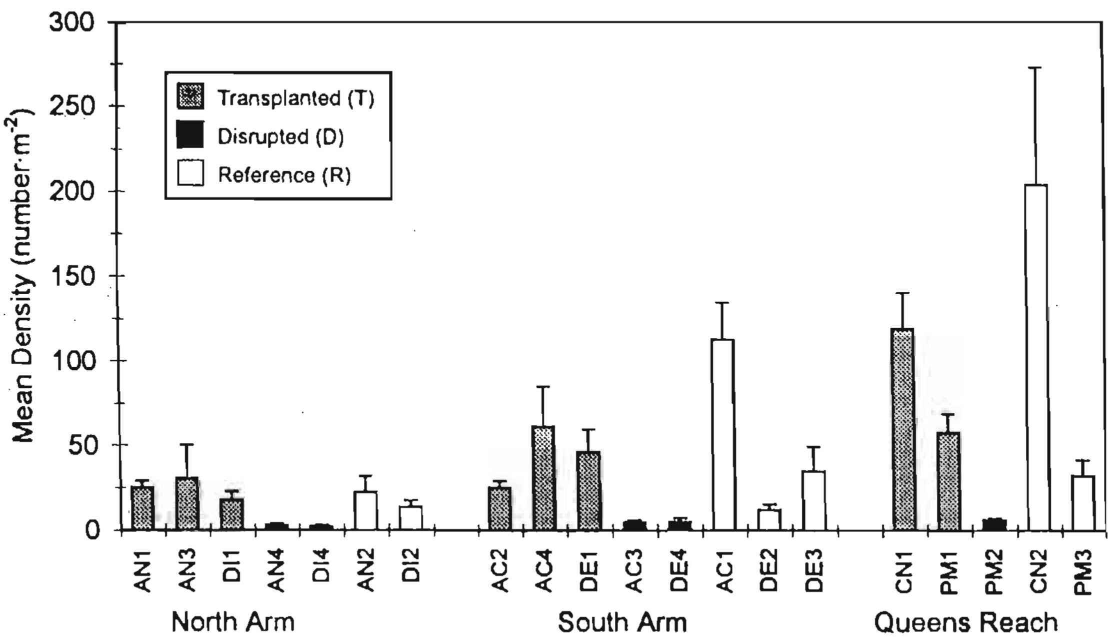


Figure 5. Mean density (number·m^{-2} ± SEM) of chironomid larvae at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

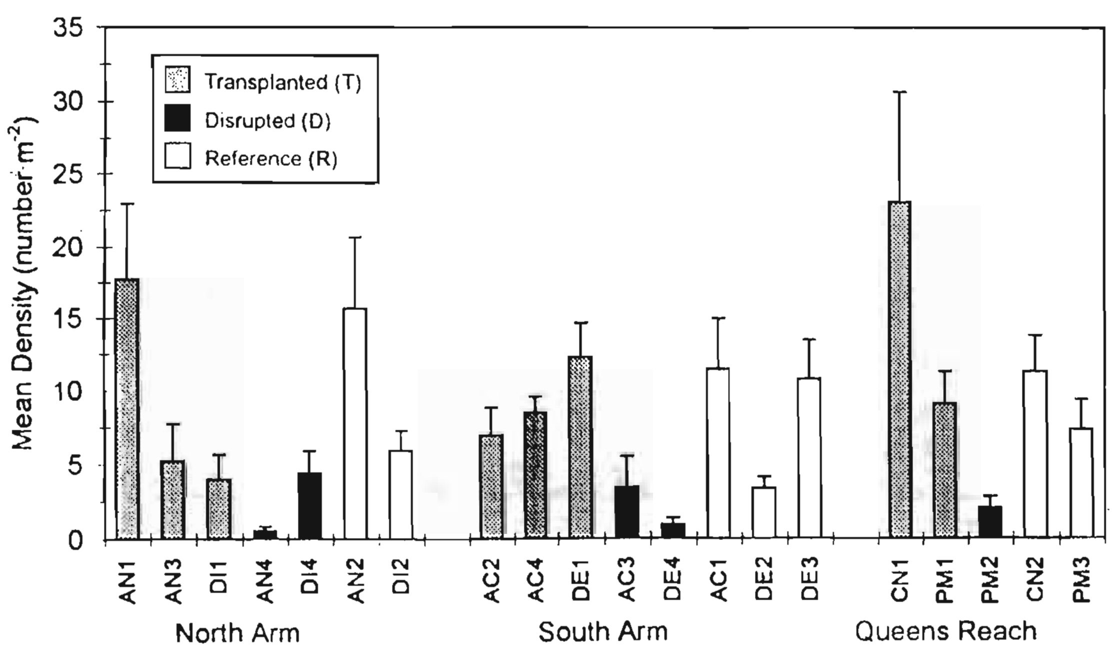


Figure 6. Mean density (number·m^{-2} ± SEM) of acarina at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

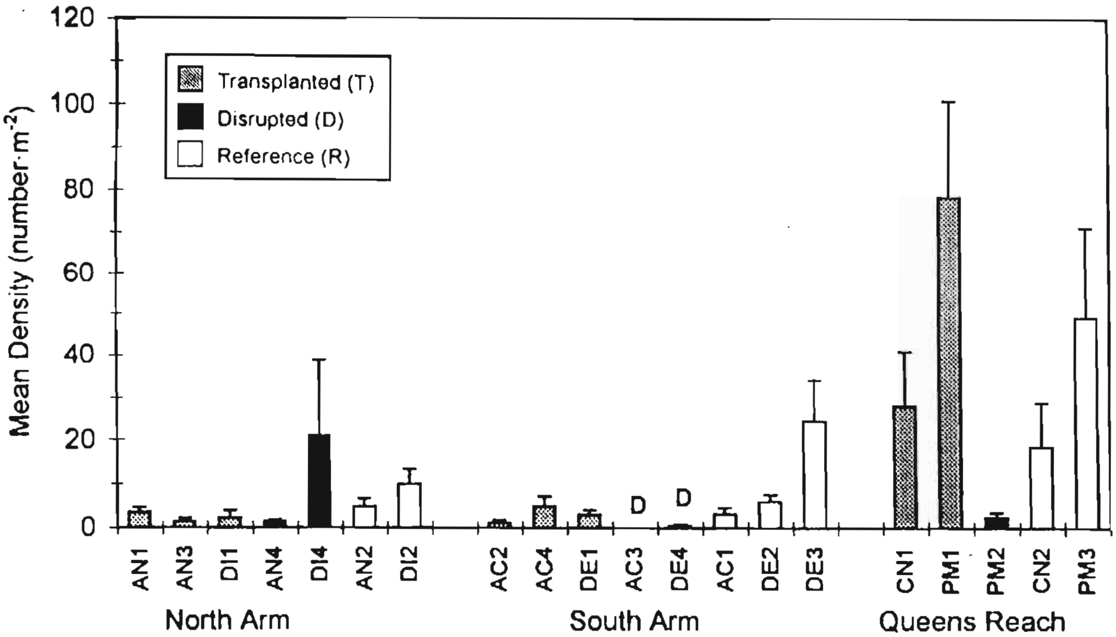


Figure 7. Mean density (number·m^{-2} ± SEM) of collembola at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

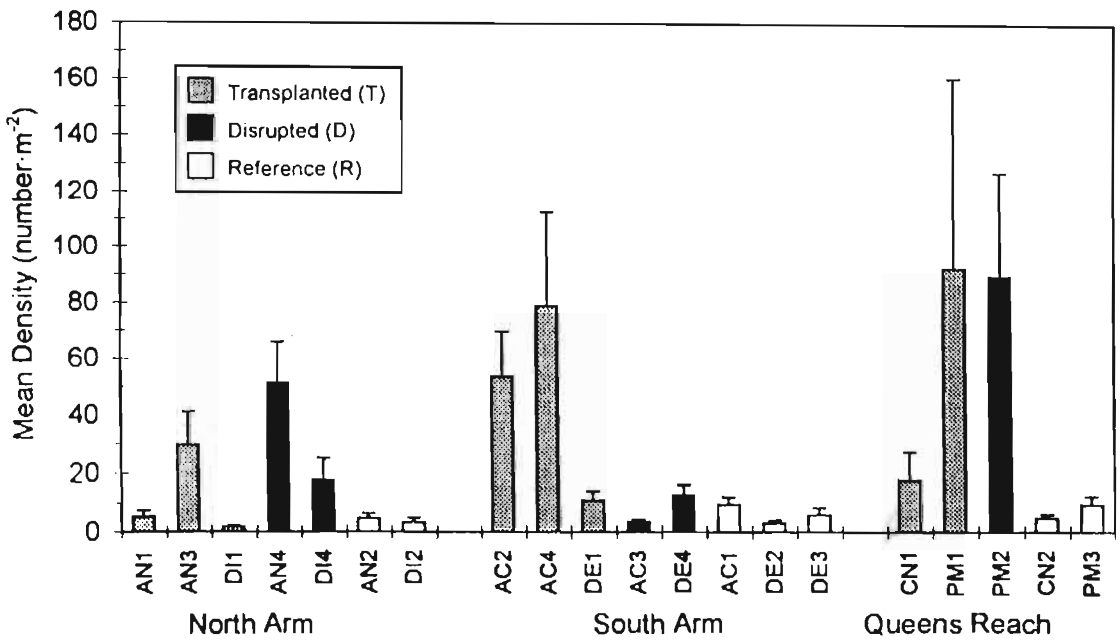


Figure 8. Mean density (number·m^{-2} ± SEM) of cladocerans at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

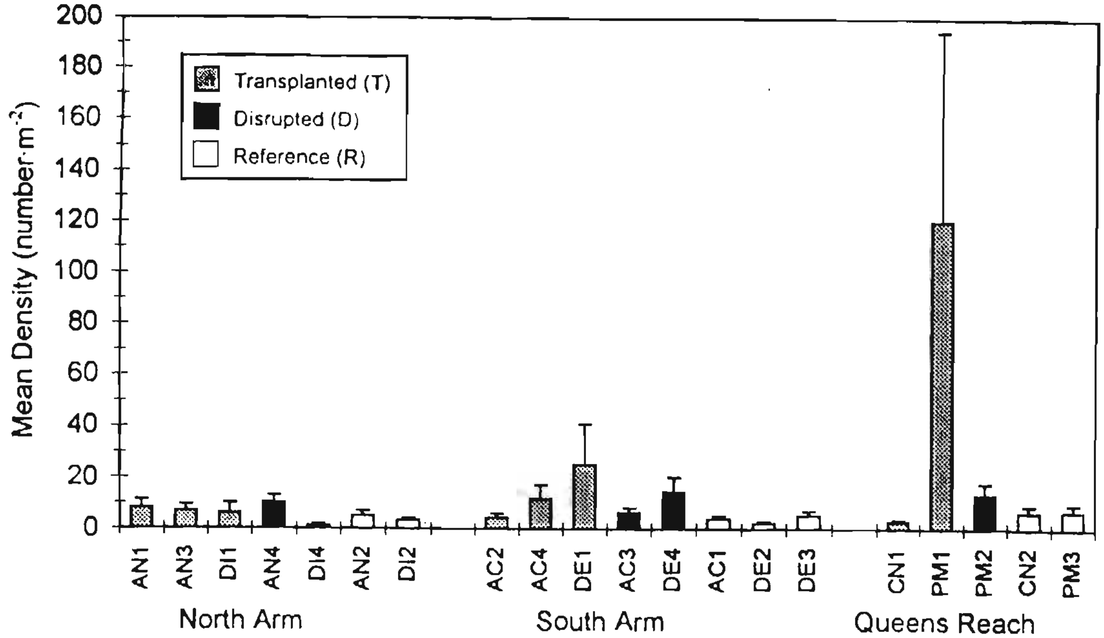


Figure 9. Mean density (number·m^{-2} ± SEM) of cyclopoid copepods at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

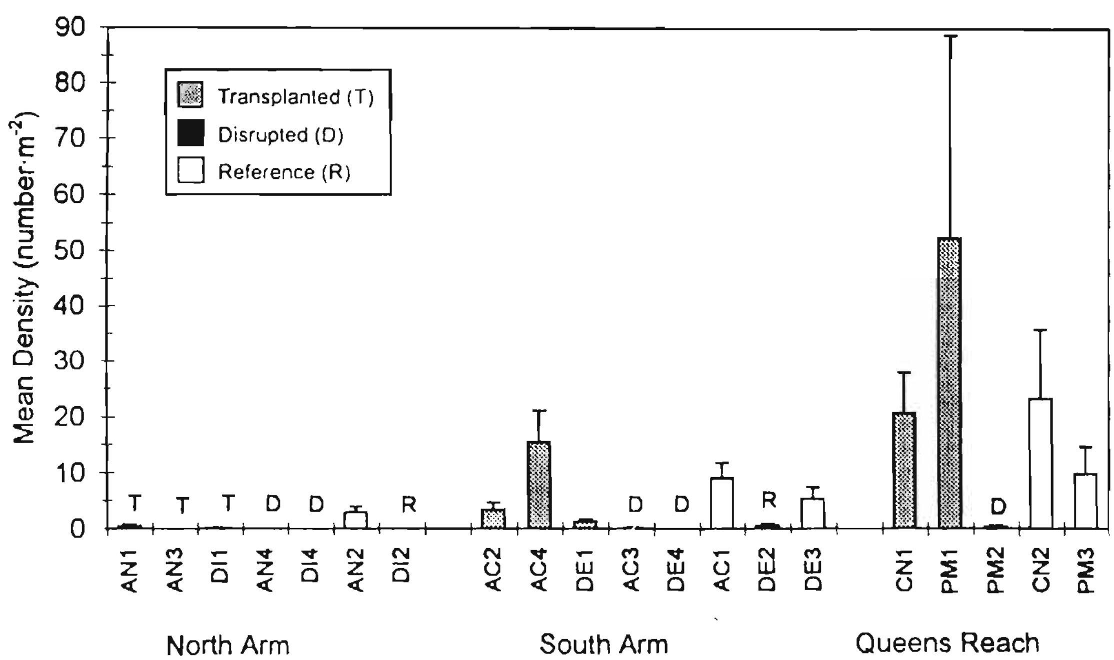


Figure 10. Mean density (number·m^{-2} ± SEM) of ceratopogonid larvae at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).

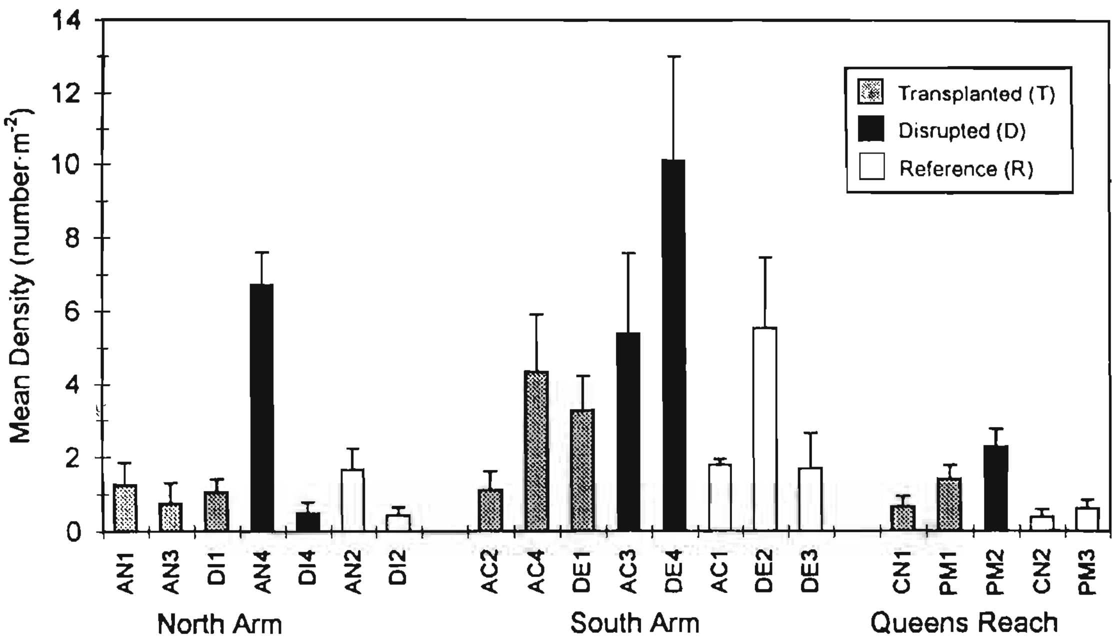
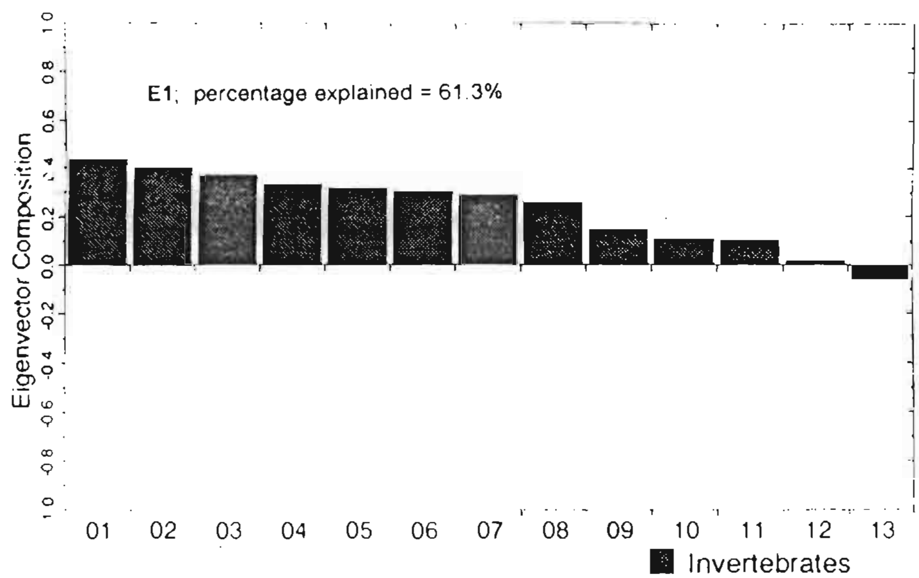
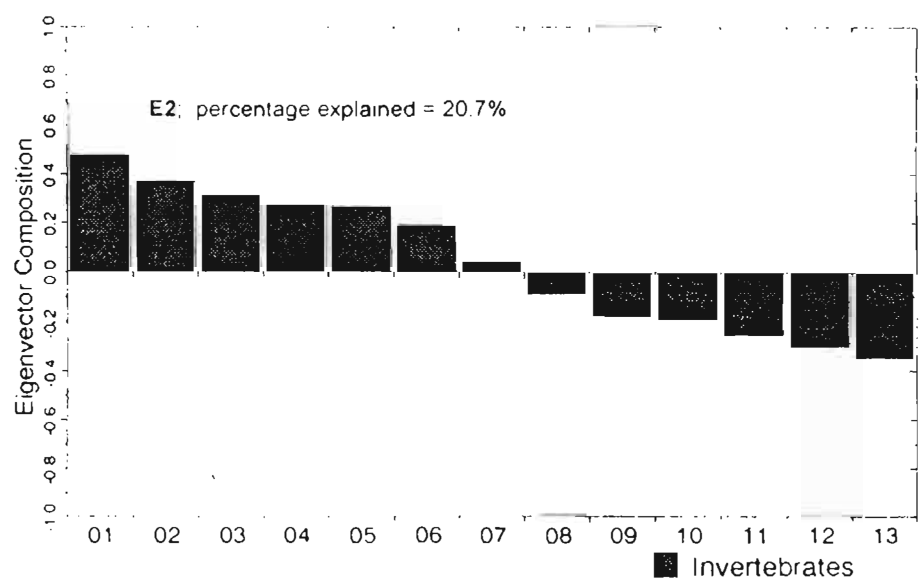


Figure 11. Mean density (number·m^{-2} ± SEM) of calanoid copepods at disrupted, transplanted and reference marshes in the Fraser River estuary, March to August 1992 (see Table 1, Chapter 1 of present document for site codes).



Invertebrates Legend

01 - Ostracods
02 - Chironomid larvae
03 - Harpacticoids
04 - Oligochaetes
05 - Other Malacostracans
06 - Ceratopogonid larvae
07 - Cyclopoids
08 - Acarina
09 - Collembola
10 - Cladocerans
11 - Calanoids
12 - Copepod nauplii
13 - Polychaetes



Invertebrates Legend

01 - Calanoids
02 - Collembola
03 - Acarina
04 - Oligochaetes
05 - Cladocerans
06 - Ceratopogonid larvae
07 - Polychaetes
08 - Ostracods
09 - Chironomid larvae
10 - Harpacticoids
11 - Other Malacostracans
12 - Copepod nauplii
13 - Cyclopoids

Figure 12 Seasonally-averaged invertebrate composition for the first and second eigenvector of CN marsh sites (Appendix J in Stronach (1995)).

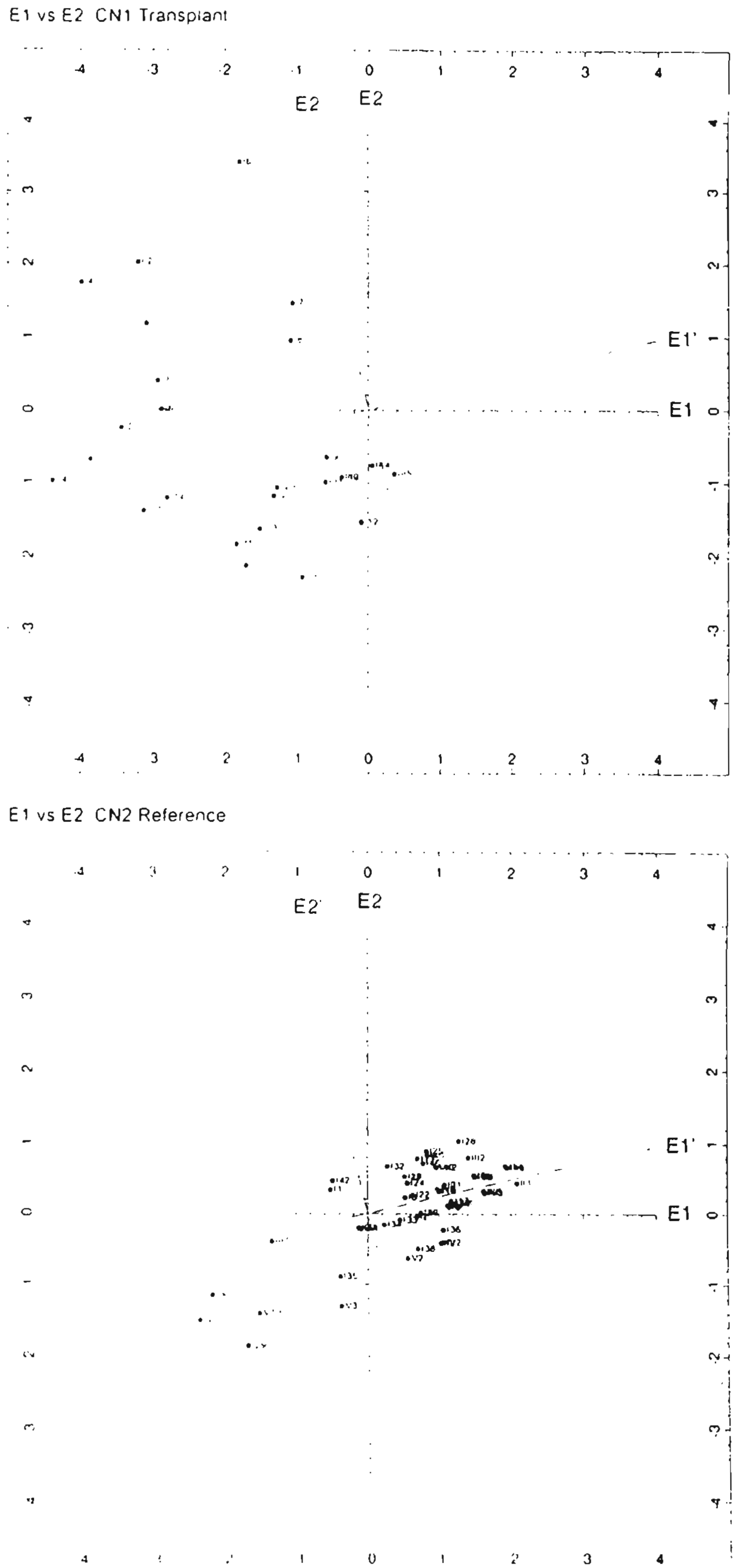
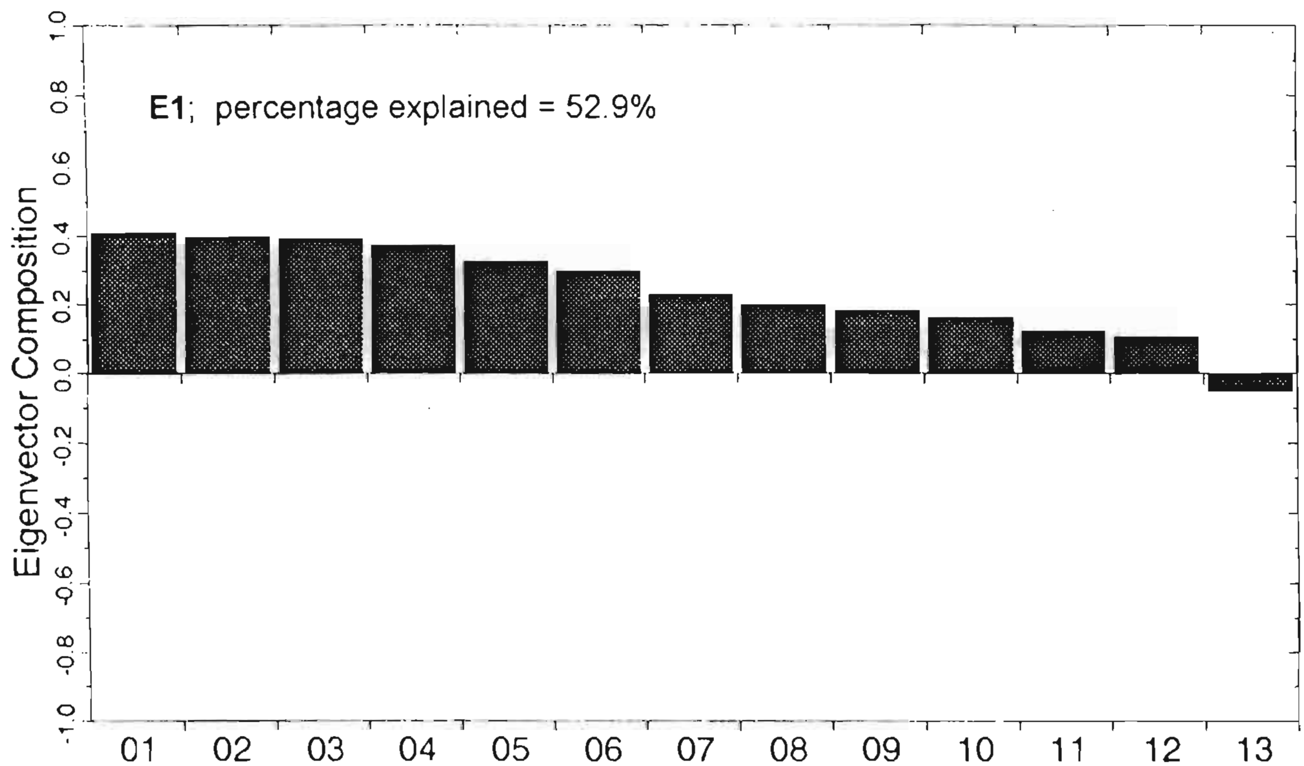


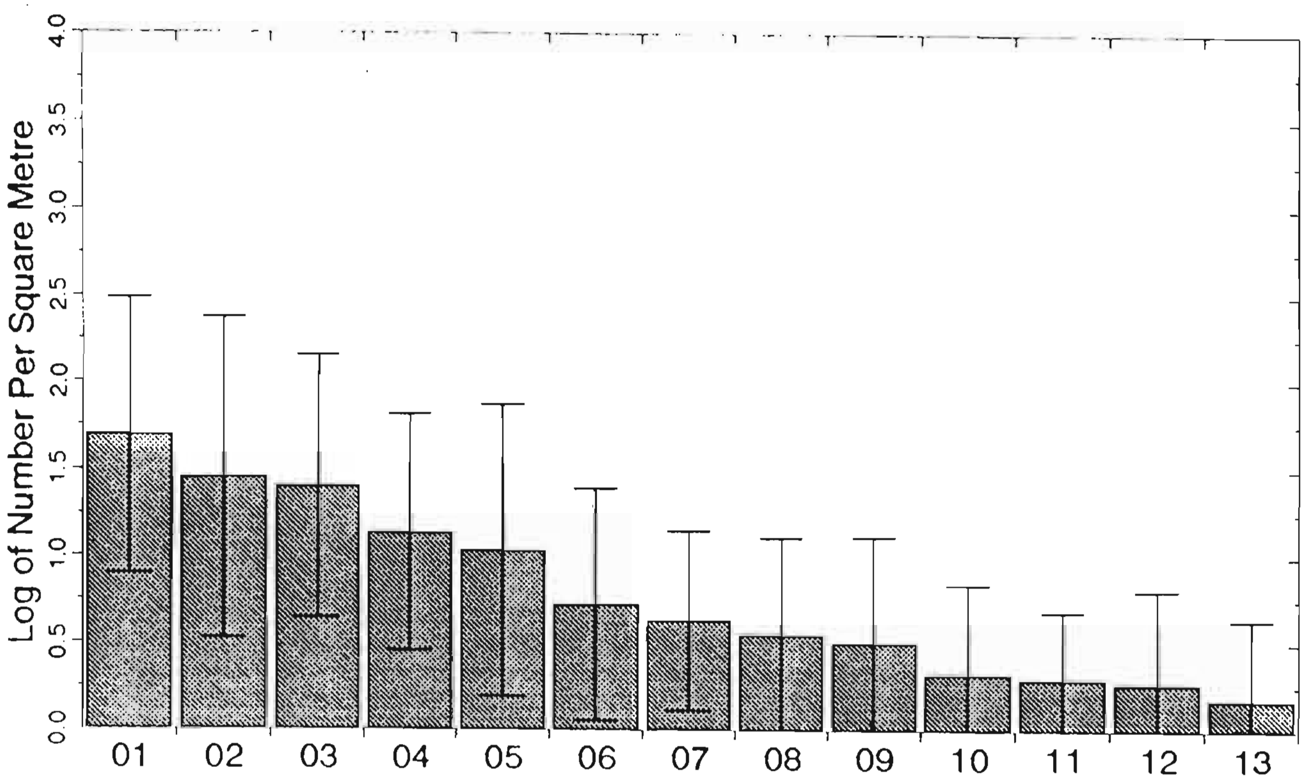
Figure 13. Scatter plots of the CN1 and CN2 marsh sites in the E1/E2 plane corresponding to the eigenvector structure for the marsh under construction (Appendix E in Stronach (1995)).



Invertebrates Legend

01 - Chironomid larvae
02 - Oligochaetes
03 - Harpacticoids
04 - Ostracods
05 - Ceratopogonid larvae
06 - Acarina
07 - Copepod nauplii
08 - Collembola
09 - Other Malacostracans
10 - Cyclopoids
11 - Polychaetes
12 - Cladocerans
13 - Calanoids

Figure 14 Invertebrate eigenvector structure of the first principal component, E1, for all invertebrate marsh sites in the Fraser River estuary (Figure 5.8 in Stronach (1995)).



Invertebrates Legend

01 - Harpacticoids
02 - Ostracods
03 - Copepod nauplii
04 - Chironomid larvae
05 - Oligochaetes
06 - Cladocerans
07 - Acarina
08 - Cyclopoids
09 - Collembola
10 - Ceratopogonid larvae
11 - Calanoids
12 - Polychaetes
13 - Other Malacostracans

Figure 15. Mean density of each invertebrate taxon for all marsh sites in the Fraser River estuary (Figure 5.7 in Stronach (1995))

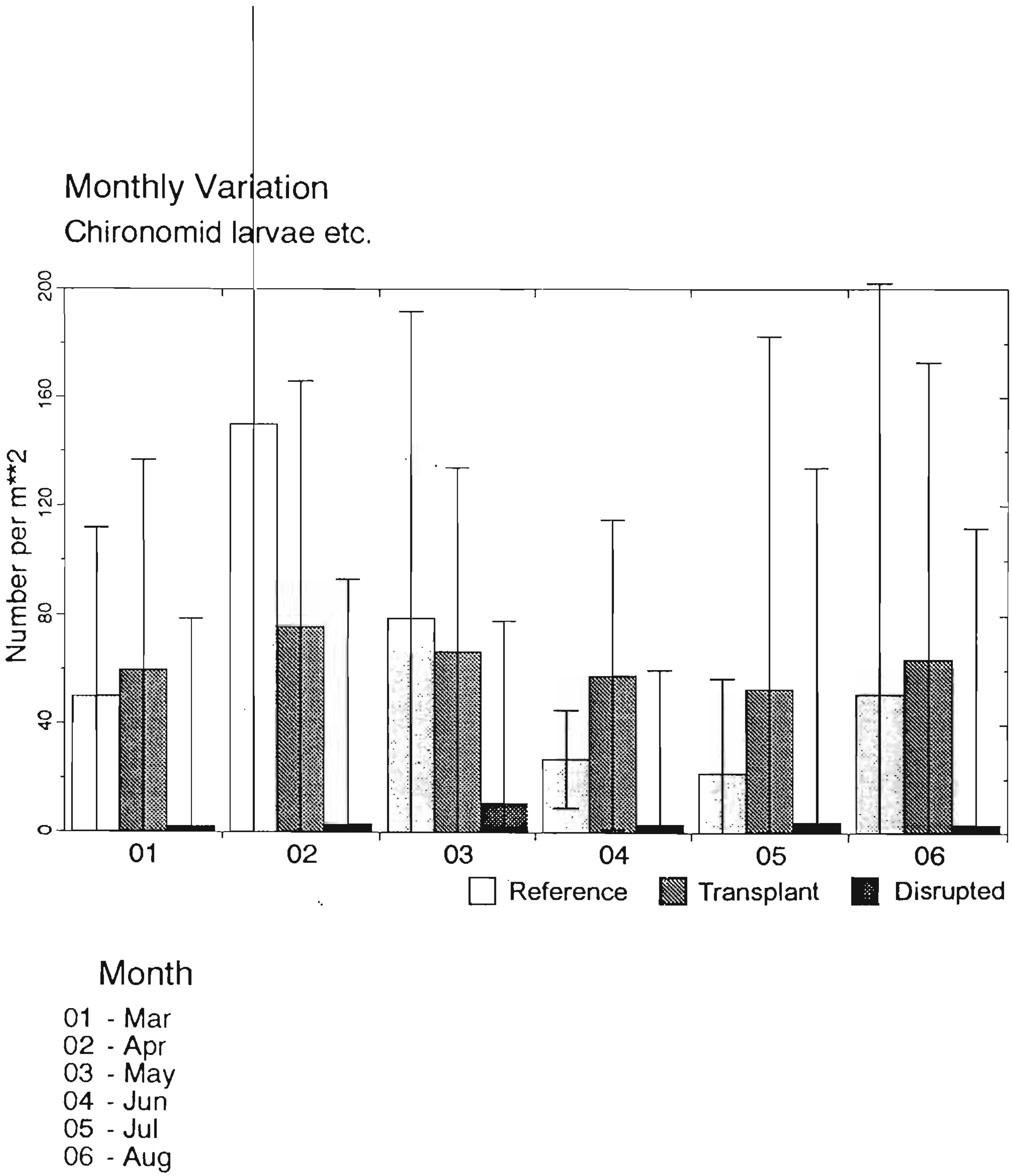
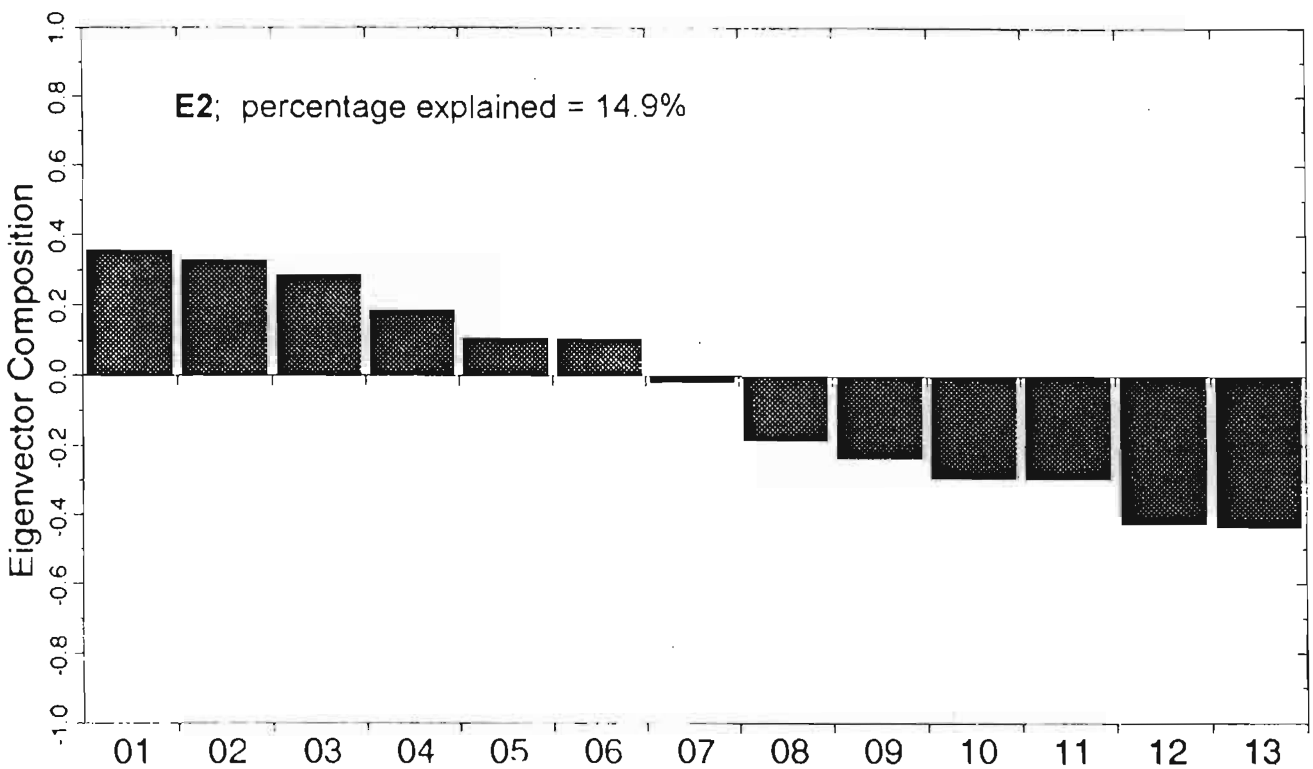


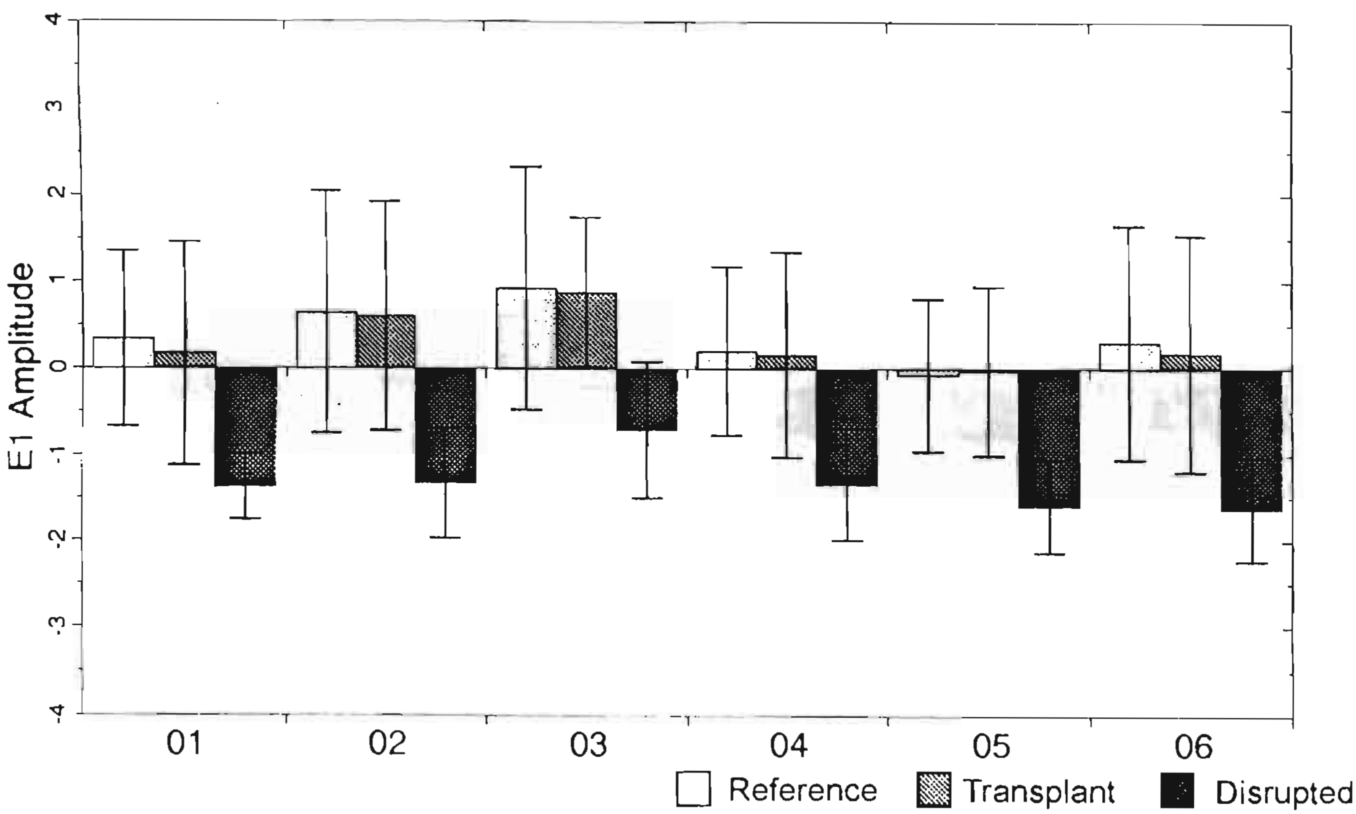
Figure 16. Mean density for each invertebrate taxon for all reference, transplanted and disrupted marsh sites in the Fraser River estuary (Figure 5.6 in Stronach (1995)).



Invertebrates Legend

01 - Collembola
02 - Acarina
03 - Ceratopogonid larvae
04 - Cladocerans
05 - Oligochaetes
06 - Chironomid larvae
07 - Ostracods
08 - Calanoids
09 - Polychaetes
10 - Harpacticoids
11 - Cyclopoids
12 - Copepod nauplii
13 - Other Malacostracans

Figure 17. Invertebrate eigenvector structure of the second principal component, E2, for all invertebrate marsh sites in the Fraser River estuary (Figure 5.9 in Stronach (1995)).



Month Legend

01 - March 1992
02 - April 1992
03 - May 1992
04 - June 1992
05 - July 1992
06 - August 1992

Figure 18 Monthly E1 amplitudes for reference, transplanted and disrupted marsh sites in the Fraser River estuary (Figure 5.10 in Stronach (1995)).

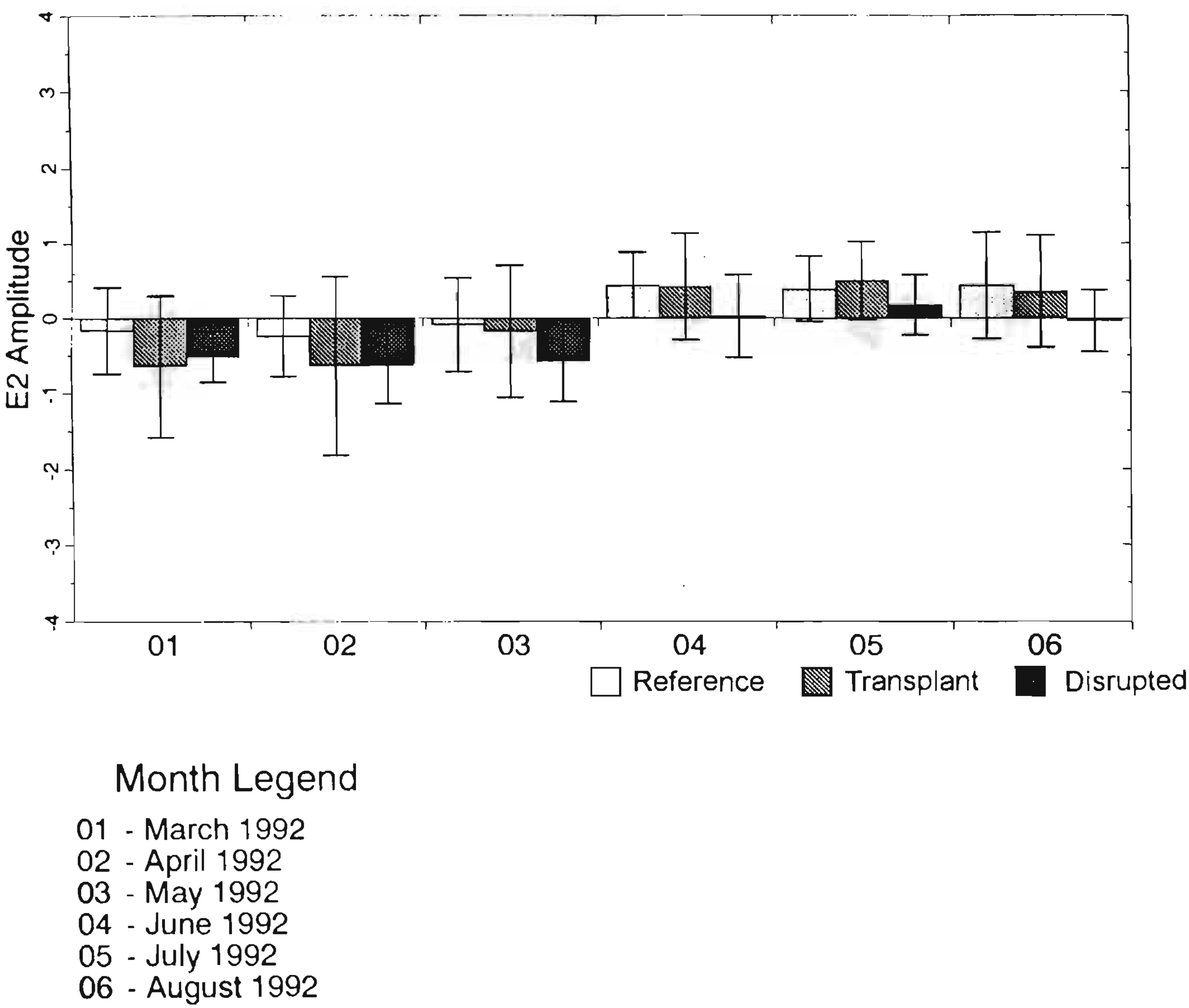


Figure 19. Monthly E2 amplitudes for reference, transplanted and disrupted marsh sites in the Fraser River estuary (Figure 5.11 in Stronach (1995)).

CHAPTER 5
Extended Abstract

Analysis of Fish Occurrence in Transplanted, Unvegetated and

Natural Marsh Habitats in the Fraser River Estuary,

1992 (extended abstract)[4]

by

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[4] This section contains revisions to the original report: Scott, K.J., and R. Susanto. 1993. Analysis of fish occurrence in restored, unvegetated and natural habitats in the Fraser River estuary, March through August 1992. Prepared for the Department of Fisheries and Oceans, West Vancouver Laboratory, by Scott Resource Services Inc. November 1993. 27 p. On file in the West Vancouver Laboratory Library, Department of Fisheries and Oceans.

INTRODUCTION

In 1992, beach seining was conducted at most of the transplanted, reference and disrupted sites to determine if there were differences in the catches of juvenile salmonids and other fish species. The complete results from this particular aspect of the study are given elsewhere, including tabulations of the original data (Scott and Susanto 1993). An extended abstract of the methods and results from the fish sampling program is given below. The sites selected for study were as follows (see also Chapter 1, present document):

1. North Arm: AN1, AN2, AN3, AN4; and DI1, DI2, DI4
2. South Arm: DE1, DE2, DE3, DE4; and AC1, AC2, AC3, AC4
3. Queens Reach: PM1, PM2, PM3; and CN1 CN2

METHODS

Fish samples were obtained when water depth over the vegetated study sites was about 50 cm deep using a 15 m beach seine deployed by hand. Samples were obtained each month between March and August 1992. Water depth at the unvegetated sites was usually deeper and a boat was sometimes used at these sites. Three replicate seine sets were made with 5 to 15 minutes between sets. Fish were held in buckets between the sets so the sampling was without replacement.

RESULTS AND DISCUSSION

Nineteen species of fish (Table 1) and 5,205 individuals were caught in the beach seining program. There were 142 sets made at the transplant sites, 127 at reference sites, and 83 at the disrupted sites. Of the fish caught 26.8% were salmonids and the remainder were smelts, cyprinids, suckers, sticklebacks, cottids and flatfish. In general, the species composition of the fish community was similar at the transplant, reference and disrupted sites.

Since the catch of all species of salmonids was greatly reduced by the end of June only data from fish captured during the spring (March to June) were examined. Mean abundance ($fish \cdot m^{-2}$) for chum (*Oncorhynchus keta*) and chinook (*O. tshawytscha*) salmon fry captured are given in Table 2. There were no statistically significant differences in catches between transplanted, reference and disrupted sites ($p>0.05$). The power of these statistical tests was relatively low because of the small number of replicate samples obtained at each site and time. When averaged over all sites in particular reaches, mean chinook and chum fry abundance ranged from 0.9 to 2.9 $fish \cdot m^{-2}$. Sockeye (*O. nerka*) fry abundance was lower and ranged from 0.1 to 0.3 $fish \cdot m^{-2}$ (Table 2). Catches of chinook and sockeye smolts were statistically different when transplant, reference and disrupted sites were compared ($p<0.05$). Smolt catches were usually higher at disrupted, unvegetated sites (Table 2), possibly because seine hauls at these locations were usually in deeper water. In addition, the lack of vegetation at these sites may have led to improved catchability of smolts.

REFERENCES

Scott, K.J., and R. Susanto. 1993. Analysis of fish occurrence in restored, unvegetated and natural habitats in the Fraser River estuary, March through August 1992. Prepared for the Department of Fisheries and Oceans, West Vancouver Laboratory, by Scott Resource Services Inc. November 1993. 27 p.

Table 1. Scientific and common names of fishes collected from the Fraser River estuary, March through August 1992.

Family	Scientific Name	Common Name
Salmonidae	*Prosopium williamsoni* *Oncorhynchus clarki clarki* *O. gorbuscha* *O. keta* *O. nerka* *O. tshawytscha*	Rocky Mountain whitefish Cutthroat trout Pink salmon Chum salmon Sockeye salmon Chinook salmon
Osmeridae	*Spirinchus thaleichthys* *Thaleichthys pacificus*	Longfin smelt Eulachon
Cyprinidae	unidentified cyprinid *Mylocheilus caurinus* *Ptychocheilus oregonensis* *Richardsonius balteatus*	 Peamouth chub Northern squawfish Redside shiner
Catostomidae	unidentified sucker *Catostomus macrocheilus*	 Largescale sucker
Gasterosteidae	*Gasterosteus aculeatus*	Threespine stickleback
Cottidae	unidentified sculpin	
Pleuronectidae	*Platichthys stellatus*	Starry flounder
Centrarchidae	*Lepomis gibbosus*	Pumpkin seed
Ammodytidae	*Ammodytes hexapterus*	Pacific sandlance

Table 2. Mean density (fish·m^{-2}) of juvenile salmon captured using a beach seine in the Fraser River estuary marsh sites during the spring of 1992.

Species and stage	Location	Site			Mean
		Transplanted	Disrupted	Reference	
chum fry	South Arm:				
	Annacis	3.72	0.55	3.78	2.68
	Deas	2.44	5.92	1.72	3.36
	North Arm:	0.48	0.73	1.40	0.87
	Queens Reach:				
	Port Mann	2.06	0.81	1.13	1.33
	Mean	**2.18**	**2.00**	**2.01**	**2.06**
chinook fry	South Arm:				
	Annacis	2.81	2.15	2.48	2.48
	Deas	0.98	4.24	2.35	2.52
	North Arm:	0.70	0.92	1.18	0.93
	Queens Reach				
	Port Mann	1.30	1.70	1.74	1.58
	Mean	**1.45**	**2.25**	**1.94**	**1.88**
sockeye fry	South Arm:				
	Annacis	0.00	0.00	0.32	0.11
	Deas	0.00	0.00	0.00	0.00
	North Arm:	0.00	0.00	0.00	0.00
	Queens Reach:				
	Port Mann	0.66	0.08	0.19	0.31
	Mean	**0.17**	**0.02**	**0.13**	**0.10**
chinook smolt	South Arm:				
	Annacis	0.19	0.65	0.51	0.45
	Deas	0.00	2.02	0.25	0.76
	North Arm:	0.00	0.53	0.43	0.32
	Queens Reach:				
	Port Mann	0.17	2.90	1.27	1.45
	Mean	**0.09**	**1.53**	**0.62**	**0.74**
sockeye smolt	South Arm:				
	Annacis	0.62	0.00	0.00	0.21
	Deas	0.00	4.00	0.00	1.33
	North Arm:	0.00	0.09	0.00	0.03
	Queens Reach:				
	Port Mann	0.08	0.00	0.35	0.14
	Mean	**0.18**	1.02	**0.09**	**0.43**

CHAPTER 6

Chum Fry Residency and Stomach Content Analysis at Transplanted Marsh Habitat Sites in the Fraser River Estuary[5]

by

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INTRODUCTION

Significant numbers of chum salmon (*Oncorhynchus keta*) fry are known to utilize the Fraser River estuary as a nursery area after migrating downstream as fry from February through May each year (Healey 1982; Salo 1991). Urban expansion and industrial development within the last century have altered the Fraser estuary area extensively and diking, siltation and reclamation have destroyed much of the marsh habitat available to salmonids (Chapter 2, present document).

One of the major objectives of the Fraser River Action Plan (Environment Canada/Fisheries and Oceans 1992) is the restoration of fish habitat, including disrupted marsh habitat which can be replanted. In the past, the Department of Fisheries and Oceans (DFO) has participated in projects to restore a variety of marsh habitats in the Fraser estuary. A number of these marshes (Chapter 1, present document) have been grouped with neighbouring reference (natural) marsh areas and disturbed (disrupted) marsh habitat areas in order to investigate the effectiveness of the restoration activities.

This study was designed to assess chum fry use and short-term residency at disturbed, reference and transplant sites in two areas of the Fraser delta through a mark, release and recapture study. More information on the sites is given elsewhere in the present document (see Table 1, Chapter 1, present document).

Study Area

North Arm

Work in this area was completed between April 13 to 22, 1993. Both the AN1 and DI2 transplant sites lay adjacent to the North Arm channel and consisted of riprap sheltering an area of transplanted marsh habitat. The AN3 transplant site consisted of a sheltered bay created by excavation of a human-made island. Within the bay, the substrate was primarily sand (>70% cover), deposited as a low profile bar, with some patches of marsh vegetation (<30% cover). The reference site (AN2) consisted of a natural vegetated marsh habitat dropping off to an unconsolidated mud/silt bank, sheltered from the main channel by a series of log pilings used for chip-barge mooring. The AN4 disturbed site was a typical non-vegetated sand/silt beach directly exposed to the main channel of the North Arm.

South Arm

Two reference sites were studied in this area: the DE2 reference site consisted of a riprap bank sheltering vegetated marsh habitat, on the mainstem river, and the DE3 reference site, situated within the sheltered slough, consisted of a 5 to 10 m wide marsh habitat area without riprap protection. Also within Deas Slough was the DE1 transplant site which consisted of narrow (<5 m wide) vegetated marsh habitat consolidated by a riprap wall. The DE4 disturbed site sampled in this area lay on the southern beach point of the BC Ferries bay and was characterized by a gently sloping, non-vegetated sand/silt beach created by dredge sediment deposited on the point.

METHODS

Chum Fry Capture for Marking

Chum fry were captured for later release using a beach seine (15 x 2.5 m, 0.3 cm knotless web). Every effort was made to capture 500 fry at the study sites or as close to the study sites as possible. Figures 1 and 2 indicate the extent of collection along the river banks for the two study areas, within which the study sites were located.

Seining effort involved one person on foot, pulling an end of the net out from the shoreline, while the bottom was on the substrate, to a depth of approximately 1.5 m. The net was moved downstream at this depth for 20 to 40 m (depending on the substrate and slope characteristics) perpendicular to the shore, whilst a second person followed the first along the shore, sweeping their end of the leadline along the shoreline. The first person then brought their end of the net into shore and the two ends of the net were brought in simultaneously with one person holding the two ends of the leadline together, to bag the catch. In areas where the bank sloped off too steeply to walk the net out, a 14-foot inflatable boat powered by a 15 hp motor was used to set and bring the net back in to shore, where it was then retrieved by hand in the manner described above.

Chum Fry Collected for Stomach Content Analysis

Chum fry were also collected at all nine sites for stomach analysis (Table 3). Fish were measured to the nearest millimetre before preservation, then were fixed in 10% formaldehyde for two to three weeks before being transferred into 20% isopropanol to facilitate later dissection.

In the West Vancouver Laboratory, the fork length and weight of the preserved fish were measured and each specimen assigned an identification number. The stomach was removed after an incision was made from the anus of the fish to its operculum. Fatty tissue and the pyloric caeca attached to the stomach were removed and placed back into the coelomic cavity of the fish.

The stomach and its contents were blotted dry with paper towel for 30 seconds and weighed on a Mettler H43 micro balance. After using a fine scalpel to cut an incision through the stomach wall, the contents were gently removed. The total stomach content weight was determined by weighing the empty stomach and subtracting the empty stomach weight from the total stomach weight. Contents were analyzed under a Wild Heerbrugg stereo microscope and identified, where possible, to order, family or genus (Borror and DeLong 1964; Borror et al. 1981; Borror and White 1970; Chu 1949; McAlpine et al. 1981; and Merritt and Cummins 1984).

In addition to partially digested food items, stomach contents included other organic and inorganic materials. Inorganic material consisted of clay, sand or pieces of plastic. Organic material was usually pieces of wood and plant. These items were weighed separately from the identifiable food items. The fish and stomach contents were placed into individual vials with 20% isopropanol and labelled with its respective identification number.

An indication of feeding activity at each site was investigated by comparing the stomach content weight using an analysis of covariance (ANCOVA; SAS Institute Inc. 1990). The ANCOVA also tested and corrected for the variation in fish weight (covariate) on stomach content weight.

Chum Fry Marking

Once caught, fish were transported to the shore base in herring buckets, aerated by a battery-powered aquarium air pump. At the shore base, the fish were transferred into 20-litre plastic buckets, with mesh windows cut into their sides to allow adequate water flow and aeration, which in turn were placed into a large (1.5 m^3) holding tank. A SCUBA tank provided aeration for the holding tank. Batches of 20 to 30 chum fry were then anaesthetized using tricaine methanesulfonate (MS222) at a concentration of 0.12 $g{\cdot}L^{-1}$ of estuarine water. Other fish were identified, counted and placed in a recovery bucket before release back into the river.

Chum fry were cold branded at the shore base. Three silver-tipped brands (dot, double-dot and bar), 2 to 3 mm in length, were used to mark one of four specific areas of the fish (right or left anterior and right or left posterior sides). Fry were branded just above the lateral line, either posterior or anterior of the dorsal fin. The brands were cooled to approximately -70°C in a dry ice/acetone bath. Up to ten fry could be marked before re-chilling the brand in the dry ice/acetone bath. A crew of three took approximately two to three hours to sort and mark 500 chum fry. Recovery from the anaesthetic normally occurred in less than two minutes. The marks used for the nine sites were as follows:

North Arm	
AN1 (Transplant)	right posterior dot (RP•)
AN2 (Reference)	left posterior bar (LP-)
AN3 (Transplant)	left anterior dot (LA•)
DI1 (Transplant)	left posterior dot (LP•)
AN4 (Disrupted)	right posterior bar (RP-)
South Arm	
DE3 (Reference)	right anterior bar (RA-)
DE1 (Transplant)	left anterior bar (LA-)
DE2 (Reference)	left posterior double-dot (LP••)
DE4 (Disrupted)	right posterior double-dot (RP••)

Marked chum fry were then held for approximately 24 hours in aerated river water in the holding tank to permit full recovery from the marking procedure. Water in the holding tank was changed every 24 hours. Dead fry were counted just before release to assess marking mortality (Table 1) and fish were inspected at random for mark formation. In all cases the marks were clearly visible before release at the sites. The mark retention results of this study concur with similar chum fry cold-branding work done in Alaska (Emmett and Convey 1991) with marks becoming increasingly darker, and therefore more visible, with time.

Chum Fry Release And Recapture

Between 372 and 547 chum fry were released at each of the nine study sites one hour before high tide. Table 1 summarizes the numbers of chum marked, mark mortalities and numbers of chum released at each site. For the North Arm, the high tides were in the early morning (between 03:00 and 05:10), and for the South Arm, the high tides were in the late afternoon/evening (between 16:20 and 19:40). Figures 1 and 2 indicate the locations within the study sites where the chum fry were released. Fish were transported to each site in 20-litre buckets and introduced slowly into shallow water (<1 m deep) so as not to stress the fish. The behaviour of released chum fry was observed immediately following release. In all cases fry appeared unstressed and generally headed for any available shelter or floating logs/debris.

The sites were then sampled one hour (at high tide), 3 hours (2 hours after high tide), 24 hours and 48 hours after release. Three overlapping seines (approximately 30% overlap) were deployed using the method described above, to cover the areas adjacent to, and including, the release points. Efforts were made to keep the seining procedure consistent and similar in extent for all the sites. For each seine sample, fish were sorted and chum fry examined for marks. To facilitate sorting and examination of larger catches, the fish were anaesthetized in the manner described above and placed in aerated 20-litre buckets on-site to recover before re-release. All fish were re-released at the sites less than 15 minutes after capture.

RESULTS

General

A detailed tabulation of seine catches for the study is provided in the Appendices of Burger (1993).

North Arm

The mark and release study in this area was conducted between April 14 to 22, 1993. A total of 2,847 chum fry were captured in all seines (142 seines in total) in the North Arm area, representing an average of 20 chum fry/seine. There was considerable variation in individual seine efforts with catches ranging from 0 to 341 chum fry/seine. A total of 60 seines were conducted at the five release sites combined for recapture purposes, yielding a total of 437 chum fry (average of 7 chum fry/seine) whilst 82 seines were conducted at non-release sites within the North Arm area (Figure 1), yielding a total of 2,410 chum fry (average of 29 chum fry/seine).

Mark release and recapture results for the five North Arm sites are summarized in Table 1. With one exception, mark mortality was less than 1%. The 12% mortality prior to release at the DI1 transplant site (Table 1) is believed to have been caused by contaminated water from a stormwater outlet upriver of the Oak Street Bridge inadvertently getting into the holding system when the fish were transferred to the holding tank. The remaining 372 fish which were released appeared "normal" in behaviour prior to release.

Some marked chum were recaptured one hour after release at all sites except the DI1 transplant site. The proportion of marked chum fry recaptured (as a percentage of total marked chum released) at the time of capture was low, ranging from 0% to 4.23% (see Figure 3a). The proportion of marked chum (as a percentage of total chum caught) caught after one hour ranged from 0% for the DI1 transplant to 59.4% for the AN3 transplant site. The highly sheltered nature of the AN3 transplant site is believed to have facilitated the comparatively high proportion of marked to unmarked chum sampled one hour after release. However, as the water in the bay of this site empties rapidly with the falling tide (as a result of the low profile sand bar that covers most of the bay), two hours after high tide only a few centimetres of water remained and the low chum fry catch (with no marked chum fry) is not surprising.

Marked chum fry were recaptured three hours after release at two of the sites -- AN1 transplant and AN4 disrupted -- although both the proportion of marked chum fry recaptured to total chum fry released (1.2% and 0.2%, respectively) and the proportion of marked to unmarked chum caught at the time (15.8% and 7.1%, respectively) were low.

No marked chum fry were caught at any of the five sites 24 or 48 hours following release. Additional seines were done at three of the release sites more than 24 hours after release (Table 2) to collect fish for stomach content analysis and no marked chum fry were recaptured in these seines either. No marked chum were captured in any of the seines conducted outside of the release sites during the study period in the North Arm area.

Catches of chinook fry fluctuated considerably throughout the field period in the North Arm area. The average number of chinook fry caught rose from 0.38 fry/seine on the April 14 to a high of 39.39 fry/seine on April 17 before dropping down to a low of 2.33 fry/seine on the April 29. However, this peak in numbers of chinook fry in the North Arm area may be artificial because sites with different physical characteristics (e.g., state of tide, substrate cover) were sampled in different parts of the North Arm during the same day.

Figure 5a shows the relative proportion of average chinook to average chum fry caught at the five study sites for all 12 recapture seines combined during the field period. Higher proportions of chum fry were captured at all sites except at the DI1 transplant site, which had a higher proportion of chinook fry. Figure 5a also shows that, on average, more chum fry were caught at AN2 reference than any of the other sites.

Analyses of catch by exposure and substrate from all capture and study sites in the North Arm showed a tendency for more chum being caught in sheltered areas with either cobble, mud or sand substrates. Areas with vegetated substrate (including all the seines done at AN2 reference, AN1 restoration and DI2 transplant sites) had the lowest average number of chum fry caught, although this may also be caused by reduced capture effectiveness when seining over a vegetated area.

South Arm

The mark and release study in the Deas Slough area was conducted between May 2 to 8, 1993. A total of 2,486 chum fry were captured in all seines (91 in total), representing an average of 27.3 chum fry/seine. There was considerable variation in catches, ranging from 0 to 313 chum fry/seine. Forty-eight seines were conducted in the four release sites combined, yielding a total of 333 chum fry (average of 6.9 chum fry/seine) whilst 43 seines were conducted at non-release areas within the Deas Slough area (Figure 2 shows the extent of seining effort in this area), yielding a total of 2,153 chum fry (average of 50.1 chum fry/seine).

Mark, release and recapture results for the four Deas Slough sites are summarized in Table 1. Mark mortality was low, ranging from 0.9% to 2.2%. Marked chum were recaptured one hour following release (at high tide) at all sites except the DE4 disrupted site, where no marked chum fry were recaptured at any time following release. The proportion of marked to unmarked chum was relatively high for both the DE2 reference (44.4%) and the DE1 transplant (40%) sites (see Figure 4b). However, the proportion of marked chum fry recaptured, expressed as a percentage of marked chum released, was low (0.8% and 4.14%, respectively) (see Figure 3b). Three hours following release, (high tide + 2 hours) both Deas Slough sites showed a similar pattern with relatively high proportions of marked to total chum caught (50% and 40%, respectively) but a low proportion of marked chum fry compared to total marked fry released (0.4% for both sites).

DE1 (transplant) site was the only site in both the North Arm and the Deas Slough areas where marked chum fry were recaptured 24 hours and 48 hours after release (see Table 1). At each time, only a single individual was recaptured (representing 0.2% of total chum fry released). In addition, two marked individuals released at the DE3 reference site were recaptured 24 hours later at the DE1 (transplant) site in Deas Slough, representing the only trans-site migration of chum fry noted in the study. No marked chum fry were recaptured at the DE3 reference, DE2 reference or DE4 disrupted sites 24 or 48 hours following release, and no marked chum were captured in any of the seines conducted at stations other than the release sites.

The proportion of chinook to chum fry caught in the Deas Slough area did not show any pattern with time as in the North Arm. The ratio of chinook to chum fry caught over the field period was close to 2:1. Figure 5b shows the relative proportion of average chinook to chum fry caught at the four sites for all recapture seines. Higher proportions of chinook to chum fry were found at all the sites, although at site DE2 this difference was negligible and the ratio was almost exactly 1:1. The DE4 disrupted site had the highest average catch of chum and chinook fry of all four sites.

Analyses of catch by exposure and substrate for the Deas Slough area as a whole showed that more chum were caught in sheltered, mud substrate areas than for any other combination of exposure and substrate type. Sheltered, vegetated areas (which included the DE3 reference, DE1 transplant and DE2 reference sites) had the lowest average catches of chum fry.

Stomach Content Analysis

A total of 119 chum fry were collected for stomach content analysis from all nine sites. There appeared to be little difference in size composition of fish from the various sites (Table 3).

Chum fry stomach contents were analyzed and identified for 45 fish from the North Arm and 47 fish from the South Arm (Deas Slough area). A total of 50 taxa of various invertebrates at different life stages were observed in the stomachs of fry from both reaches; 37 items were observed in the North Arm fry and 48 items were observed in the South Arm fry. Harpacticoid and cyclopoid copepods; chironomid larvae, pupae and adults, cladocerans and aphids comprised (by number) 63.6% and 65.9% of the North and South Arm chum fry stomach contents, respectively (Figures 6 to 12).

Harpacticoids were the most prevalent food item in both reaches (Figure 6). In the North Arm, twice the number of harpacticoids were observed in the chum stomachs at transplanted sites AN1 and AN3 compared to AN4 disrupted and AN2 reference. The chum captured at DE1 transplant site, in the South Arm, had consumed a higher number of harpacticoids compared to the disrupted and reference sites. Although the above mentioned food items comprised the major groups observed in the chum fry stomachs, the number of fish captured was small (<5) at sites DI1 transplant and DE2 reference. This was a problem as the data could not be statistically tested for differences.

Significant spatial difference (ANCOVA, $p<0.001$) was observed in total stomach content weight by site in the South Arm fry (Table 4). Fish captured at disrupted sites had a lower stomach content weight than those collected from reference and transplant sites. No significant difference was observed in the North Arm fry. Again, small sample sizes may have influenced the results and, therefore, should be regarded cautiously.

DISCUSSION

Chum fry were present at all nine sites during the study. However, of the nine sites examined, only one site, AN2 (reference) in the North Arm, appeared to support consistently high numbers of chum fry at high tide.

All of the marsh areas were completely exposed during low tides. Chum fry would therefore be forced off the vegetated marsh habitats and into the mainstream or adjacent non-vegetated areas. The presence of chum fry on the marsh habitats at high tide, as shown by this study, would suggest that fry are utilizing these areas for feeding and/or refuge.

Residency of marked chum fry at all the sites was short-term. The results of this study showed that, in general, some chum fry stayed in the area of release over the high tide immediately following release but had moved away from the release sites after a full tidal cycle. The DE1 transplant site was the only site at which marked fry were recaptured 24 hours and 48 hours following release (in both cases only a single chum fry was caught). The DE1 transplant site was also the only site where marked chum fry (two in total) were caught which had been released at another site (DE3 reference site, more than 1 km "downstream" in the slough), suggesting that chum fry may reside in Deas Slough for up to 48 hours.

On average, larger numbers of chum fry were caught in sheltered areas regardless of substrate type. Chum fry did not appear to favour vegetated marsh habitats over any other habitats; on the contrary, chum fry catches in vegetated areas were the lowest for both the Queens Reach and the Deas Slough areas. However, availability of marsh habitat relative to unvegetated shorelines may affect this relationship and this requires further study.

Although a diverse range of food items were observed in the stomachs of chum fry collected in the North and South Arms of the Fraser River, the contents were mainly comprised of a select group of invertebrates. This may reflect the abundance of particular invertebrates found in these areas. In 1992, epibenthic sled samples of the invertebrate community found harpacticoid copepods to be the most prevalent taxonomic group at the same areas (Boyle et al. 1995; Chapter 4, present document). Healey (1979) found harpacticoids made up 50% of juvenile chum diet in the Nanaimo estuary. Due to the small number of fish collected for stomach analysis, it was not possible to test for differences in feeding strategies at transplanted, disrupted and reference sites. However, fish were feeding at the transplant sites, indicating the invertebrates present were being consumed.

This study highlights the importance of careful planning when constructing habitats for marsh transplants. The AN1, AN3 and DE1 transplant sites, as well as the DE2 reference site, all had high elevations with relatively low profiles so that their marsh vegetation was only available to salmonid fry for short periods of time during high tides. The AN2 reference site, which had a slightly steeper slope and lower elevation overall was submerged for longer periods, providing more access to fish.

A minor shortcoming of the study design was that up to three sites had to be sampled on a single high tide cycle, which forced the field crew to sample at least two of the sites at times not exactly on high tide. Further work on salmonid fry residency at fish habitat sites in the Fraser River estuary could concentrate on fewer sites, allowing sampling to be conducted more precisely during the high tide period. Increasing seining effort relative to site size (total area) could also provide better within-site salmonid fry residency patterns. Intensified release of marked salmonid fry (successive high tide release of differentially marked fry, for example) at any site could also improve our understanding of salmonid fry residency patterns in the Fraser River estuary.

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Table 1. Summary of chum fry mark and recapture results for the North Arm and South Arm area (three seines were made at each site at each time interval).

						Recaptures:											
						At High			High +2H			+24H			+48H		
Site	CHUM MARKED	MORTALITIES	% MORTALITY	CHUM RELEASED	Date and Time of Release	TOTAL CHUM	MARKED CHUM	% MARKED	TOTAL CHUM	MARKED CHUM	% MARKED	TOTAL CHUM	MARKED CHUM	% MARKED	TOTAL CHUM	MARKED CHUM	% MARKED
North Arm																	
AN1 Transplant	496	2	0.4	494	16/4 0200	13	4	30.8	38	6	15.8	9	0	0.0	2	0	0.0
AN2 Reference	510	0	0.0	510	17/4 0305	137	22	16.1	68	0	0.0	17	0	0.0	58	0	0.0
AN3 Transplant	449	0	0.0	449	18/4 0315	32	19	59.4	5	0	0.0	6	0	0.0	11	0	0.0
DI1 Transplant	422	*50	12.0	372	19/4 0322	1	0	0.0	6	0	0.0	1	0	0.0	1	0	0.0
AN4 Disrupted	551	4	0.7	547	20/4 0410	13	1	7.7	14	1	7.1	1	0	0.0	5	0	0.0
South Arm																	
DE4 Disrupted	464	4	0.9	460	3/5 1520	8	0	0.0	19	0	0.0	94	0	0.0	21	0	0.0
DE2 Reference	502	6	1.2	496	4/5 1648	33	7	21.2	53	2	3.8	3	0	0.0	1	0	0.0
DE3 Reference	511	11	2.2	500	5/5 1738	9	4	44.4	4	2	50.0	8	0	0.0	8	0	0.0
DE1 Transplant	543	12	2.2	531	6/5 1840	55	22	40.0	5	2	40.0	37	1	2.7	16	1	6.3

* believed to be caused by contaminated water from a collection site getting into holding system.

Table 2. Additional seines completed in the North Arm.

Site	# Seines	Hours after release	Total Chum	Marked Chum	Total Chinook
AN1 Transplant	1	149	0	0	1
DI1 Transplant	2	76	3	0	24
AN4 Disrupted	1	37	9	0	0

Table 3. Numbers and size ranges of chum fry collected for stomach analysis at the nine study sites.

Site	n	Range (mm)	Mean	S.D.
AN2 Reference	11	37-45	40.6	0.83
AN1 Transplant	5	39-41	40.2	2.65
AN3 Transplant	11	37-56	42.3	5.26
DI1 Transplant	3	41-42	41.7	0.58
AN4 Disrupted	14	39-52	42.0	3.11
North Arm (Total)	**44**	**37-56**	**41.5**	**3.42**
DE3 Reference	12	37-44	40.2	1.64
DE1 Transplant	40	37-57	42.0	5.14
DE2 Reference	4	39-42	40.3	1.25
DE4 Disrupted	19	36-47	40.5	2.65
South Arm (Total)	**75**	**36-57**	**41.4**	**4.14**

Table 4. Mean stomach content weight ± SEM of chum fry collected in the North and South Arms of the Fraser River estuary.

Site	n	Mean (mg)	± SEM
North Arm			
AN1 Transplant	5	4.845	1.414
AN2 Reference	12	1.638	0.931
AN3 Transplant	11	3.622	0.958
AN4 Disrupted	14	4.057	0.842
DI1 Disrupted	3	4.824	1.838
South Arm			
DE1 Transplant	10	9.161	1.226
DE2 Reference	4	7.294	1.970
DE3 Reference	14	5.108	1.036
DE4 Disrupted	19	2.316	0.892

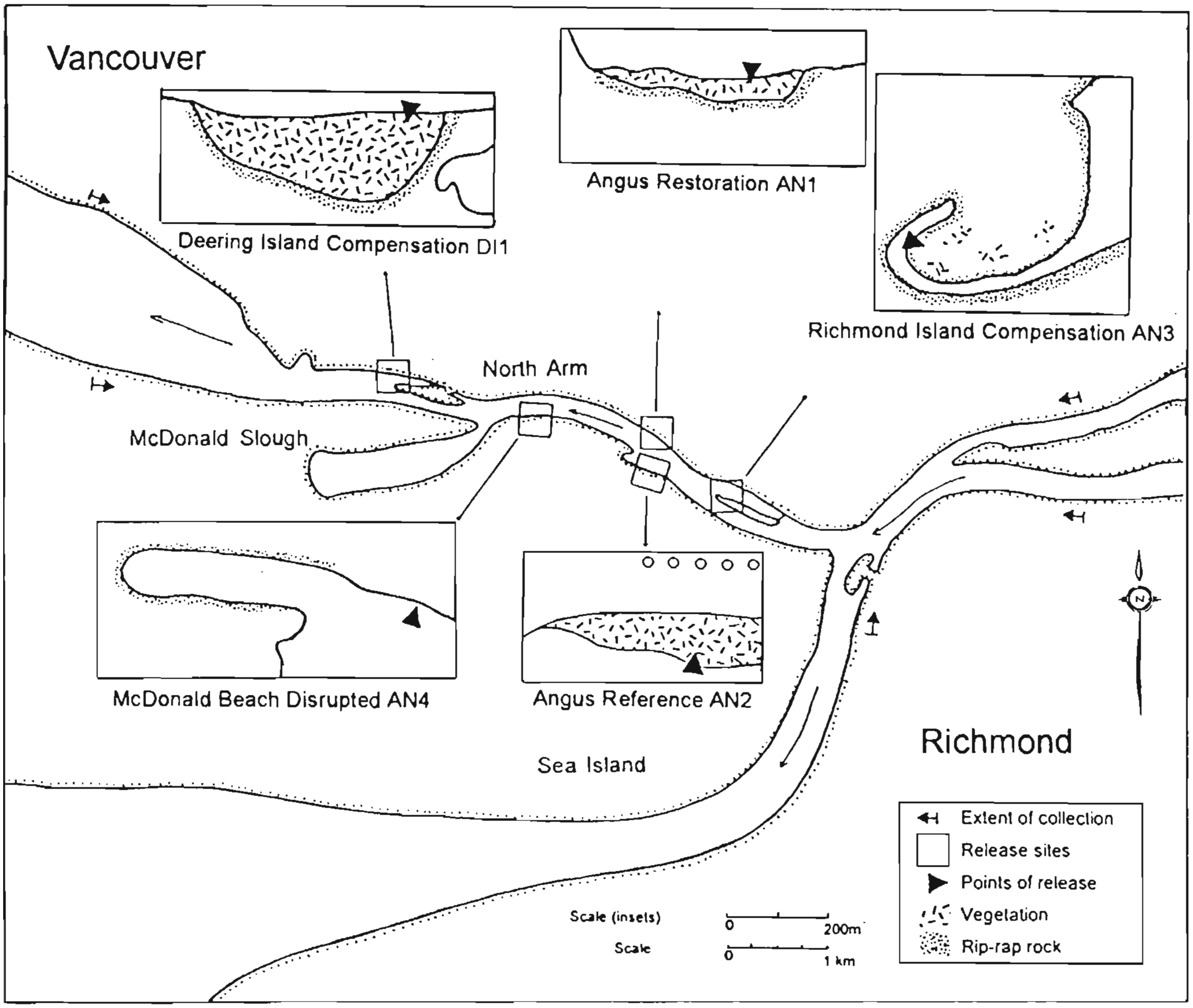


Figure 1. Position of the five release sites in the North Arm (fish were captured between the arrows on the main figure). Note: this is a schematic diagram.

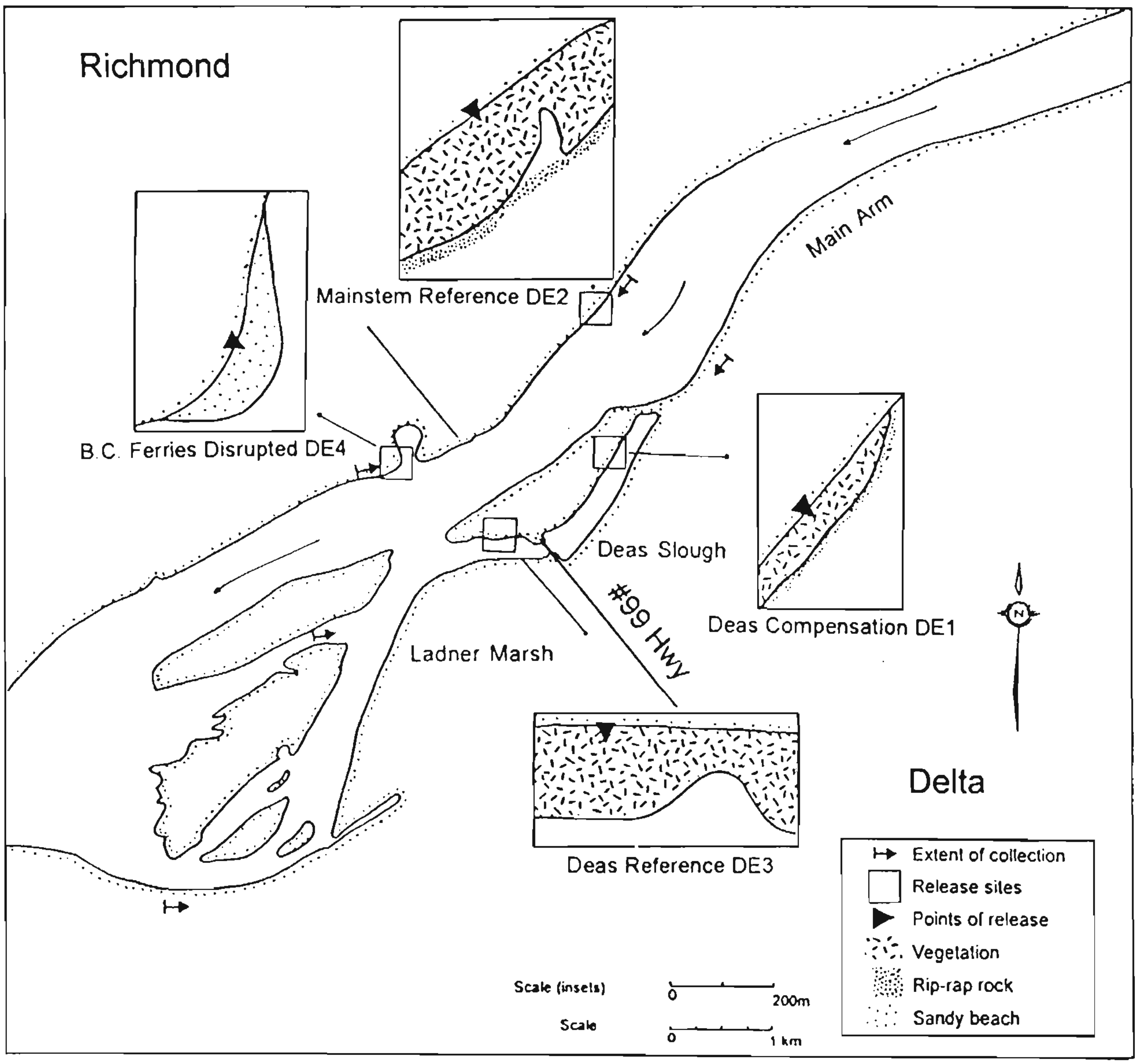


Figure 2. Position of the four release sites in the South Arm, captured between the arrows on the main figure). Note: this is a schematic diagram.

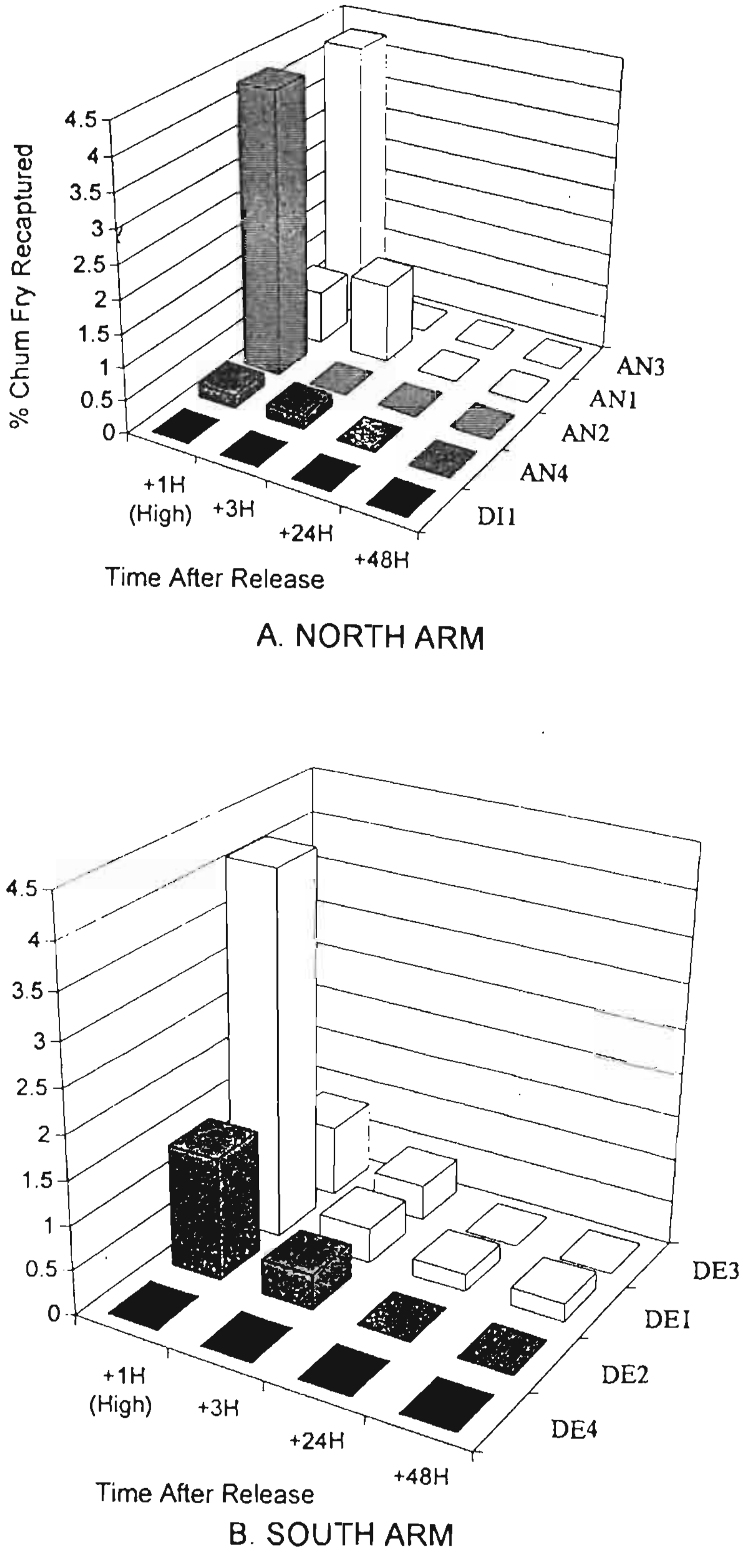
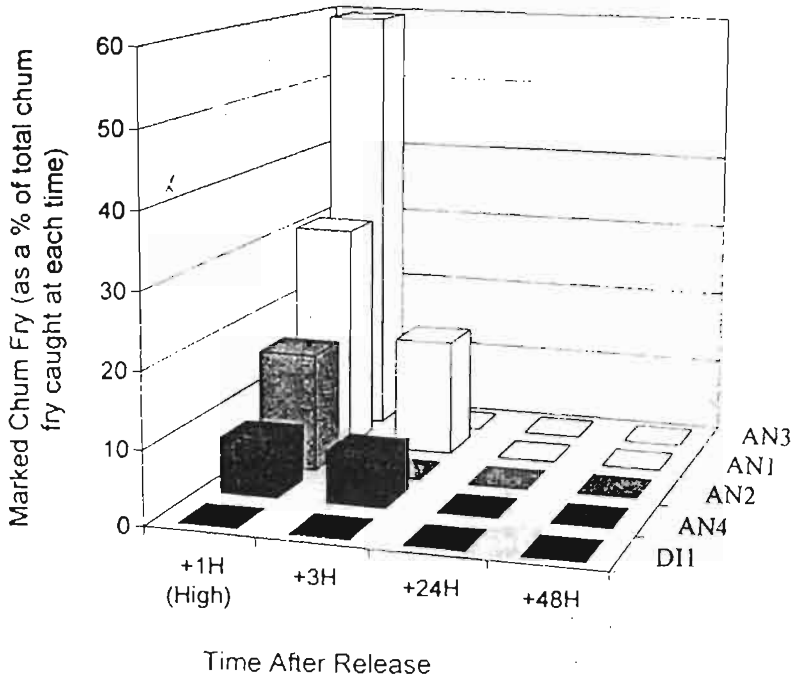
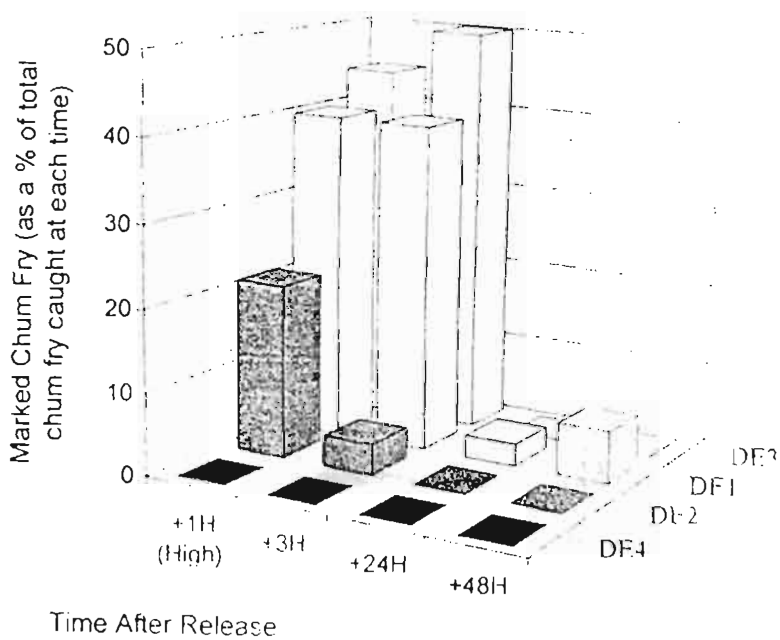


Figure 3. Percentages of marked chum fry recaptured (as a % of total chum fry released at each site at each time interval) for the North Arm (A) and South Arm (B).



A. NORTH ARM



B. SOUTH ARM

Figure 4 Percentages of marked chum fry recaptured (as a % of total chum fry captured at each site at each time interval) for the North Arm (A) and the South Arm (B)

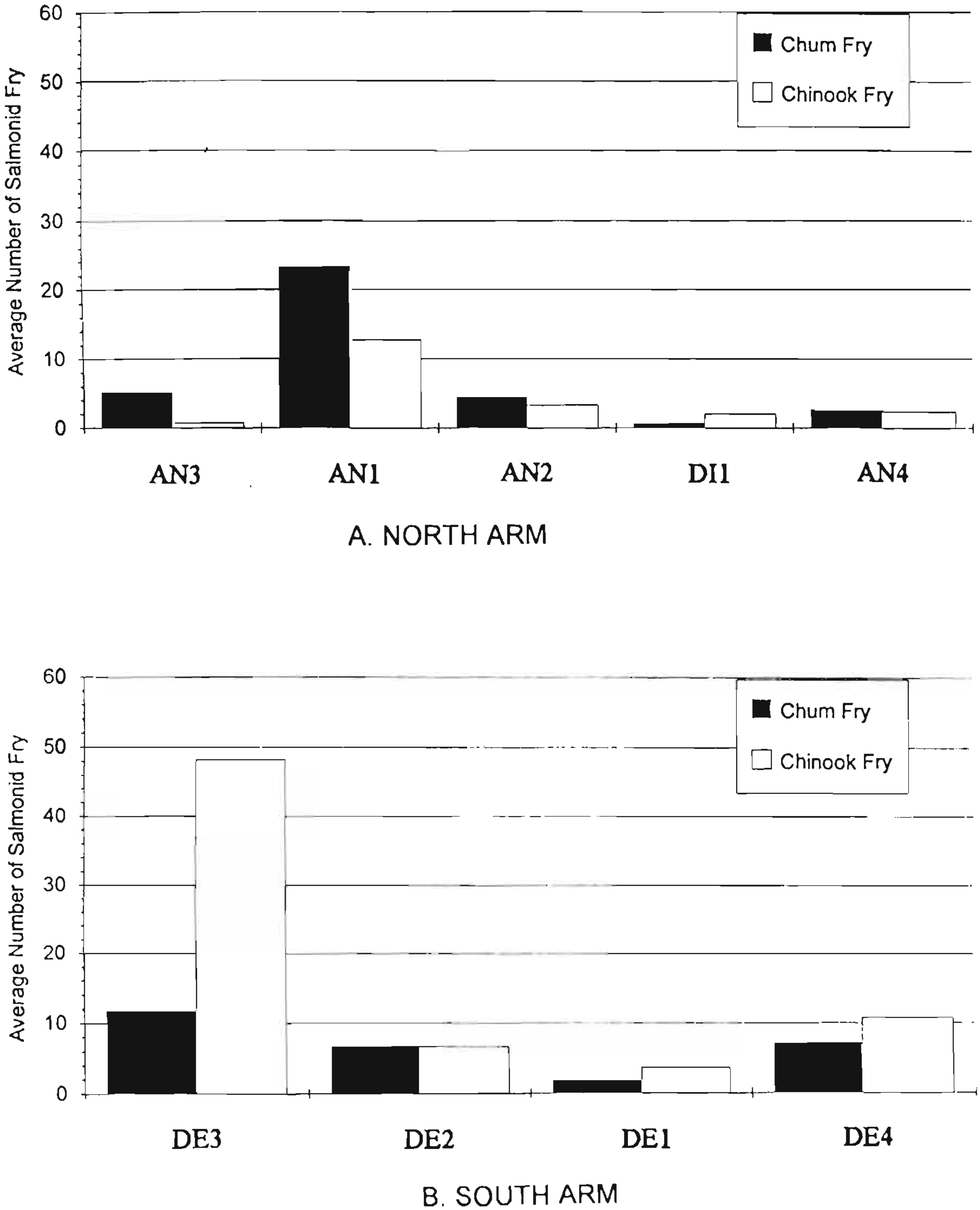


Figure 5. Average numbers of salmonid fry caught in 12 seines at study sites in the North Arm (A) and the South Arm (B)

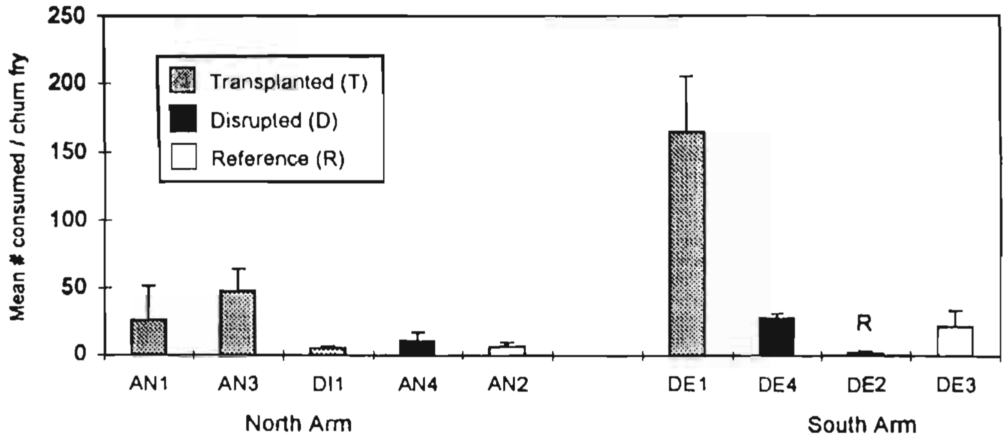


Figure 6. Mean number of harpacticoid copepods per chum fry collected at transplant, disrupted and reference sites in the North and South Arms, Fraser River estuary (see Table 3 for number of fry examined).

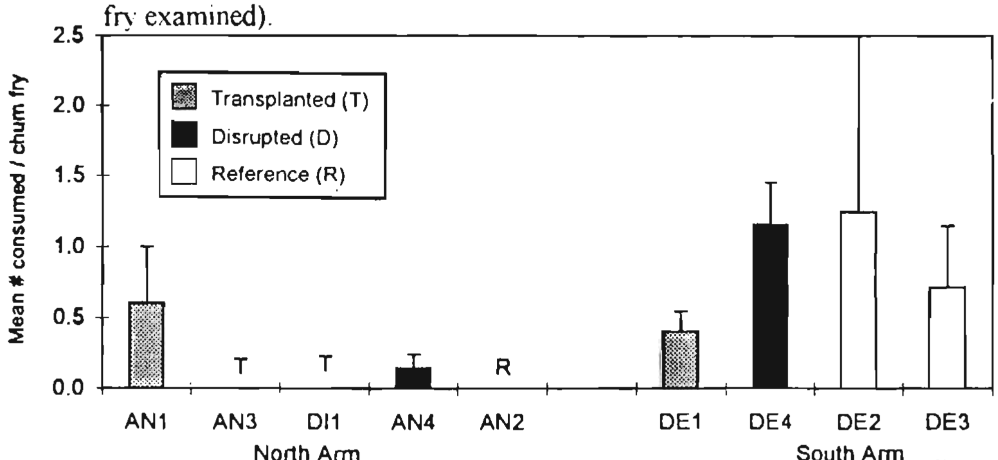


Figure 7. Mean number of cladocerans per chum fry collected at transplant, disrupted and reference sites in the North and South Arms, Fraser River estuary (see Table 3 for number of fry examined).

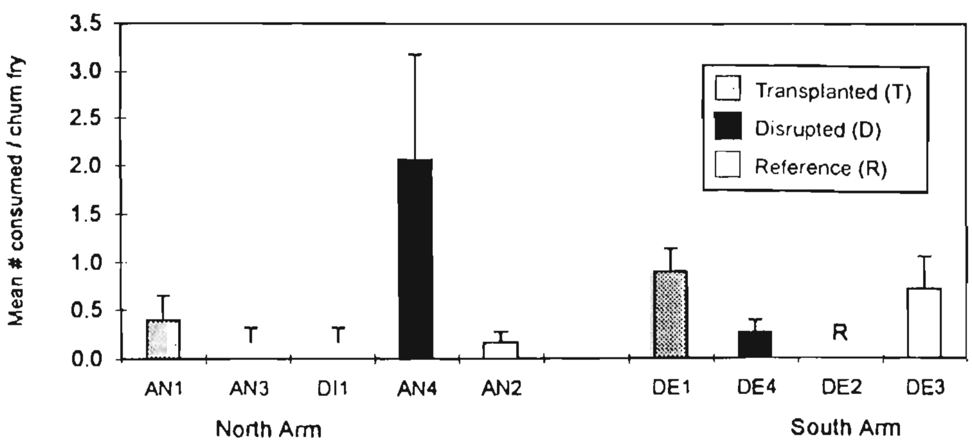


Figure 8. Mean number of aphids per chum fry collected at transplant, disrupted and reference sites in the North and South Arms, Fraser River estuary (see Table 3 for number of fry examined).

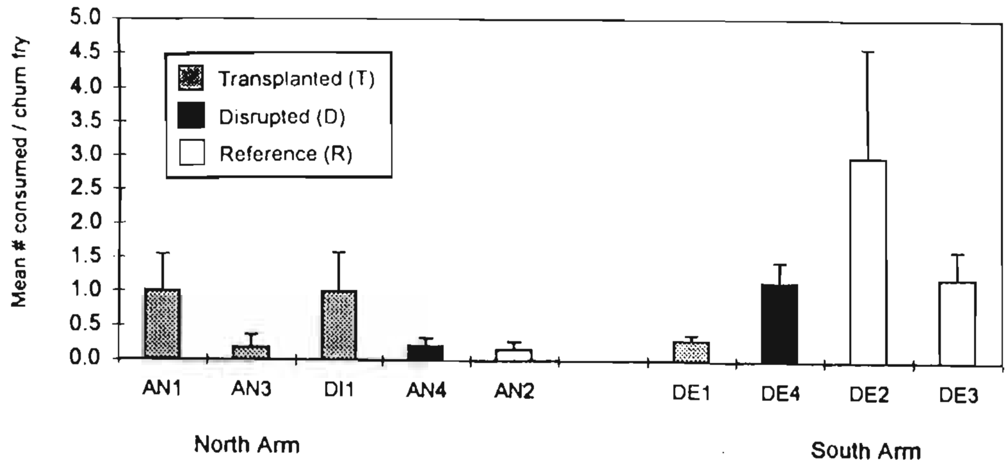


Figure 9. Mean number of chironomid larvae per chum fry collected at transplant, disrupted and reference sites in the North and South Arms, Fraser River estuary (see Table 3 for number of fry examined).

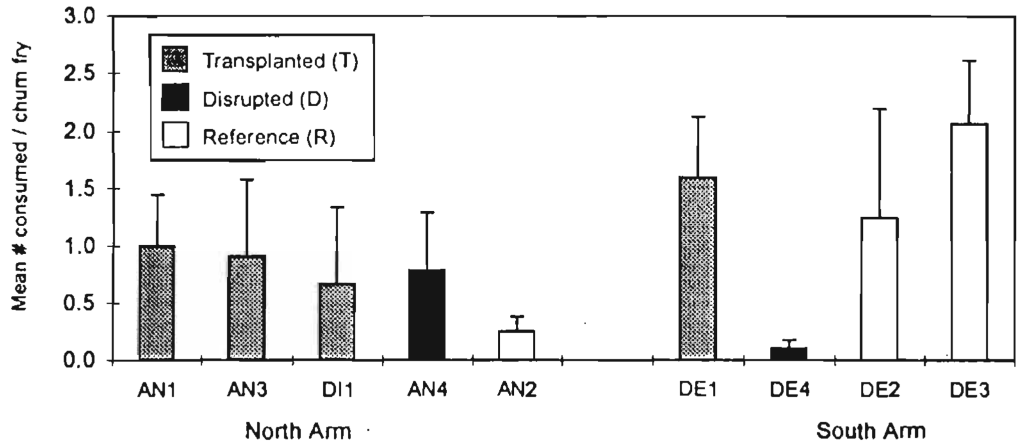


Figure 10. Mean number of chironomid pupae per chum fry collected at transplant, disrupted and reference sites in the North and South Arms, Fraser River estuary (see Table 3 for number of fry examined).

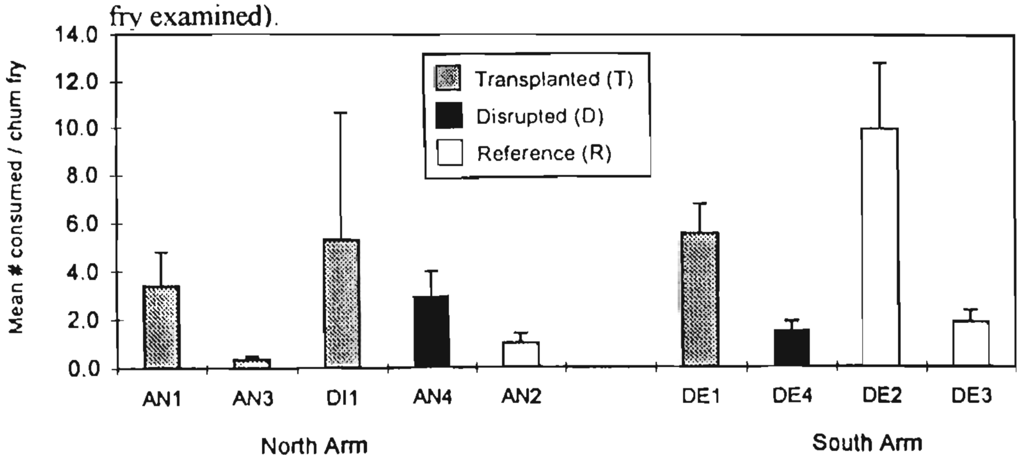


Figure 11. Mean number of chironomid adults per chum fry collected at transplant, disrupted and reference sites in the North and South Arms, Fraser River estuary (see Table 3 for number of fry examined).

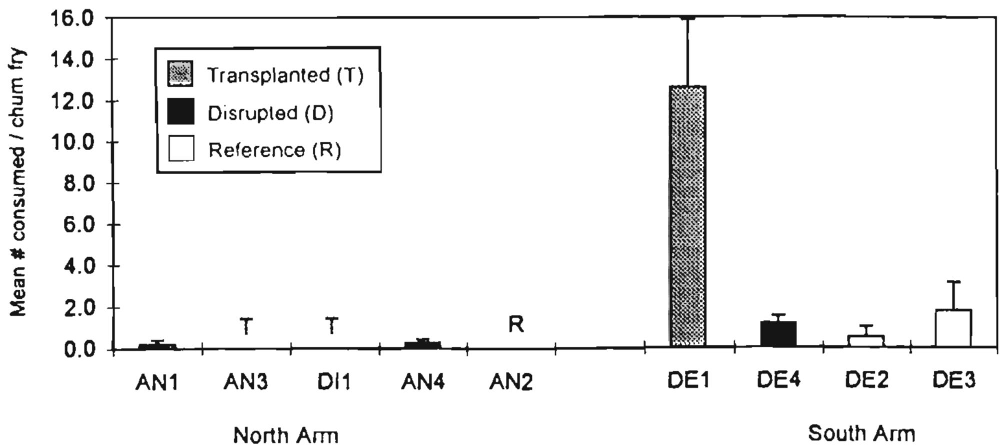


Figure 12. Mean number of cyclopoid copepods per chum fry collected at transplant, disrupted and reference sites in the North and South Arms, Fraser River estuary (see Table 3 for number of fry examined).

CHAPTER 7

Synthesis and Conclusions

by

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SYNTHESIS AND CONCLUSIONS

As the present study only dealt with selected elements of the estuarine ecosystem, our study results must be accompanied by several major caveats. First of all, we recognize that coastal wetlands are very complex habitats (e.g., Zedler and Powell 1993) and therefore, an understanding of all the ecological functions of the transplanted marshes would be well beyond the limitations of our study. For example, we did not evaluate if nutrient dynamics of the transplanted marshes were similar, as was investigated by Langis et al. (1991) and Craft et al. (1988, 1989). In addition, because the marshes we evaluated were transplanted between two and seven years ago, time was a confounding factor in this assessment. The following will focus on functional attributes of the transplanted marshes which are important for fish, particularly:

I. geomorphological aspects of the marsh which may affect fish use, including submergence/emergence, perimeter, surface area and stability;

II. provision of organic material by marsh plants for use in detrital food webs;

III. refuge function afforded by marsh plant community;

IV. availability of fish food invertebrates; and,

V. use by the estuarine fish community and residency by juvenile chum and chinook salmon.

A final section will comment on risk of failure and a recommendation for integration of management, science and monitoring.

I. Geomorphology

Submergence

Mean submergence for each of the marshes was computed for spring-summer (May to August) and winter (January to March) runoff and tidal conditions using survey data given in Chapter 3 (present report) and modelling procedures as described in Stronach (1995). Results for spring-summer, when juvenile salmon are rearing in the river, are given in Table 1. The model developed by Stronach (1995) may be useful for planning future marsh restoration projects as the method can be used to predict submergence/emergence at any particular site in the lower river where elevation has been determined. The method takes into account water level changes due to river discharge as well as tidal fluctuations. An example of the graphic presentations that can be produced from the model is given in Figure 1. Models have also been published for designing the hydrology of restored intertidal salt marshes (*Salicornia virginica*) in San Francisco Bay (Coats and Williams 1990) and may have some application in our region.

Site PM1 (see Figure 4, Chapter 1) was submerged 26.4% during spring-summer, suggesting fish would have much less access to resources at this site relative to adjacent natural habitats or the other transplanted marshes surveyed in the study. Mean submergence for the transplanted areas were generally more variable, ranging from 26.4 to 60.1%, compared to a range of 33.2 to 50.7% for natural marshes (Table 1).

It should be noted that our surveys did not assess the presence or extent of tidal channels in the habitats examined and therefore, our analysis of submergence is an underestimate of the total area flooded in a particular season. Estuarine and freshwater tidal channels have been shown to be important habitat in the lower Fraser River (Levy and Northcote 1979; Levings et al. 1995). Created tidal channels can add significant fisheries value to transplanted marshes by increasing perimeter (e.g., Levings and Macdonald 1991) or edge habitat (e.g., Minello et al. 1994) for feeding.

Surface Area and Perimeter

The transplanted habitats we evaluated ranged from 53 to 7,000 m^2, with a mean area of 1,468 m^2 (Kristritz 1996). A detailed quantitative analysis of surface area and perimeter gain or loss was not attempted in the study. Observations during field work showed that transplanted marshes were generally lacking in edge habitat along small tidal channels that characterize natural marshes. One exception was the oldest marsh evaluated (seven years; AN1) which showed a few small drainage channels. It is likely that these channels developed through erosion due to runoff (Ranwell 1973) and therefore are a natural feature in the evolution of marsh habitats. Other exceptions were created embayments (AN3 and AC2). Because of the circular shape of these areas, significantly more perimeter area was added relative to the simple "platform" habitats developed at the other sites. However, it should be noted that at site AC2, where an embayment was constructed by dredging into natural riparian habitat, there was a loss of edge habitat. Other exceptions were the channels cut into existing marsh near McDonald Slough (MD2 and MD3). Created channels were found to support juvenile salmon in Washington (Miller and Simenstad 1993) and results of a marsh restoration project in Texas explicitly suggested that channelizing a created salt marsh could improve ecological values of the habitat (Minello et al. 1994).

At most sites in our study, the majority of the edge habitat was on the "deep water" side of the marshes because the seaward perimeter was often characterized by a change in elevation owing to the presence of protective structure such as riprap. These edges would be usable by grazing fish for a relatively short period of time as water levels rose and fell.

Stability

As observed in previous studies concerning marsh restoration in the lower Fraser (e.g., Envirocon 1980) and elsewhere (e.g., USCE 1988), currents and wave action are key factors which can influence the stability and longevity of transplanted marshes. Envirocon (1980) used a small-scale physical model to evaluate the wave and current energy regime of proposed marsh creation sites in the lower Fraser. In the present study, Stronach (1995) presented data on the average current velocity at a particular elevation on transplanted and natural marshes, based on the cross-sectional averaged flow computed by a numerical model. The results, which do not take into account small-scale local conditions, showed that the predicted currents at the lowest elevations of transplanted marshes were within 0.2 $m \cdot s^{-1}$ of those at the corresponding reference sites (range 1.0 to 0.5 $m \cdot s^{-1}$; from Stronach 1995).

Five of the transplanted marsh habitats we surveyed were protected from river wave and current erosion by riprap revetment and their platforms might therefore be considered stable in the medium term. The two embayments examined (AN3 and AC2) were not provided with revetment. At AC2, the slope of the shore around the perimeter of the embayment was steeper (est. 1:10) than the natural angle of repose in low-energy areas, which is usually about 1:20 in the Fraser estuary (e.g., Williams 1993). This steep slope was the result of dredging to achieve a relatively deep embayment. The embayment was dredged to a depth of approximately -2 m relative to local chart datum (CD) and so the total length of the tidal gradient was relatively long. The embayment at AN3 was relatively shallow, with bottom at +2 m CD and the shoreline consisted of hard substrate such as rock and concrete rubble. There was little evidence of slumping at this site. Bank stability around the shorelines of created embayments and channels could be improved by planting riparian vegetation such as willows (*Salix* spp.) as recommended by various authors (e.g., Adams and Whyte 1990; Shields et al. 1993)

As documented in the first marsh transplants in the Fraser estuary (Pomeroy and Levings 1980) it is likely that protection from erosion and impact of logs and other debris is a factor influencing the survival of newly transplanted sedge marshes. Gouging of marsh substrate from floating wood debris was noted as a factor causing erosion at site AN1. Where protective shear booms were installed and functioning (sites DI1, AC2 and AC4), damage from debris was not as evident. Sediment instability due to wave and current energy at a transplanted marsh near Mitchell Island on the North Arm (Williams 1993) led to partial failure of the project (personal observations). Arrangement of log booms offshore of the site may have alleviated the problem.

II. Provision of Organic Material by Marsh Plants for Use in Detrital Food Webs

One of the most important functions of estuarine marshes is provision of organic material for use by heterotrophic bacteria in fish food webs (e.g., Sibert et al. 1977). To assess this aspect, in addition to cover estimates (Moody 1993, Chapter 3, present document), above-ground vegetation was clipped in August 1991 within three replicate quadrats (0.75 x 0.75 m), randomly selected at several of the transplanted and natural sites. In the laboratory, the vegetation was washed to remove excess mud and sand, damp dried and then weighed. The material was dried for 24 hours at 100°C and then reweighed.

Analyses of cover data from all sample sites showed that cover values for reference marshes were approximately 87% of those of transplant sites, with ratios of reference:transplant ranging from 1.31 at CN2 vs. CN1 to 0.71 at AC1 vs. AC2 (Table 2). However, there were major differences in taxa accounting for the cover. Further data are therefore needed on the qualitative differences in the detritus arising from the various plant taxa. Correlations between taxa cover estimates at reference and transplant sites ranged from 0.11 at AC1 vs. AC2 to 0.91 at AN2 vs. AN1. Overall, the correlation was only 0.56 (Table 2). Stronach (1995) resolved cover data from the study into several species for each major plant family. Therefore, conclusions from this correlation should only be drawn after consideration of the detailed species list involved. Figure 2 showed that much of the difference was due to the preponderance of two species of *Juncus* at transplant sites compared to reference sites. A preliminary review of the designs of the transplant projects (Kistritz 1996) showed that many of them targeted the sedge *Carex lyngbyei*. However, cover estimates for this taxa at transplant sites were generally lower compared to reference areas (Figure 2).

Results from clippings showed that above-ground biomass in August, which was an approximation of the annual net primary production of the marsh community, varied significantly between the various sites. Except for site AC2, above ground biomass was >1000 g dry wt·m^{-2} , at all transplanted and natural marsh sites evaluated (Figure 3), which was within the range of the annual peak biomass of natural sedge marshes assessed earlier in the estuary by Kistritz et al. (1983.). There did not seem to be any relationship between age of the transplanted marsh and biomass. Site DI1 (two years old) showed significantly higher biomass than site AN1 (6 years old). Physical stability seemed to be a factor influencing biomass. Site AC2, characterized by a very steep slope (see above), showed the lowest standing crop assessed in the survey.

III. Refuge Function

There are few data showing how vegetation in north-east Pacific estuaries provides refuge for juvenile salmonids and so assessment of plant function is difficult. It is possible that the refuge function only becomes available to fish when stem density reaches a critical level or that certain vegetation communities offer more refuge than others. Near Chilliwack, northern squawfish (*Ptychocheilus oregonensis*) caught in nearshore vegetation contained few juvenile salmon (Gregory et al. 1993; and unpublished data). Tompkins and Levings (1991) as well as Gregory and Levings (1995) showed, using laboratory experiments, that juvenile salmon can avoid predation by hiding in vegetation,. However, this work did not include tests or field work to evaluate the significance of plant stem density as a variable affecting fish function. Empirical data on this topic are needed and could be obtained by diver observations in a tributary characterized by low turbidity levels such as the Harrison River.

IV. Availability of Fish Food Organisms

Numerous factors (e.g., size, colour, behaviour) influence the choices that juvenile salmon make when "deciding" to eat prey (Levings 1994) but it is clear that abundance of prey is a key index of availability. At most of the transplanted sites assessed, invertebrate communities were similar to natural sites and, with a few exceptions, supported densities of fish food organisms that were higher than those at unvegetated habitats (Chapter 4, present document). These findings were also reported in earlier preliminary work at another created marsh on the Fraser near Mitchell Island on the North Arm (Whitehouse et al. 1993). The colonization rate of the various invertebrate taxa is dependent on the ecology of the particular taxa involved. For example, free-swimming gammarid amphipods can move into a habitat almost immediately (Levings and Macdonald 1991; Batzer and Resh 1992) whether vegetation is present or not. An earlier study in the Fraser River estuary showed that adult and larval chironomids were clearly more abundant in vegetated habitat relative to sand and mudflats (Whitehouse et al. 1993). At a transplanted marsh in the northeastern U.S., elevation (Fell et al. 1991) and plant density were found to be an important factor for colonization by snails (Peck et al. 1994). Other taxa, such as polychaetes, are dependent on pelagic larval stages for colonization (Levings and Macdonald 1991; Niesen and Lyke 1981). At Miller Sands on the Columbia River, age of the transplanted marsh, as well as seasonal cycles in invertebrate abundance, were identified as factors influencing the composition of the fish food communities (Newling and Landin 1985). At Miller Sands, a four-year-old transplanted marsh was found to be very similar in terms of species composition, relative to a reference marsh. However, the most abundant invertebrate taxon in the transplanted marsh was oligochaete worms. These are sometimes reported as food for juvenile salmon in estuaries (e.g., Anderson et al. 1982 in the Fraser River estuary). They, therefore, should be considered potentially available as fish food but in the Columbia River marsh were most numerous at a high elevation in the marsh where availability to the fish was likely low.

V. Use by the Estuarine Fish Community and Residency by Juvenile Chum and Chinook Salmon

Fish Communities

The species composition of fish communities using the transplanted marshes was similar to that of reference areas (Scott and Susanto 1995; (Chapter 5, present document). There was no evidence that transplanted marshes were avoided by juvenile salmon or dominated by non-salmonids such as cyprinids. Our study did not examine seasonal changes in fish communities. Studies of fish communities on seagrass beds established on dredged sand in Florida showed seasonal differences relative to natural habitats even in areas where initial colonization over 30 years before the studies were conducted (Brown-Peterson et al. 1993). However, these authors concluded that the recolonized seagrass meadows appeared to be as suitable a habitat as natural meadows for juvenile and small adult fishes.

Residency

Another index of the utility of transplanted marshes for juvenile salmon is the period of time that they spend in the habitat. There is an obvious relationship here between fish use and submergence, as explained above. Experiments with marked chum fry suggested that fish remained in the transplanted marshes at least as long as in natural habitats (Burger and Nishimura 1995; Chapter 6, present document). Significant residency of chinook fry was shown for a transplanted marsh in the Puyallup estuary, Washington, by Shreffler et al. (1990). Further experiments with chinook fry need to be conducted in transplanted marshes on the Fraser River to compare residency data with those from natural marshes given by Levy and Northcote (1982). The behaviour and residency of chinook smolts in the embayments at sites AN3 and AC2 were investigated in a related study using radiotagged fish (Hvidsten et al. 1995). This study showed that smolts remained in the embayment until a tide change and then moved out into the mainstem of the river to migrate downstream. A similar behaviour pattern was observed in a semi-natural slough.

Prognosis for Transplanted Marshes as Fish Habitats

Bradshaw (1988) pointed out that it is unlikely that restoration projects will recreate habitats that existed before industrial disruption changed their ecological characteristics. This generalization is probably true for sedge marshes in the lower Fraser. There are many biological and geophysical problems associated with developing and maintaining created marshes in this area, as we have shown. We, therefore, would not recommend the destruction of fish habitat subject to replacement with compensatory habitat unless absolutely necessary. Mitigation by avoiding impacts is clearly preferable to avoid risk of failure in a compensatory habitat (Race and Christie 1982), especially as there appears to be a relatively poor monitoring and response system in place to correct problems if they arise. Criteria need to be developed, as has been done in other jurisdictions (e.g., Shreffler et al. 1995) which give targets to determine whether or not particular transplanted marshes are actually functioning as designed.

However because of the major losses of sedge marsh fish habitat that have occurred in heavily industrialized reaches of the estuary such as the North Arm, we advocate continuation of restoration. Other reaches of the estuary where shorelines are devoid of vegetation, such as the mainstem in the vicinity of New Westminster, could also be focused on since fish food invertebrate production is low in that reach (Table 3). More attention should be given to assessing the success of contemporary restoration projects which might be improved by attention to some of the key factors described above (Table 4). These factors, together with other key design criteria, are presented as a check list in Table 5. Monitoring programs on audit reaches could be designed to answer scientific questions, help fill in some of the data gaps identified in the present study, and expand or contract the number of items in the check list.

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Table 1. Percent submergence and standard deviation, averaged over all elevations, of transplanted and natural marshes on the lower Fraser River between May and August 1991 (Table 3.2 in Stronach (1995)).

Site	Mean % Submergence	Standard Deviation
DI1 Transplant	44.1	8.8
DI2 Reference	47.1	22.4
AN1 Transplant	33.2	14.9
AN2 Reference	37.2	13.2
DE1 Transplant	42.4	6.3
DE2 Reference	37.1	11.1
DE3 Reference	52.1	11.7
AC1 Reference	50.7	13.7
AC2 Transplant	60.1	19.1
AC4 Transplant	47.5	12.4
PM1 Transplant [1]	26.4	15.7
PM3 Reference	49.7	11.8
CN1 Transplant	46.8	22.9
CN2 Reference	49.5	13.3

[1] Site was prepared for transplanting of sedges, however, vegetation was established naturally.

Table 2. Comparison of reference:transplant marsh pairs for the mean vegetative cover and for the correlation of vegetative species composition, in the Fraser River estuary, in 1991 and 1992 (Table 4.1 in Stronach (1995))

Reach	Site (reference:transplant)	Mean Ratio [a]	Correlation [b]
Estuary	All Sites	0.87	0.56
North Arm	DI2:DI1	0.92	0.56
	AN2:DI1	0.94	0.76
	AN2:AN1	0.82	0.91
South Arm	DE2:DE1	0.71	0.52
	DE3:DE1	0.78	0.19
	AC1:AC2	0.71	0.11
	AC1:AC4	1.08	0.28
Queens Reach	PM3:PM1 [c]	0.81	0.19
	CN2:CN1	1.31	0.42

[a] ratio of mean cover at reference site to mean cover at transplant site.
[b] product-moment correlation between species composition at reference and transplant site.
[c] this site was prepared for transplanting of sedges, however, vegetation was established naturally.

Table 3. Biomass (g $dw \cdot km^{-1} \cdot d^{-1}$) of adult chironomids, at maximum emergence (estimated 200 $chironomids \cdot m^{-2} \cdot day^{-1}$), contributed by marshes on four reaches of the Fraser River estuary. The biomass of an individual chironomid was estimated as 0.12 mg. Estimates of marsh area are from FREMP (1990) (from Whitehouse et al. (1993)).

Reach	Marsh Area (m^2)	Length (km)	Biomass ($g\ dw \cdot km^{-1} \cdot d^{-1}$)
Port Mann to Patullo Bridge	194,695	17.0	275
Patullo Bridge to Trifurcation	13,205	8.5	37
Trifurcation to south end Annacis Island	30,985	9.5	278
South end Annacis Island to Deas Tunnel	951,750	7.0	3,263

$g\ dw \cdot km^{-1} \cdot d^{-1}$ = grams dry weight per kilometre per day

Table 4. Summary comments on key functions of sedge marshes for fish habitat.

Function	Comments
Plant survival and permanence of vegetative cover	Hydrology and sediment stability are key factors and are predictable in various reaches of the estuary.
Detrital supply	Can vary with dominant plant species and submergence.
Invertebrate use	Insects will likely begin use quickly; submergence will determine use by most crustaceans.
Fish use	Expected even if vegetation not present.
Fish feeding	Submergence and extent of edge habitat will determine availability of food; consumption at offshore habitats likely as well.

Table 5. Some factors to consider in an estuarine sedge marsh transplant project

Substrate	i. grain size of donor vs. transplant site ii. slope of transplant site iii. erosion and stability of transplant site iv. organic content v. longshore drift of sand
Hydrology and Oceanography	i. salinity ii. currents iii. wave energy from vessel traffic iv. fetch v. submergence/elevation with respect to freshet (vertical control) vi. submergence/elevation with respect to tides (vertical control)
Biological	i. target species or plant community ii. potential for invasion from nearby plant communities iii. planting density iv. season v. impact and recovery of donor site
Data management	i. positioning and boundaries (horizontal control) ii. monitoring plans iii. performance criteria iv. reference area/benchmarks v. photos/videos vi. registration of data

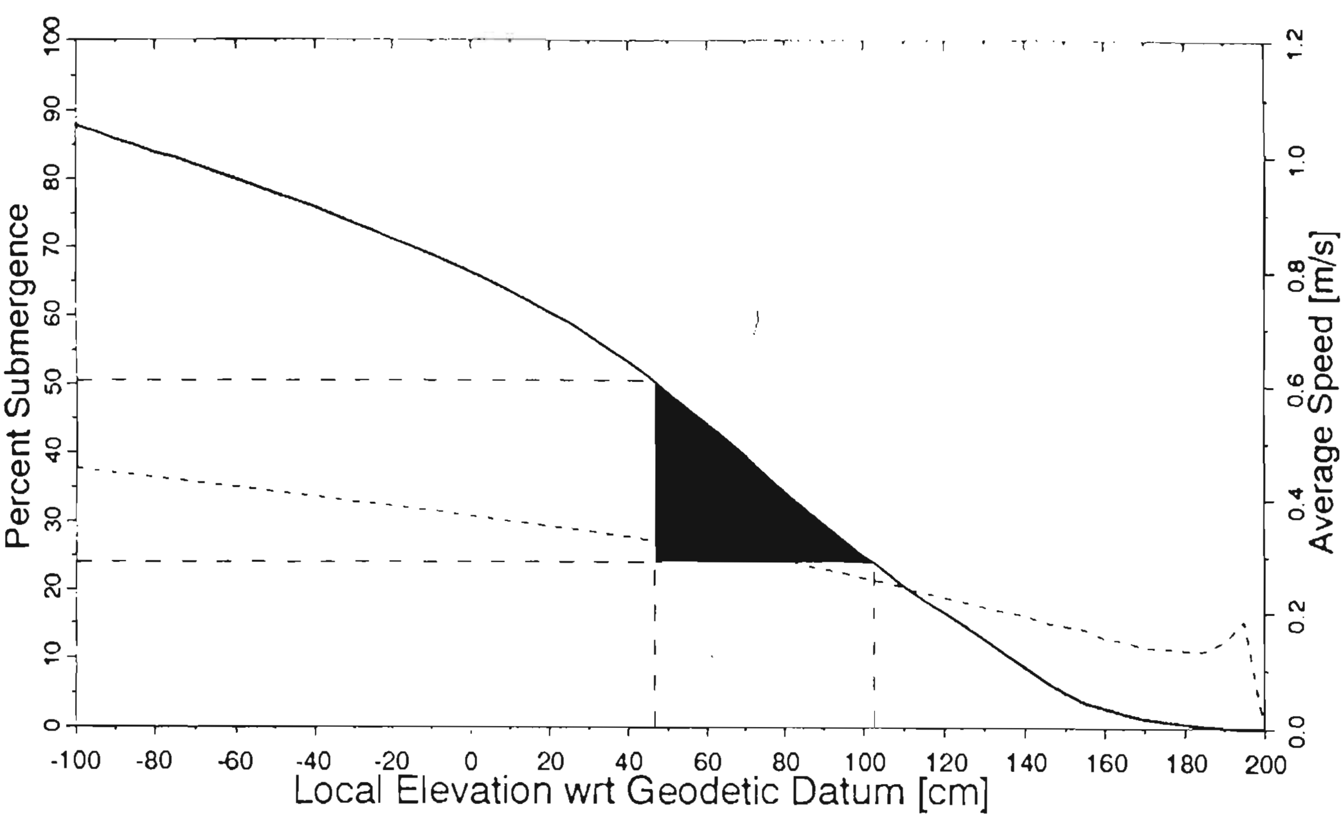

Figure 1. The submergence versus elevation relationship for AN2 reference site in the summer. The fine dashed line represents the relationship between elevation and average velocity, and the solid line represents the submergence/elevation relationship. The shaded area represents the region that had vegetative cover (Figure 3.2 in Stronach (1995)).

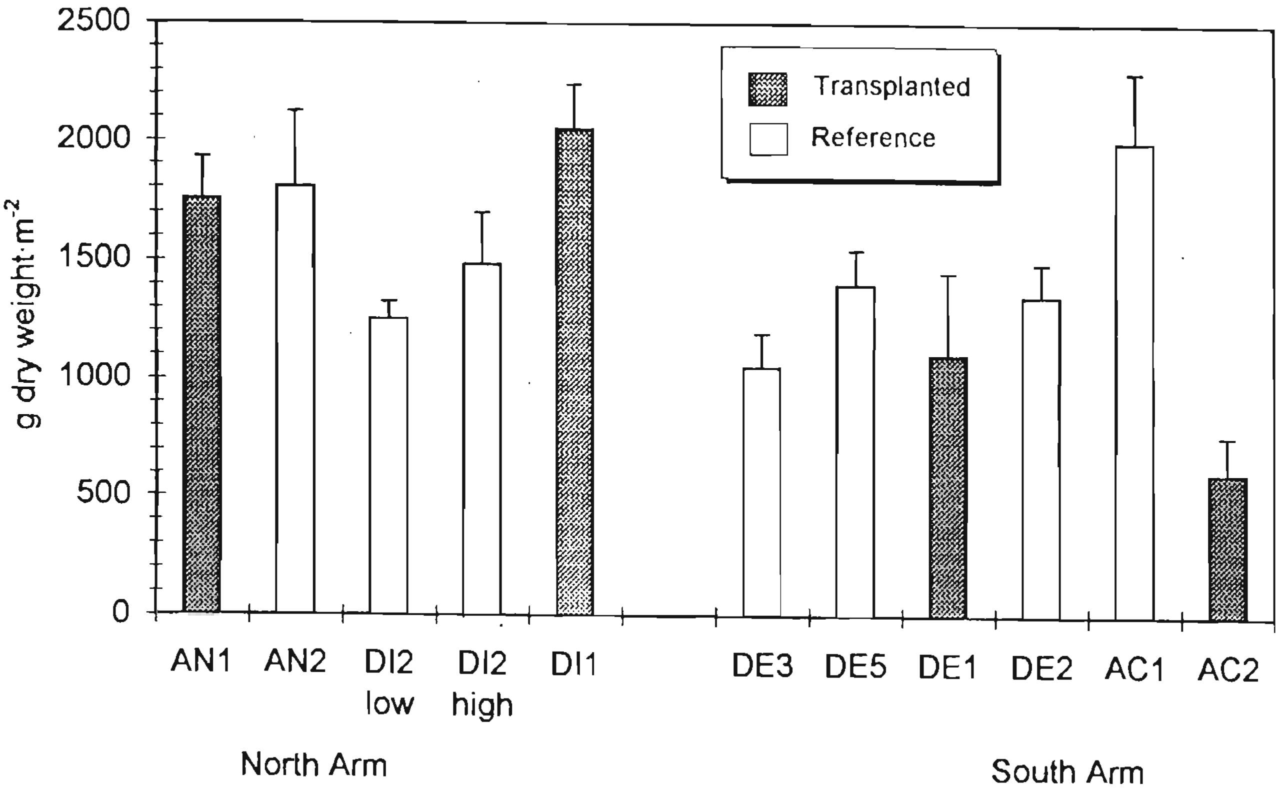

Figure 2. Standing crop biomass (horizontal bar indicates S.D.) of vegetation measured at transplanted and natural marsh sites in the lower Fraser River estuary, August 1991.