EFFECTS OF EFFLUENT FROM A NUCLEAR RESEARCH FACILITY ON BENTHIC MACROINVERTEBRATES

A Thesis

Submitted to the Faculty

of

Graduate Studies

The University of Manitoba

by

William Alexander Wilson Brown

In Partial Fulfillment of the

Requirements for the Degree

of

Master of Science

Department of Entomology

January 1982

EFFECTS OF EFFLUENT FROM A NUCLEAR RESEARCH FACILITY ON BENTHIC MACROINVERTEBRATES

BY

WILLIAM ALEXANDER WILSON BROWN

A thesis submitted to the Faculty of Graduate Studies of the University of Manitoba in partial fulfillment of the requirements of the degree of

MASTER OF SCIENCE

ABSTRACT

Species composition, abundance, and diversity of Winnipeg River benthic macroinvertebrates were studied upstream and downstream of the Whiteshell Nuclear Research Establishment (WNRE) to determine the environmental effects of the facility's liquid effluent. Concentration and distribution of total beta activity in sediment and invertebrates were determined. Radiation dose absorbed by the benthos was estimated.

Reduced abundances of some major taxa, most notably Hexagenia limbata and the chironomids, indicated that some of the benthos immediately downstream (90m) of WNRE were negatively affected by the effluent. However, no reduction in diversity, and the presence and high abundance of some pollution-sensitive taxa, suggested the impact was not severe. Adverse impact to the benthos was attributed to organic coolant, which leaked from the reactor's cooling circuit and deposited on the bottom sediment.

Total beta activities in benthos and sediment downstream of WNRE ranged from 1 to 116 Bq/g dry wt., with the highest concentrations occurring close to the outfall. No bioaccumulation of beta-emitting nuclides (mostly Cesium-137) occurred. It was proposed that the ratio 'unit radioactivity of organism/unit radioactivity of sediment' is a more realistic assessment of nuclide accumulation in benthic organisms than the commonly used expression 'unit radioactivity organism/unit radioactivity surrounding water'. Radiation dose rates received by the benthos (0.0006-0.069 rads/day) were three to five orders of magnitude lower than the rate at which biological effects due to radiation have been observed.

Environmental effects of WNRE liquid effluent were isolated to the area immediately downstream of the outfall. It was recommended that the study of benthic macroinvertebrates be repeated within the next five to ten years, as there have been no measurable organic coolant leaks since 1979 and some changes may be expected in the downstream zone.

ACKNOWLEDGEMENTS

I wish to express my sincere thanks and appreciation to my supervisor Dr. J.E. Guthrie for his understanding and guidance throughout the study. I wish also to express my gratitude to Dr. A.G. Robinson for his guidance throughout the study and to Dr. D.M. Rosenberg for his advice and valuable critical review of the thesis manuscript.

Grateful acknowledgement is made to the Atomic Energy of Canada Limited for making this study possible by providing equipment, facilities, and funds. The assistance given by Mr. O. Acres, Mr. T. Wiewel, and other staff members of Environmental Surveillance of the Whiteshell Nuclear Research Establishment is appreciated. I wish to thank Mrs. R. Andres, Ms. C. Andres, Ms. C. Brown, and Ms. M. Hodgson for able technical assistance. Thanks are also extended to all the staff and graduate students of the Department of Entomology for their assistance. I would like to acknowledge the advice and help on statistical analysis given by Dr. N.J. Holliday. Dr. T.D. Galloway provided valuable advice throughout the study. Thanks to Mr. B. Bilyj, Mr. J.F. Flannagan, Mr. A. Wiens, and other staff members of the Freshwater Institute for providing assistance during the identification of the invertebrates. My thanks to Dr. G.R.B. Webster for his critical review of the thesis manuscript and to Mrs. M. Funk for typing the manuscript. I wish to express special thanks to my mother Mrs. Mary Brown and to my wife Debby for their moral support.

TABLE OF CONTENTS

TABLE OF CONTENTS

TABLE OF CONTENTS

LIST OF APPENDICES

LIST OF TABLES

LIST OF TABLES

LIST OF FIGURES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

The Whiteshell Nuclear Research Establishment (WNRE) of Atomic Energy of Canada Limited is situated on the east bank of the Winnipeg River, about 8 km south of Lac du Bonnet, Manitoba (Fig. 1). The establishment was opened in 1963. It comprises laboratories for handling radionuclides, a 60 megawatt (thermal) research reactor (WR-1), hot cells for examining intensely radioactive materials, and other facilities. WNRE's operating licences, issued by the Atomic Energy Control Board, authorize the establishment to release up to specified quantities of radionuclides to the environment. WR-1 was started up in November, 1965. Since that time, the Winnipeg River has continuously received varying amounts of low-level radioactive liquid effluent released from WNRE, in addition to the normally non-radioactive cooling water discharged from the heat exchanger circuit of the WR-1 reactor. The organic coolant from the reactor's primary cooling circuit leaked into the river between January and May, 1978 (Guthrie and Acres, 1979). Some of this material was deposited on the riverbed while the remainder was transported downstream.

In order to determine the environmental effects of the liquid effluent released from WNRE, the benthic macroinvertebrates upstream and downstream of the establishment were studied in 1966-67 (Ireland, 1968) and again in 1978-79. The results of the 1978-79 study are presented in this thesis. The 1966-67 study served as background information for the present study.

The objectives of the present study were:

(1) to determine the changes that have occurred in the benthic macroinvertebrate community of the Winnipeg River attributable to WNRE liquid effluent;

(2) to determine the concentration and distribution of total beta activity in river-bed sediment and benthic macroinvertebrates near the WNRE effluent outfall, and;

(3) to estimate the radiation dose absorbed by the benthic macroinvertebrates in the vicinity of the WNRE effluent outfall.

CHAPTER II

LITERATURE REVIEW

1. Pollution and pollution assessment

An aquatic pollutant is any substance causing or inducing objectionable conditions in any watercourse which adversely affects the use of its water (Federal Department National Health and Welfare, 1978). Because of the increasing demand for the use of surface waters, the need for assessment of pollution has increased. Most often, water quality in rivers has been determined by chemical analysis. However, chemical tests indicate river conditions only at the time of sampling, and indicate nothing about the biological consequences of a pollutant (Wilhm, 1972). In recent years, many workers have used river biota to assess water quality. Biological methods are based on the assumption that pollutants produce distinct and measurable effects on structural and functional parameters of organisms or groups of organisms (Wilhm, 1972). Thus, organisms are used as ecological indicators.

2. The use of benthic macroinvertebrates in pollution studies

Benthic macroinvertebrates have frequently been used in pollution studies. These animals live in or on the substrate of a body of water (EPA, 1973). The major benthic taxa in freshwaters are insects, annelids, molluscs, crustaceans, and nematodes. They occupy intermediate levels of the aquatic food chain, between the producers and the top carnivores. They may be divided into the following functional groups: shredders, collectors, scrapers (grazers) and predators (Brinkhurst and Cook, 1974; Fuller, 1974; Merritt and Cummins, 1978). Shredders process living or dead vascular plant tissue. Collectors process suspended or deposited particulate organic matter. Scrapers ingest periphyton, and predators generally feed on other macroinvertebrates. Many species of benthic macroinvertebrates are a major food source for larger carnivores, particularly fish.

Benthic macroinvertebrates are suited for pollution studies because their specific habitat preferences and limited mobility cause them to be continuously affected by substances entering their environment (Cook, 1976). Benthic species vary considerably in their sensitivity to various pollutants (Roback, 1974). A pollutant released into a river may eliminate or reduce the densities of species sensitive to the stress, until only the organisms that can survive the adverse conditions predominate. Thus, benthic macroinvertebrate communities in stressed habitats usually are characterized by a reduction in total number of species, and are often numerically dominated by individuals of a few tolerant species (Cairns and Dickson, 1972).

(A) Methods used for the assessment of water quality using macroinvertebrates

Various methods have been used to assess environmental conditions, including indicator organisms (Forsyth and Fox, 1976; Scullion and Edwards, 1980), indices of community structure based on richness and diversity (Cairns and Dickson, 1971; Dills and Rogers, 1974; Howell and Gentry, 1975; Ruggiero and Merchant, 1979), and invertebrate density (Cole and Kelly, 1978; Langford and Aston, 1972; Lenat, 1979). However, relying entirely on only one of these methods can lead to a misunderstanding of prevailing environmental conditions (Harman, 1974). Two of these methods have been used for the present study, as follows:

a. Invertebrate density

The density of benthic macroinvertebrates is usually expressed in number of organisms per m^2. In pollution studies, both density of the total number of invertebrates (standing crop) and the densities of individual taxa or taxonomic groups, have been used to determine environmental conditions. Pollution-sensitive invertebrates exposed to a pollutant decline in density, while the densities of tolerant forms do not change or may increase (Armitage, 1980; Lenat, 1979). Fremling (1970) demonstrated that the densities of the pollution-sensitive mayfly nymph *Hexagenia limbata* could be used to indicate varying degrees of organic pollution in the Mississippi River. No nymphs were found in grossly polluted areas of the river; intermediate densities ($400/m^2$) were found in moderately polluted regions; and larger densities ($800/m^2$) in clean water areas.

b. Indices of community structure

Richness refers to the total number of taxa in a defined area. The more taxa that are present, the greater the richness. Various studies indicate pollutants released into a river reduce the richness of benthic macroinvertebrate populations (Dills and Rogers, 1974; Gaufin and Tarzwell, 1956; Wilhm, 1970). For example, an area of the Savannah River receiving thermal effluents supported 10 benthic macroinvertebrate species while an undisturbed control area supported 29 species (Howell and Gentry, 1975).

The measurement of diversity takes into account how equally individuals are distributed among the taxa and the number of taxa involved

(Pielou, 1969). The more equal the distribution (evenness) and the greater the number of taxa, the greater the diversity. A number of equations have been proposed to measure diversity (e.g., Good, 1953; Hill, 1973; May, 1975; Pielou, 1969). The one most widely used is the Shannon-Weaver diversity index:

$$d = - \Sigma \, {}^{n_i}/N \log {}^{n_i}/N,$$

where d is the diversity index, n_i is the number of individuals in the ith taxon, and N is the total number of individuals (Pielou, 1966, 1969). It has been used often for pollution studies (e.g., Cole and Kelly, 1978; Dills and Rogers, 1974; Emery and McShane, 1980; Hughes, 1978; Johnson and Brinkhurst, 1971; Kaeslar and Herricks, 1979; Ruggiero and Merchant, 1978) because it is simple, it combines evenness and richness, and it is relatively independent of sample size (Wilhm, 1970, 1972). In this thesis, diversity will refer to abundance of invertebrate taxa as summarized by the Shannon-Weaver equation.

Many workers have shown an inverse correlation between diversity index values for benthic macroinvertebrate communities and environmental stress (Cairns and Dickson, 1971; Dills and Rogers, 1974; Howell and Gentry, 1975; Wilhm, 1970, 1972; Wilhm and Dorris, 1966, 1968). The mean diversity of benthic macroinvertebrates in Skeleton Creek for example was 0.8 in those areas receiving large amounts of organic wastes; 1.59 in areas receiving smaller amounts of waste; and 3.4 in the areas not receiving wastes (Wilhm, 1970).

Measurements of richness and diversity have been criticized because their application results in a loss of information, data being reduced to a single number (MacArthur, 1972; May, 1975). Since richness and diversity indices provide only a measure of structure, and exclude qualitative data, environmental changes which might result in species replacement are not detected (Kaesler and Herricks, 1979). Another problem is that more factors affect species richness and diversity in an environment than the stress imposed by pollutants. Low values of richness and diversity do not necessarily indicate polluted conditions. For example, interspecific competition in a homogeneous environment often appears to reduce richness and diversity (Harman, 1972). Also, factors such as sediment type (Cole and Kelly, 1978; de March, 1976; Ruggiero and Merchant, 1979); sampling method (Hughes, 1978); time of year samples were taken (Wilhm and Dorris, 1966); and taxonomic level used to calculate the index (Hughes, 1978; Kaeslar and Herricks, 1979) may also affect richness and diversity values. These factors must be standarized before comparisons between richness values, and between diversity index values, can be made with any justification (Hughes, 1978).

3. Potential effects of WNRE liquid effluents on Winnipeg River benthic macroinvertebrates

(A) Heated water

Benthic macroinvertebrate communities in waters receiving heated-water effluents are often characterized by a reduction in standing crop, richness, diversity and biomass (Cairns, 1967; Coutant, 1962, 1977; Howell and Gentry, 1975; Shiomoto and Oslo, 1978). Water 10^{o}C above ambient discharged into Lake Keowee from the Oconee Nuclear Station eliminated Ephemeroptera, Odonata, Trichoptera and Hemiptera (Ferguson and Fox, 1978). The only invertebrates which survived the stress were from several species of Chironomidae and Ceratopogonidae. Heated effluent can also increase initial growth rates and spur precocious maturation of some species, resulting in decreased adult size and abbreviated life span (Parker and Krenkell, 1969). Other sub-lethal functional responses such as effects on metabolism, behavior and reproduction have also been observed (Clark, 1969; Parker and Krenkell, 1969).

(B) Organic coolant

Organic coolant is, initially, a mixture of partially hydrogenated ortho-, meta-, and para-terphenyls, similar in appearance to cooking oil. Exposure to the heat and ionizing radiation in the core of the reactor causes pyrolytic and radiolytic reactions in the coolant, producing a complex mixture of compounds with a wide range of molecular weights (Guthrie and Acres, 1979). This irradiated coolant is a viscous black liquid.

To date, no workers have investigated the effects of organic coolant on benthic macroinvertebrates. Fish appear to be sensitive to compounds similar to organic coolant, such as various terphenyl compounds (Guthrie and Acres, 1968) and similar aromatic compounds (Holland *et al.*, 1960; Truelle, 1958). Benthic invertebrate communities in waters receiving other types of organic material, such as oil refinery wastes and sewage, are often characterized by a reduction in richness and diversity (Damback and Olive, 1969; Fremling, 1970; Johnston and Brinkhurst, 1971; Loch and Gregory, 1973; Wilhm and Dorris, 1966).

(C) Radiation

Pertinent aspects to the study of effects of radiation on Winnipeg River macroinvertebrate populations include: (1) the types of radionuclides released to the river from WNRE, (2) the behavior of ecologically important radionuclides in the aquatic environment and its biota, and (3) the radiation dose absorbed by the biota.

a. Radiological background

When the ratio of numbers of neutrons to protons in the nucleus of a given nuclide lies outside the stability range for that mass number, the nuclide will be radioactive. The unstable nucleus, called a radionuclide, will become more stable usually by emitting alpha or beta particles and gamma photons. This stabilization process is called radioactive decay. The rate of decay is a characteristic property of the nuclide. Since alpha and beta particles and gamma photons may remove the orbital electrons from atoms, they are called ionizing radiations. The damage to tissue caused by the passage of ionizing radiation is proportional to the number of ions and the concentration of free radicles produced per unit mass of tissue. The amount of radiation received by a tissue or by an organism is called the absorbed dose.

b. Radionuclides of ecological concern released to the Winnipeg River from WNRE

The major radionuclides released to the Winnipeg River from WNRE are Cesium-137 (Cs-137) and Strontium-90 (Sr-90). Cs-137 emits gamma and beta radiation. Sr-90 emits beta radiation only. Neither radionuclide is chemically toxic. Cs-137 and Sr-90 are ecologically important because of their longevity (physical half-lives are 28 and 30 yr respectively) and tendency to be assimilated by aquatic organisms (Reichle et al., 1970; Rice and Baptist, 1974).

Cs-137 is the largest contributor to potential radiation dose received by benthic macroinvertebrates in the Winnipeg River. Suspended and deposited sediments in the Winnipeg River, because of their high clay content, readily sorb Cs-137 by ion exchange (Guthrie, 1964). Suspended particulate material with sorbed Cs-137 eventually will settle in quiet-water areas. Thus, benthic macroinvertebrates in slow-water areas in the vicinity of WNRE's effluent outfall may be exposed to local concentrations of the nuclide. Conversely, Sr-90 does not readily sorb to suspended material in the Winnipeg River (Guthrie, 1964) so is transported downstream in the water and high concentrations are unavailable to the benthic macroinvertebrates in the vicinity of WNRE.

c. Behavior of Cs-137 in aquatic organisms and aquatic food chains

Aquatic organisms accumulate Cs-137 in three ways: (1) surface sorption, (2) intake via the gills and/or mouth, and (3) assimilation from ingested food (Rice and Baptist, 1974). Small organisms with large surface area to volume ratios (e.g., zooplankton) probably acquire most of their Cs-137 via surface sorption (Cushing, 1970; King, 1964; Reichle *et al.*, 1970). Larger organisms, such as macroinvertebrates and fish, obtain most of their Cs-137 from food (Reichle *et al.*, 1970).

The term "biological half-life" is defined by Cushing (1970) as the time it takes the initial radionuclide burden within an organism to decrease by 50%. Species, age, size, and physical activity all affect the biological half-life of Cs-137 in aquatic organisms but body size, or some particular metabolic parameter correlated with body size, is the most important of these factors (Reichle, 1970). The biological half-life of Cs-137 in *Chironomus plumosus* (Gerking *et al.*, 1976), *Lethocerus americanus* (Guthrie and Brust, 1969), and a variety of fish (Hasanen *et al.*, 1967; Hewett and Geffenson, 1978; Williams and Pickering, 1961), was 4, 4.5, 10 and 40-200 days respectively.

Retention and elimination of Cs-137 in aquatic organisms is also affected by physical and chemical conditions of the aquatic environment. Accumulation of Cs-137 by a variety of aquatic vertebrates and invertebrates is inversely related to the potassium concentration in the water (Gustafson, 1962; Guthrie and Burzynski, 1972; Kolehamainen *et al.*, 1967). Organisms in oligotrophic (nutrient-poor) lakes generally have higher Cs-137 concentrations than organisms in eutrophic (nutrient-rich) lakes (Carlson and Liden, 1978; Kolehamainen *et al.*, 1967; Reichle *et al.*, 1970). Increased water temperature increases the rate of uptake and accumulation of Cs-137 by chironomid larvae (Gerking *et al.*, 1976) and by mosquito larvae (Guthrie and Burzynski, 1972).

Bioaccumulation is pertinent to the study of radiation effects on aquatic orgamisms, because it will result in a higher radiation exposure for the organisms involved. Although it is known that aquatic food chains play a role in the transfer of Cs-137, the fate of this radionuclide at each successive level in the food chain is largely unknown. The highest concentrations of Cs-137 in aquatic ecosystems are usually found in algae and sediment (Guthrie, 1970; Shure and Gottschalk, 1976; Wrenn *et al.*, 1971). Guthrie (1970) found that the specific activity of Cs-137 in aquatic insects decreased from the primary consumer to the predator level. Mean Cs-137 activity in the primary consumers (e.g., ephemeropteran nymphs, chironomid larvae) was one order of magnitude higher than the acitivity in predators such as dytiscid larvae and dragonfly nymphs (Guthrie, 1970). Wrenn *et al.* (1970) found the Cs-137 concentration in the fish species of the Hudson River did not increase with trophic level. However, other studies indicate that Cs-137 increases in concentration between trophic levels (Aoyama, 1978; Carlson and Liden, 1978; Gustafson, 1967).

d. Absorbed radiation dose

The total radiation dose absorbed by benthos consists of an external and an internal, component. The external component is the result of exposure to radioactivity in the surrounding water and sediment. The internal component is received from ingested radionuclides (Blaylock, 1972). The radioactive effluent discharged from a nuclear facility may contribute to both components of the total dose. In most cases the internal dose accounts for the largest fraction of the total absorbed radiation dose (IAEA, 1976).

One of the problems in evaluating radiation effects in natural situations is estimating the actual radiation dose received by the animal (Ophel, 1979; Turner, 1975). Dose rates have been calculated from measured concentrations of radionuclides in the biota, water, and sediment (Blaylock, 1972; IAEA, 1976). Accurate estimates of radiation dose using this method are difficult to make because: several sources of radiation must be considered (IAEA, 1976; Turner, 1975); the geometry of organisms can only be approximated (IAEA, 1976); and the concentrations of radionuclides in an organism fluctuate during its life time.

e. Chronic low-level radiation and benthic macroinvertebrates

Most studies of the effects of radiation on aquatic biota have investigated the effects of acute (high dosage for a short period of time) exposure (IAEA, 1976). Few studies have investigated the effects on natural benthic populations of chronic exposure to low dose rates of radiation (Blaylock, 1972). However, the studies to date indicate that aquatic biota is unaffected by the dose typically occurring in the vicinity of nuclear power stations (Blaylock, 1972; IAEA, 1976). Radionuclides discharged into disposal ponds from the Hanford laboratories and reactors for over 20 yr had not adversely affected the ponds' benthic invertebrate populations (Emery and McShane, 1980). The ponds were grouped into 10, 10^2 - 10^3, and 10^5 - 10^6 millirads per week categories, according to dose rates received at the water/sediment interface. Invertebrates commonly found in small uncontaminated ponds used as controls also were found in the Hanford ponds. Similar benthic invertebrate populations existed in the higher and lower dose rate ponds.

Most studies on benthic populations exposed to chronic radiation have been conducted on populations of snails and insects inhabiting White Oak Lake, Tennessee, a former settling basin for low-level radioactive liquid effluents discharged from the Oak Ridge National Laboratory. Organisms living in the bottom sediments of the lake received about 240 rads/yr. An increased frequency of chromosomal aberrations occurred in benthic chironomid larvae, Chironomus tentans (Blaylock, 1966) but these genetic aberrations were rapidly eliminated by natural selection or genetic drift and had no effect on the population. The population structure of the snail Physa heterostropha living in White Oak Lake was similar to that from a control lake (Cooley and Nelson, 1970). Frequency of egg capsule production by the exposed snail population was reduced but there was no reduction in total egg production because each capsule contained an increased number of eggs.

CHAPTER III

DESCRIPTION OF STUDY AREA

1. The Winnipeg River

The Winnipeg River extends approximately 320 km from Kenora on the Lake of the Woods, to Traverse Bay on Lake Winnipeg. The river flows through an area underlain by granite and granitic gneisses of the Precambrian Shield (Davies _et al._, 1962). This area is the transitional zone between the coniferous forests of the Canadian Shield and the aspen parklands of the prairies (Bird, 1930).

The Winnipeg River is classified as a medium sized lowland river (Elton, 1966). It has a drainage basin of 125,500 km^2, which includes a large portion of northwestern Ontario and eastern Manitoba. The river's mean discharge in the vicinity of WNRE is 730 m^3/s (calculated from measurements taken by Manitoba Hydro at the Seven Sisters Generating Station between 1963 and 1978). Discharge varied between 320 and 2200 m^3/s during this period. A series of hydro-electric dams have been built along the river because of this large dependable discharge. Consequently, a number of lakes exist along its course to Lake Winnipeg.

The Winnipeg River flows north past WNRE at a rate of 0.1 to 0.5 m/s (Merritt, 1965). The stretch of river in the vicinity of WNRE, between Seven Sisters Hydroelectric Generating Station and Lac du Bonnet (Fig. 1), has a mean depth of 4.8 m, a mean width of 390 m, and a fall of 0.3 m in 4 km. Banks in this area are moderately steep and eroded. The regional overburden is clay. Cover along this stretch is dominated by thick stands of birch, poplar, and pine.

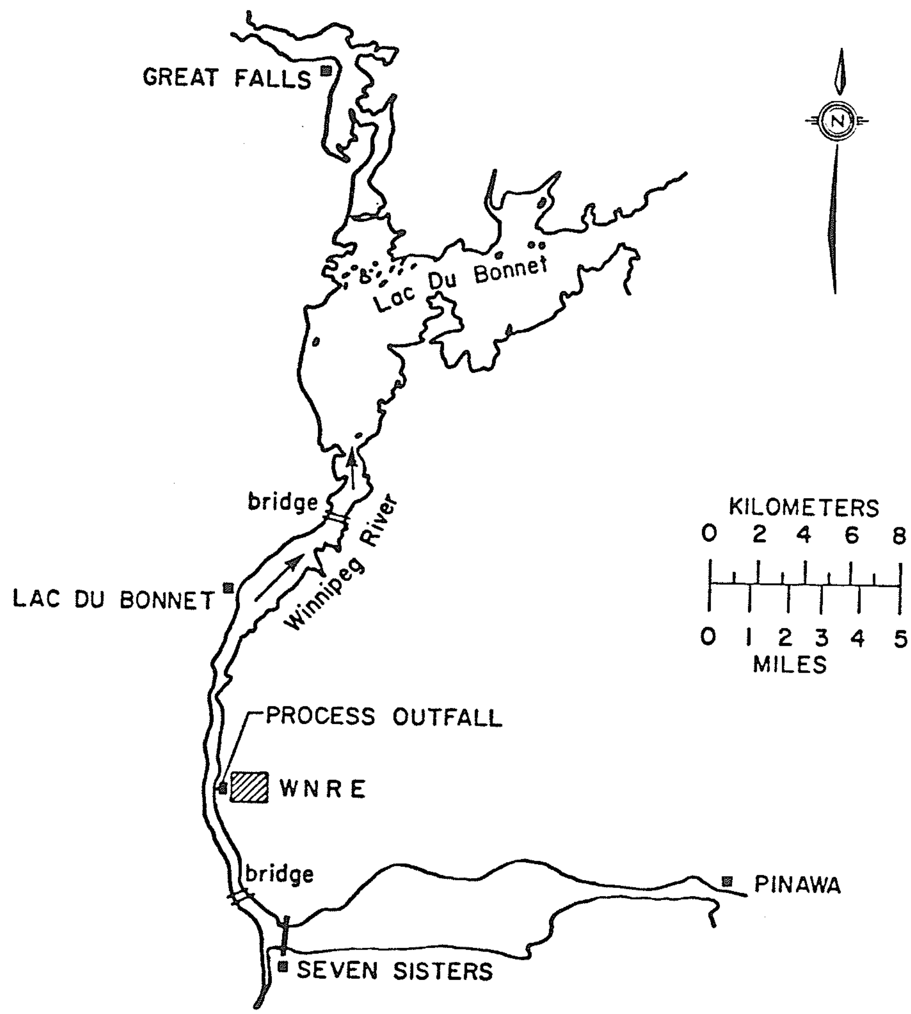


Figure 1. Location of Whiteshell Nuclear Research Establishment (WNRE).

No industrial effluents are released into the Winnipeg River between the Manitoba-Ontario boundary and WNRE. However, domestic effluents from a number of small towns and summer cottages enter this section of the river.

2. The study area

Merritt (1965), using a dye injected into the WNRE effluent outfall, showed that liquid effluent forms a plume along the east bank of the river (Fig. 2). Therefore, this study concentrated on the east side of the river, using a control area upstream and a treatment area downstream of the WNRE effluent outfall. The treatment (T-) zone (Fig. 3) began 90 m downstream of the effluent outfall and extended a further 1 km. The control (C-) zone (Fig. 4) began 2.9 km upstream of the outfall and extended a further 0.5 km. Both zones were somewhat sheltered from the main channel of the river by points of land. Current speeds were low in both zones. Fine grained sediments predominated. Rooted aquatic plants were abundant.

Sampling was restricted to stations, used by Ireland (1968), having fine particulate substrates (clay, silt or very fine sand). The sampling transects established by Ireland (Appendix A(1) and A(2)) in T- and C- zones were re-established in 1978. Of these, five transects were selected in each of the T- (Fig. 3) and C- zones (Fig. 4). The 1.0 m and 2.0 m depth contours were sampled on each transect. A typical transect marked by red and white stakes is shown in Fig. 5.

Figure 2. The Winnipeg River in the vicinity of the Whiteshell Nuclear Research Establishment (WNRE). The control (top) and treatment (bottom) sampling zones are shown. WNRE appears just above the square which outlines the treatment zone.

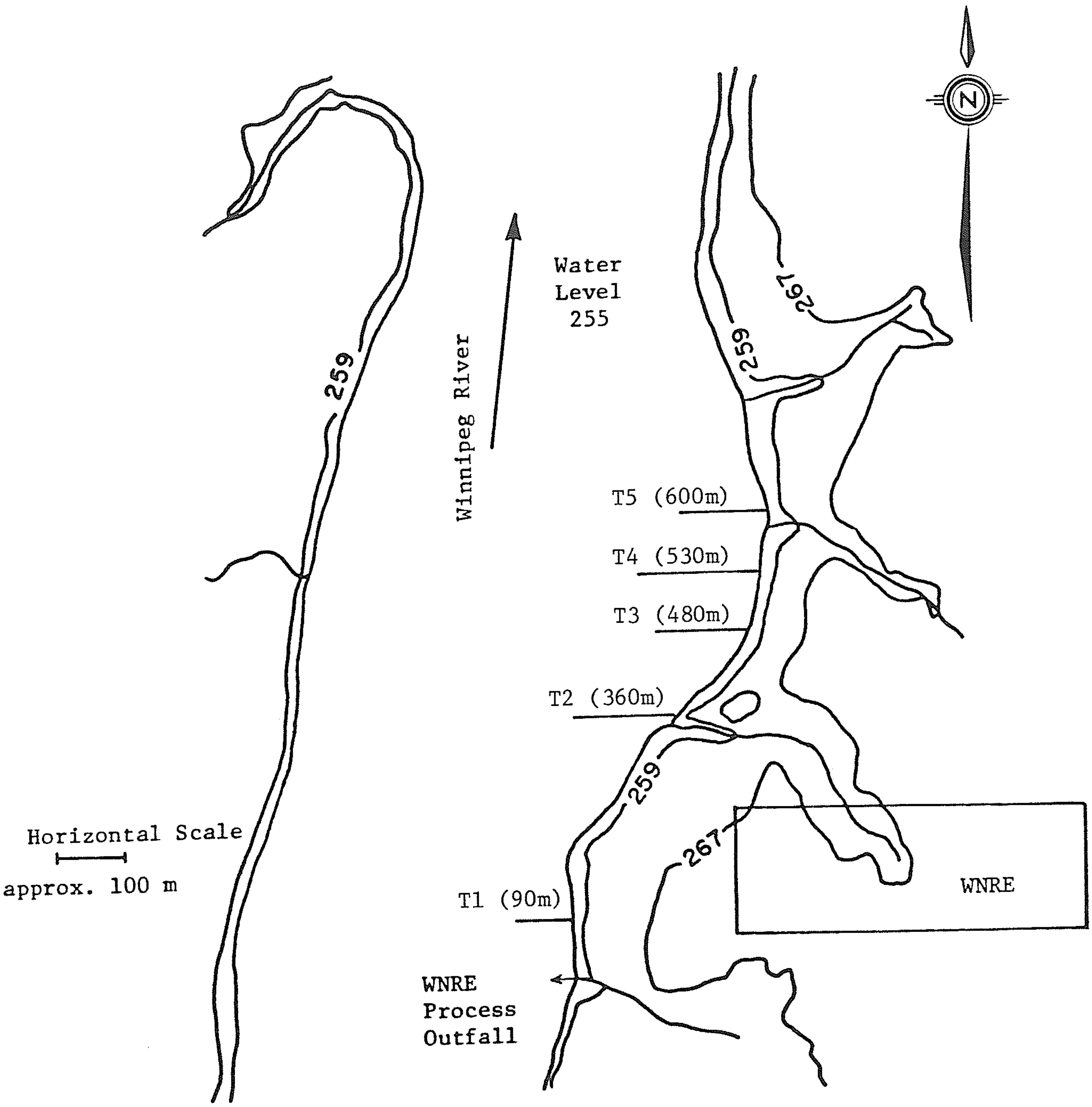


Figure 3. Location of treatment (T)-zone sampling transects, downstream of Whiteshell Nuclear Research Establishment (WNRE). Sampling stations are located at 1m and 2m depths on each transect and are designated: T1-1m, T1-2m, T2-1m, T2-2m, T3-1m, T3-2m, T4-1m T4-2m, T5-1m, T5-2m. Distance of each transect from the WNRE effluent outfall is given in parentheses.

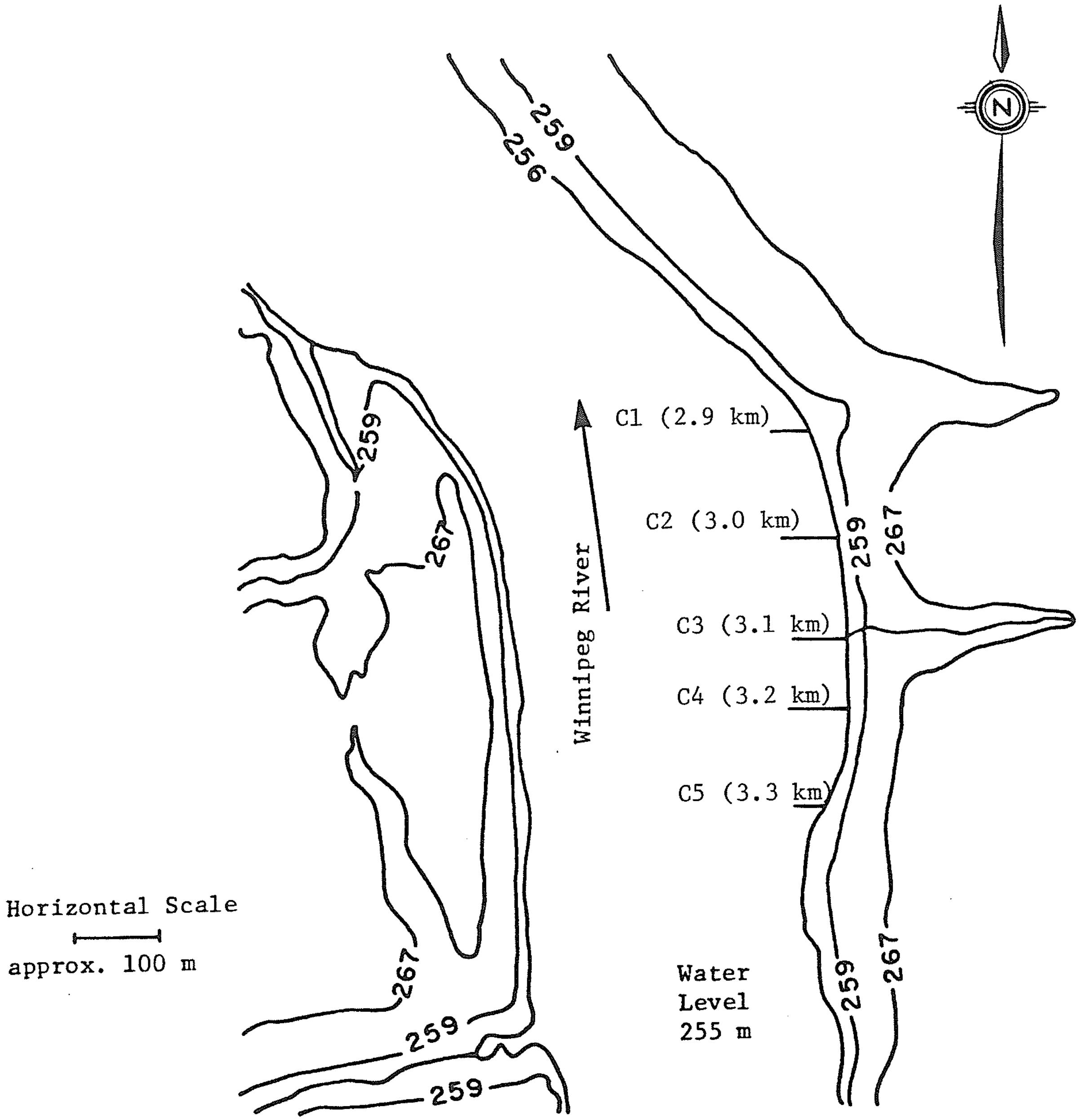


Figure 4. Location of control (C)-zone sampling transects, upstream of the Whiteshell Nuclear Research Establishment (WNRE). Sampling stations are located at 1m and 2m depths on each transect and are designated: C1-1m, C1-2m, C2-1m, C2-2m, C3-1m, C3-2m, C4-1m, C4-2m, C5-1m, and C5-2m. Distance of each transect from the WNRE effluent outfall is given in parentheses.

Figure 5. Treatment zone sampling transect T3.

CHAPTER IV

MATERIALS AND METHODS

1. Physical and chemical measurements

Physical and chemical measurements were taken to describe the general habitat in which the benthos live; to assess whether T-zone and C-zone habitats were sufficiently similar to warrant comparisons of their benthic faunas; and to assess whether effluents released from WNRE had affected habitat of the benthos.

Temperature and dissolved oxygen (DO) of the river water were measured with a Yellow Springs Instrument (YSI) Company Model 54 ARC DO meter equipped with a YSI 5739 DO probe. These measurements were made at weekly intervals between 12:00 P.M. and 4:00 P.M. from June through September of 1978 and 1979. They were taken ∿0.2 m above the river-bed at locations T1-2 m and C1-2 m. In addition, measurements of temperature and DO were taken at all stations in T- and C-zones on 25 July 1978 and 30 August 1979.

Mean weekly discharges of the river were calculated from mean daily discharge records compiled by Manitoba Hydro at the Seven Sisters Generating Station, 6 km upstream from WNRE.

Velocity of the river water was measured with a Teledyne Gurley 622-E current meter at each sampling station ∿0.2 m above the bottom sediment on 24 August 1979. Another measurement was taken 3 m below the water surface at the middle of the river, midway between the east and west bank.

Secchi disc readings (Welch, 1948) were taken weekly to measure relative transparency of the Winnipeg River water. Measurements were taken in the middle of the T-zone and also in a region located 5 km upstream of the Seven Sisters Generating Station.

Samples of macrophytes collected in the vicinity of each sampling station, were pressed, dried, and identified to species using Scoggan (1957) and by comparing samples to the WNRE botanical reference collection. Relative plant densities were noted at the time of sampling.

River-bed sediments were sampled using a modified version of the Ireland sampler (Ireland, 1968). The sampler was constructed of stainless steel and teflon and consisted of a 20x15x3 cm steel pan with a door that slides through teflon grooves (Fig. 6). The door was closed by a rope which passed through a guide fitted to a stainless steel pipe handle attached to the pan. To operate, the sampler with the door open, was pushed firmly into the bottom sediment and the sliding door was closed by pulling the nylon rope. The sampler was then retrieved from the bottom and the contents examined.

Each sediment sample was qualitatively classified according to texture and appearance and the presence or absence of organic coolant was noted. A small (1 cm^3) sub-sample was taken from each sediment sample for total beta counting.

2. Benthos sampling

(A) Sampling procedures and schedule

Samples from the Ireland sampler were washed through 500 μm mesh Nitex cloth (Fig. 6). Organisms and debris retained by the net were placed in jars half-filled with water, and left undisturbed for at

Figure 6. The Ireland sampler.

least 6 h to give the organisms time to eliminate their gut contents. The sample (organisms and debris) was then preserved in 10% buffered formalin, from which the organisms were subsequently removed and counted, using a 2x dissecting microscope, and preserved in 70% ETOH.

Single macroinvertebrate samples were collected once a month from each of the 20 sampling stations, from June to September 1978 and 1979 (Table 1). The time separating the first and last sample of a period was usually <1½ days. The same substrate surface area was not sampled more than once during a season.

(B) Identification of organisms

The following reference keys were used to identify the invertebrates: Ephemeroptera: Burks (1953); Odonata: Walker (1958); Plecoptera: Frison (1935); Megaloptera: Usinger (1963); Trichoptera: Wiggins (1977); Coleoptera: Brown (1972) and Usinger (1963); Diptera: Mason (1973); Oliver et al. (1978), Saether (1971, 1977), Stewart and Loch (1973), and Usinger (1963); general: Merritt and Cummins (1978); Amphipoda: Bousfield (1958); and other invertebrates: Pennak (1953). Invertebrate identifications were verified by Mr. J. Flannagan (Ephemeroptera); Dr. T. Galloway (Trichoptera); Dr. W. Hilsenhoff (Coleoptera); and Mr. B. Bilyj (Chironomidae).

Chironomid larvae were reared to the adult stage to provide life cycle stages for identification. Adults and immatures were slide mounted according to Beirne (1963). A reference collection of reared and dissected Chironomidae was prepared for the WNRE area. Identifications of larvae collected in the 1978 sampling season were verified using the reference collection. Chironomid larvae collected in 1979

Table 1. Dates and duration of benthos sampling.

Sampling period	Date and duration of sampling period
1978	
#1	June 2-9
#2	July 20-21
#3	August 16-17
#4	September 10-11
1979	
#1	June 1-2
#2	July 4-5
#3	August 14-15
#4	September 11-12

were used for total beta counting and, hence, were not identified past the family level.

3. Radioassay of sediment and macroinvertebrates

Sediment and invertebrate samples were radioassayed by counting total beta emissions (Guthrie and Grummitt, 1963). Wet sediment samples were dried overnight in a hot air oven at 110°C. The dry samples were ground to a fine powder, and coarse grit and detritus were removed. The ground samples were evenly spread on tared 3 cm diameter stainless steel planchets, moistened with a 1:10 colloidon-acetone mixture, dried under heat lamps, and weighed. Total beta emissions were counted in a Nuclear Chicago end window gasflow beta detector, equipped with an automatic sample changer. The average background of the counter was 2.5 counts per minute. The K-40 equivalent procedure used to calibrate the counter is described in Guthrie and Grummitt (1963). Total beta activity was expressed as Becquerels (Bq)/gram (g) dry weight (K-40 equivalent).

Individual invertebrates of the same taxon were pooled to obtain sufficient biomass to allow total beta counting. They were washed with distilled water, weighed, dried overnight at 110°C, then weighed again. They were then dissolved in a minimum amount of concentrated nitric acid, spread out on stainless steel planchets, dried under heat lamps, and total beta activity was counted.

Two of the most radioactive *Hexagenia limbata* samples were examined by gamma spectroscopy to determine which gamma emitting radionuclides were present.

Maximum radiation dose rates received by the benthic macroinvertebrates were estimated for each sampling transect. The equation used to calculate total dose received by benthic invertebrates from all beta-emitting radionuclides is as follows (Blaylock, 1972; Killough and McKay, 1976):

$$Di = (1.3824 \times 10^{-3})\ S_i\ C_i^b\ E_i$$

where,

(1.3824×10^{-3}) = constant to convert Bq/g of biota to dose rate in rads/day,

S_i = the concentration of beta emitting radionuclides in sediment expressed in Bq/g,

C_i^b = bioaccumulation factor (conc. in biota)/(conc. in sediment) for biota,

E_i = the effective absorbed energy of the beta emitting radionuclides in the biota.

This equation implies (1) the diameter of the organism is larger than the maximum range of the beta particles in tissue and, (2) no self-shielding of particles. Consequently, it assumes all radiation emitted within an organism is absorbed.

In this study, the bioaccumulation factor defined as (conc. in biota)/(conc. in sediment), was used rather than the expression (conc. in biota/conc. in water) to make the above equation a more realistic dose estimate for benthic macroinvertebrates (See V.3.B.c.).

4. Statistical methods

Invertebrate collections were separated into counts of *H. limbata* and counts of total numbers of invertebrates (standing crop). Data were transformed using $\log_e(x+1)$ to ensure homogeneity of variance

(Snedecor and Cochran, 1967). The null hypothesis of no difference in numbers of invertebrates between sampling transects, periods, and depths, and no interaction between these, was tested using a three-way analysis of variance (AOV) (Snedecor and Cochran, 1967). The multiplicative model can be written:

$$X_{ijk} = \mu . T_i . P_j . D_k . (TP)_{ij} . (TD)_{ik} . (PD)_{jk} . \xi_{ijk}$$

where: μ = mean effect

T_i = transect effect (T5, T4, T3, T2, T1, C1, C2, C3, C4, C5), i = 1 to 10

P_j = period effect (June, July, Aug., Sept.), j = 1 to 4

D_k = depth effect (1m, 2m), k = 1 to 2

$(TP)_{ij}$ = effect due to the interaction of the i^{th} level of T with the j^{th} level of P

$(TD)_{ik}$ = effect due to the interaction of the i^{th} level of T with the k^{th} level of D

$(PD)_{jk}$ = effect due to the interaction of the j^{th} level of P with the k^{th} level of D

ξ_{ijk} = experimental error

All factors in this model were treated as fixed effects. Orthogonal contrasts (Snedecor and Cochran, 1967) were used to compare T- and C-zones, and to determine the source of significant F values from the AOV.

Preliminary data analysis showed no significant difference ($P>0.05$) in diversity between 1 m and 2 m depths on any given transect, so these samples were pooled. Shannon-Weaver diversity index values were calculated for pooled samples on each transect for each sampling period. Diversity index values were calculated at the genus level. Diversities were not calculated for the 1979 data because the chironomids were not identified below family level.

Two-way AOV was applied to the diversity data. The multiplicative model can be written:

$$Y_{ij} = \mu \cdot T_i \cdot P_j \cdot \xi_{ij}$$

where:

T_i = transect effect (T5, T4, T3, T2, T1, C1, C2, C3, C4, C5), i = 1 to 10

P_j = period effect (June, July, Aug., Sept.), $_j$ = 1 to 4

ξ_{ij} = experimental error

All factors in this model were treated as fixed effects. Two-way AOV, using the above model, was also applied to the water temperature and dissolved oxygen data.

CHAPTER V

RESULTS AND DISCUSSION

1. Benthic habitat

(A) Water temperature and dissolved oxygen

In 1978 and 1979, the temperature of Winnipeg River water rose quickly in May and June to about 17°C, reached a maximum of 21 to 23°C at the end of July, and declined gradually in August and September to about 18°C (Fig. 7, 8). The difference between T- and C-zone water temperatures on the same sampling day never exceeded 2.0°C. Daily temperatures were not significantly different (two-way AOV; $P>0.05$) between stations (Table 2). There was no detectable increase in water temperatures beyond a 70 m radius of the effluent outfall (O. Acres, pers. comm.). Thus, heated water released from WNRE effluent outfall was cooled by the Winnipeg River and had no impact on the water temperatures of T-zone.

Percent saturation of oxygen was high during both summers, ranging from 85 to 99% (Table 3). O_2 concentrations and % saturation did not differ significantly (two-way AOV; $P>0.05$) between sampling stations on 25 July 1978 or on 30 August 1979 (Table 2). Thermal discharges from WNRE had no impact on the DO concentration in the river. Uniformly high DO levels are probably due to velocity and turbulence of the river (Ireland *et al.*, 1973).

(B) River discharge

Discharge of the Winnipeg River ranged between 1000 and 1200

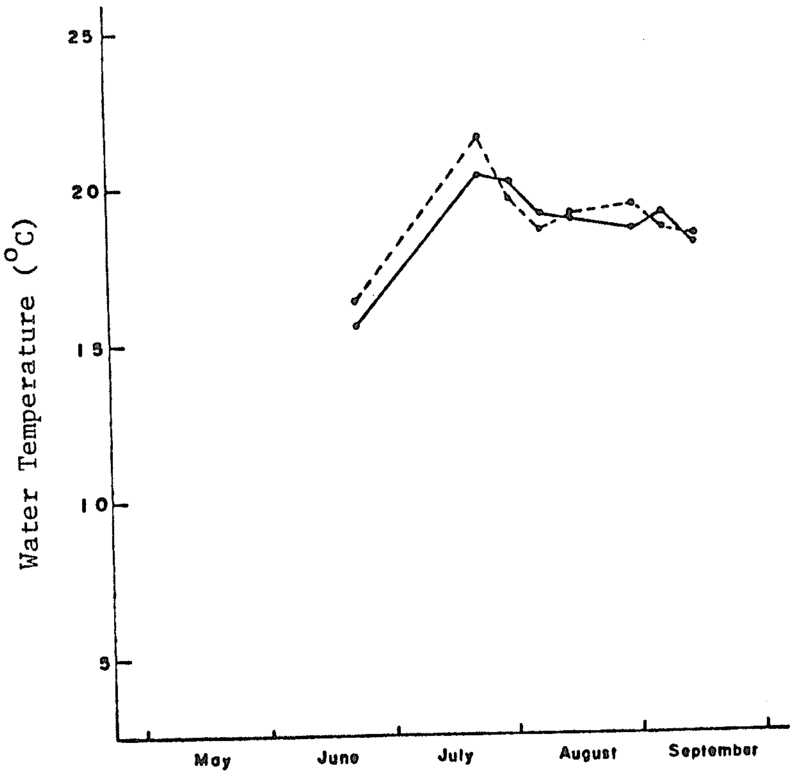


Figure 7. Water temperature of the Winnipeg River in control (-----) and treatment (———) zones in the vicinity of the Whiteshell Nuclear Research Establishment in 1978.

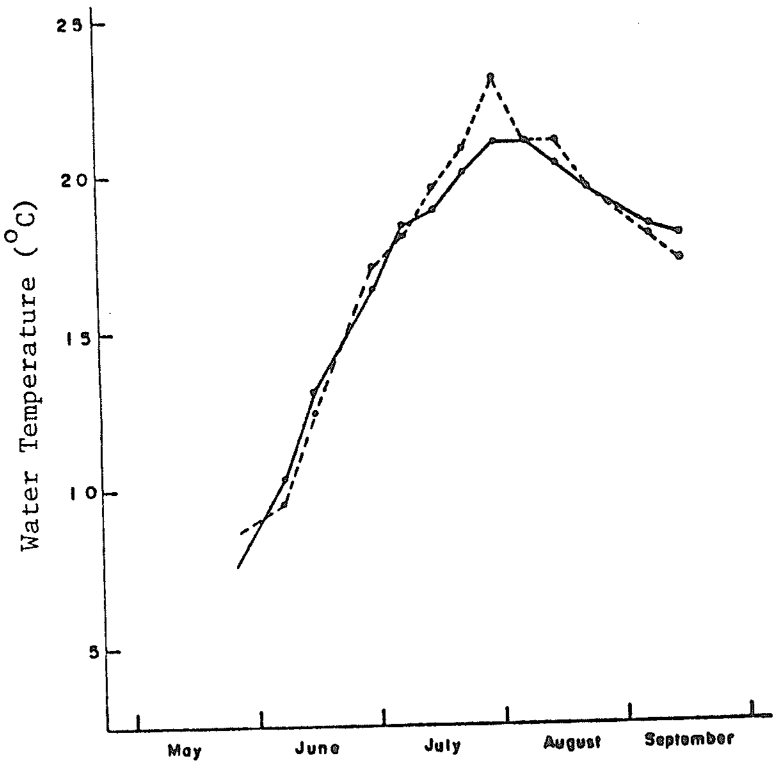


Figure 8. Water temperature of the Winnipeg River in control (-----) and treatment (———) zones in the vicinity of the Whiteshell Nuclear Research Establishment in 1979.

Table 2. Temperatures and dissolved oxygen (DO) of Winnipeg River measured upstream (control zone) and downstream (treatment zone) of the Whiteshell Nuclear Research Establishment on 25 July 1978 and 30 August 1979. Measurements were taken 0.2 m above the bottom substrate.

Station	July 25, 1978		Aug. 30, 1979	
	Temp (°C)	DO (ppm)	Temp (°C)	DO (ppm)
Treatment				
1 - 1 m	20.0	8.7	19.5	8.1
- 2 m	20.0	8.5	19.0	8.4
2 - 1 m	20.5	8.3	19.5	8.4
- 2 m	21.0	8.3	19.5	8.1
3 - 1 m	20.0	8.3	18.5	8.6
- 2 m	20.5	8.6	19.0	8.7
4 - 1 m	20.0	8.6	19.0	8.1
- 2 m	20.0	8.3	19.0	8.2
5 - 1 m	20.5	8.7	19.5	8.1
- 2 m	20.5	8.4	19.0	8.4
Control				
1 - 1 m	20.0	8.3	19.0	8.0
- 2 m	20.5	8.1	19.0	8.0
2 - 1 m	20.5	8.7	19.5	8.6
- 2 m	20.5	8.7	20.0	8.6
3 - 1 m	20.5	8.8	19.5	8.1
- 2 m	20.5	8.4	19.5	8.3
4 - 1 m	21.0	8.5	19.0	8.7
- 2 m	21.0	8.3	19.5	8.1
5 - 1 m	20.0	8.3	19.5	8.1
- 2 m	20.0	8.1	19.5	8.2

Table 3. Dissolved oxygen (DO) in the Winnipeg River upstream (control zone) and downstream (treatment zone) of the Whiteshell Nuclear Research Establishment. Measurements were taken 0.2 m above the bottom substrate. Means of two measurements are given.

	1978				1979			
	Treatment		Control		Treatment		Control	
Month and week	DO (ppm)	DO (% satn)	DO (ppm)	DO (% satn)	DO (ppm)	DO (% satn)	DO (ppm)	DO (% satn)
May 1	-	-	-	-	12.1	99	11.6	97
" 2	-	-	-	-	-	-	-	-
June 1	-	-	-	-	11.0	97	10.8	96
" 2	-	-	-	-	9.5	90	9.6	85
" 3	9.0	91	8.7	88	-	-	-	-
" 4	-	-	-	-	8.8	89	8.8	91
July 1	-	-	-	-	8.2	86	8.1	85
" 2	-	-	-	-	8.5	91	8.4	91
" 3	8.7	95	8.5	97	8.6	93	8.7	97
" 4	8.7	95	8.3	93	8.3	92	8.1	93
Aug. 1	8.6	92	8.7	90	8.3	92	8.4	95
" 2	8.5	91	8.1	87	8.1	88	8.7	97
" 3	-	-	-	-	8.4	91	8.8	96
" 4	8.4	90	8.4	92	-	-	-	-
Sept. 1	8.3	89	8.6	87	8.1	85	8.5	89
" 2	8.5	89	8.6	91	8.4	88	8.2	85

m^3/s during May, June and August 1978 (Fig. 9), with increases up to 1400 m^3/s in mid-July and late September. Fluctuation of discharge was greater in 1979, reaching a maximum of 1700 m^3/s in late May and declining thereafter. Discharge remained comparatively stable during August and September (500-700 m^3/s). Manitoba Hydro lowered the water level at Seven Sisters Generating Station about 1 m in spring and early summer 1979 to accommodate dam repairs. The consequent release of water upstream increased discharge in early summer, but low upstream water levels in late summer decreased discharge.

Although there was concern that fluctuating discharge may severely affect water levels at the sampling stations, water levels fluctuated only ± 0.1 m between May and September 1978. Fluctuations were more pronounced in 1979, with water levels dropping about 0.2 m between May and August. However, invertebrates in the sampling areas apparently were unaffected by this decrease in water levels.

(C) Current velocity

Water velocities at the sampling stations ranged between 3.7 and 11.9 cm/s (Table 4) whereas a velocity of 30 cm/s was recorded in the middle of the river. The current velocity at all sampling stations was <20 cm/s, the velocity under which fine suspended material will settle out (Hynes, 1970). As a result, fine-grained sediments predominated in both sampling zones. Although there were insufficient data to determine significant differences in current speeds between stations, the predominance of very fine sand at C-zone stations compared to clay at T-zone stations (V. 1. F) suggests that current velocity was slightly greater in C-zone.

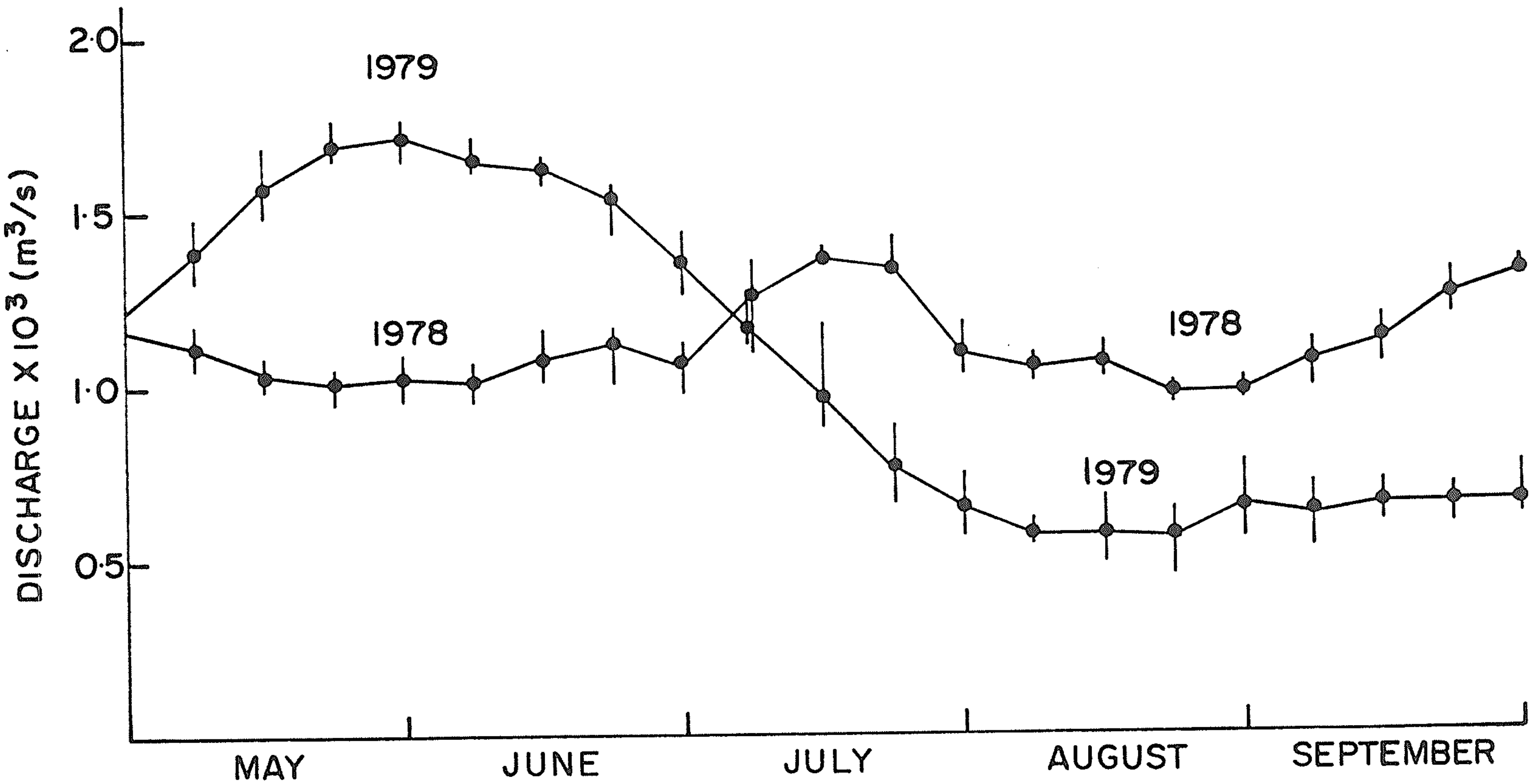


Figure 9. Discharge of the Winnipeg River during 1978 and 1979, measured at the Seven Sisters Hydroelectric Station. Mean weekly rates are indicated by dots. Vertical bars indicate weekly fluctuations.

Table 4. Water velocity (cm/s) 0.2 m above the bottom substrate upstream (control) and downstream (treatment) from the Whiteshell Nuclear Research Establishment on 24 August 1979.

	Sampling zones									
	Treatment					Control				
Depth contour	T5	T4	T3	T2	T1	C1	C2	C3	C4	C5
1 m	8.2	3.7	5.8	4.9	4.0	11.9	6.1	9.5	9.8	11.6
2 m	8.5	8.5	9.4	9.1	6.7	6.7	7.2	9.1	9.4	11.3

(D) Water transparency

The mean Secchi disc depth for 1978 in T-zone (Table 5) was fairly constant at 0.97 m, range 0.79 to 1.22 m. In 1979, water transparency was slightly greater during May and June, than during July through September (Table 5). Control Secchi disc measurements were taken 5 km upstream of Seven Sisters Hydro Electric Dam in 1979. Water transparency ranged from 1.21 to 1.31 m in May and June, and was lower during the rest of the summer ($\bar{x}$ = 1.05 m).

There appeared to be a slight seasonal fluctuation in the transparency of Winnipeg River water, but there was no difference between upstream and downstream of WNRE. Although there is usually a positive correlation between discharge and turbidity (Hynes, 1971), the transparency of Winnipeg River water was greatest during periods of high discharge. The decrease in transparency from July through September may be due to increased phytoplankton growth caused by the warmer water temperatures.

(E) Aquatic macrophytes

Potamogeton richardsonii (pondweed), *Vallisneria americana* (tape grass), and *Elodea canadensis* (water weed) were collected from T- and C-zone sampling stations (Table 6). *P. richardsonii* and *V. americana* were the most widely distributed aquatic plants. *P. richardsonii* was found at all sampling stations, except T1-2 m and C1-2 m. *V. americana* was found at all sampling stations except T1-1 m. *E. canadensis* occurred at 10 of the 20 sampling stations. At least two of the plant species were found at each sampling station. Plant abundances were lowest at the transect immediately downstream from the outfall (Table 6), possibly a reflection of an environmental disturbance caused by WNRE's effluent.

Table 5. Secchi disc depths in the Winnipeg River during 1978 and 1979. T-zone is downstream from the Whiteshell Nuclear Research Establishment (WNRE); the Seven Sisters zone is upstream from WNRE.

Month and week	Secchi disc depth (m)		
	T-zone (1978)	T-zone (1979)	Seven Sisters zone (1979)
May 1	-	1.22	-
" 2	-	-	1.30
June 1	-	1.28	1.21
" 2	-	-	-
" 3	-	-	-
" 4	-	1.09	-
July 1	-	0.99	1.10
" 2	1.22	-	-
" 3	0.95	1.10	0.94
" 4	0.79	1.07	-
Aug. 1	-	1.28	-
" 2	1.09	-	1.15
" 3	-	1.07	-
" 4	0.79	-	1.03
Sept. 1	-	1.09	-
" 2	0.95	0.99	-

Table 6. Species and relative abundances of macrophytes at treatment (T) and control (C) stations in the vicinity of the Whiteshell Nuclear Research Establishment. + = presence of a species. xxx = >1 macrophyte/400 cm^2; xx = 1 macrophyte/1600 cm^2; x = 1 macrophyte/3200 cm^2.

Sampling station	Potamogeton richardsonii (pondweed)	Vallisneria americana (tape grass)	Elodea canadensis (waterweed)	Relative abundance of macrophytes
T1-1m	+		+	x
T1-2m		+	+	x
T2-1m	+	+		x x
T2-2m	+	+	+	x x x
T3-1m	+	+		x x
T3-2m	+	+		x x x
T4-1m	+	+		x x
T4-2m	+	+		x x x
T5-1m	+	+	+	x x x
T5-2m	+	+		x x
C1-1m	+	+	+	x x x
C1-2m		+	+	x x x
C2-1m	+	+		x x x
C2-2m	+	+		x x x
C3-1m	+	+	+	x x
C3-2m	+	+	+	x x
C4-1m	+	+		x x
C4-2m	+	+	+	x x
C5-1m	+	+	+	x x
C5-2m	+	+		x x

Ireland (1968) also found plant abundances to be low in this area.

(F) Bottom sediment

Clay sediments predominated in T-zone with T1-1 m, T1-2 m, T5-1 m, and T5-2 m almost entirely clay, and the remaining stations comprised of nearly equal proportions of clay, silt, and very fine sand (Table 7). Sediments in C-zone were chiefly very fine sand with small percentages of clay and silt, except for C4-1 m and C4-2 m which were comprised of equal amounts of clay, silt and very fine sand. Sediments at all T- and C-zone stations contained small amounts (∿5% of grab sample volume) of plant debris. However, plant debris at T4-2 m and C4-1 m composed about 20% of the sample volume. Sediment type did not change at any sampling station during the study period.

Sediment type is the most important factor controlling benthic community characteristics at any given location (EPA, 1973). Consequently, only fauna from sites having similar substrates provide valid data for comparison (EPA, 1973; Ruggiero and Merchant, 1979). All the sampling stations used in this study were dominated by burrowing benthic macroinvertebrates, especially Hexagenia limbata, chironomids, and oligochaetes. These taxa are characteristic of fine particulate sediments. The minimal variation in sediment type between sampling stations did not appear to influence the benthic invertebrate fauna and, thus, the comparability of the stations.

(G) Organic coolant

A viscous film of organic coolant was visible on the surface of the sediment samples from T1 in 1978, but was not perceptible in the

Table 7. Bottom sediment at treatment and control stations in the Winnipeg River near the Whiteshell Nuclear Research Establishment. v.f. sand = very fine sand; org. db. = organic debris.

Transect	Depth	
	1m	2m
<u>Treatment</u>		
1	clay	clay
2	clay + silt + v.f. sand	clay + silt + v.f. sand
3	clay + silt + v.f. sand	clay + silt + v.f. sand
4	clay + silt + v.f. sand	clay + silt + v.f. sand + org. db.
5	clay	clay
<u>Control</u>		
1	v.f. sand	v.f. sand
2	v.f. sand	v.f. sand
3	v.f. sand	v.f. sand
4	clay + silt + v.f. sand + org. dbr.	clay + silt + v.f. sand + org. db.
5	v.f. sand	v.f. sand

other T-zone sediment samples at this time. Although the concentration of coolant was not measured in this study, the concentration in the region of T1 in 1977 (Guthrie and Acres, 1979) was about 8 g/m^2.

In 1979, coolant was plainly visible on the surface of all sediment samples taken in T-zone. The largest amounts were found in the samples from T1, where it was present as a viscous film. Coolant at the remaining stations may have been the result of a recent leak from WNRE. Conversely it may have been washed in from deeper parts of the T-zone bay where the concentrations were comparatively high (Guthrie and Acres, 1979), or transported by the river current from the region of high deposition in the immediate vicinity of the outfall.

(H) Radioactivity

Radioactivity in the benthic habitat in the vicinity of WNRE is discussed in V. 3.

(I) Summary and conclusion

a. Benthic habitat and comparability of sampling stations

The benthic habitat at the study sites is characterized by low river current velocity, soft fine-particulate sediments, and three species of deep-rooted macrophytes. Benthic faunas probably are comparable from station to station because of the similarity of environmental characteristics. Fluctuations in river discharge occurred during the study period, but the resulting minimal changes in water level are considered to have inconsequential effects on the benthos.

b. Effects of heated water and organic coolant on the benthic habitat

No increase in ambient water temperatures attributable to WNRE liquid effluent was detected at any of the sampling stations. Therefore, heated water released from WNRE was not a limiting factor for the benthic fauna. Visual inspection indicated that the sediment samples taken in 1978 and 1979 immediately downstream of the outfall contained the largest amounts of organic coolant. Smaller amounts of coolant were found in the sediment further downstream in 1979.

2. Benthic macroinvertebrates

(A) Introduction

Standing crops, diversity indices, and densities of benthic macroinvertebrates, were compared between sampling transects to ascertain if WNRE effluents have disturbed these populations in the Winnipeg River.

(B) General description of the benthic macroinvertebrate fauna

Organisms collected in 1978 and 1979 are listed in Appendices B1-4 and C1-4. Seventy taxa were collected during 1978 and 1979. The benthic fauna was dominated numerically by Hexagenia limbata (Serville), larval Chironomidae (26 genera), Hyalella azteca (Saussure), Pelecypoda, Gastropoda, and Oligochaeta (Tables 8, 9). These taxa accounted for 86% of the total number of organisms collected.

(C) Ephemeroptera

Ephemeropteran nymphs comprised 23% of the macroinvertebrate

Table 8. Macroinvertebrates collected in all samples taken upstream (control) and downstream (treatment) from the Whiteshell Nuclear Research Establishment in 1978.

Taxon	Number collected	Mean density (no./m^2)	% Composition of total no. of orgs. collected	Taxa within major taxonomic group
Ephemeroptera	1074	449.2	23.1	7
Odonata	11	4.6	.2	2
Plecoptera	0			
Megaloptera	28	11.7	.6	1
Trichoptera	34	14.2	.7	6
Coleoptera	101	42.1	2.2	4
Diptera (excluding Chironomidae)	80	33.3	1.7	5
Chironomidae	1163	484.6	25.0	26
Acarina	7	2.9	.2	1
Amphipoda	1000	416.7	21.5	5
Mollusca	537	223.8	11.5	2
Annelida	564	235.0	12.1	2
Nematoda	43	17.9	.9	1
Others (unidentified)	14	5.8	.3	-
TOTALS	4660	1941.8	100.0	64

Table 9. Macroinvertebrates collected in all samples taken upstream (control) and downstream (treatment) from the Whiteshell Nuclear Research Establishment in 1979.

Taxon	Number collected	Mean density (no./m^2)	% Composition of total no. of orgs. collected	Taxa within major taxonomic group
Ephemeroptera	1144	476.7	24.1	5
Odonata	6	2.5	.1	2
Plecoptera	18	7.5	.4	1
Megoloptera	41	17.1	.9	1
Trichoptera	33	13.7	.7	4
Coleoptera	117	48.7	2.5	4
Diptera (excluding Chironomidae)	169	70.4	3.6	3
Chironomidae	806	335.8	17.0	1
Amphipoda	360	148.9	7.6	4
Mollusca	924	384.9	19.5	2
Annelida	1036	442.9	22.4	2
Nematoda	59	24.6	1.2	1
Others	1	.4	.1	1
TOTALS	4741	1975.1	100.0	30

community in 1978 (Table 8), and 24% in 1979 (Table 9). *Hexagenia limbata* comprised 89-96% of the Ephemeroptera collected and was the most abundant invertebrate collected during the study (Tables 10, 11). *H. limbata* was the most abundant macroinvertebrate in the Winnipeg River in the vicinity of WNRE in 1966 and 1967 as well (Ireland, 1968). Nymphs of *H. limbata* construct U-shaped burrows in the bottom sediment, and are largely restricted to a substratum which is soft, yet firm enough to permit maintenance of the burrow (Edmunds *et al.*, 1976). The nymphs, often found in large lakes and shore regions of large rivers (Fremling, 1970), are abundant in Lake Winnipeg (Flannagan, 1970) and the Winnipeg River (Gregory and Loch, 1973). Two size groups of *H. limbata* nymphs were found in the sediment samples. This suggests the presence of two distinct cohorts, each having a generation time of about 2 years, with emergence in alternate years. A similar life cycle was reported by Ireland (1968).

There was a significant difference ($P<0.01$) in the number of *H. limbata* collected between sampling months in 1978 (Table 12). These temporal changes are to be expected from their life cycle patterns (Hunt, 1953). Following a peak of numbers in July (Table 10), the number of nymphs declined due to emergence of the adults. Numbers obtained in August were relatively low. The September increase in numbers consists of one year old nymphs and newly hatched nymphs.

There were no significant differences ($P>0.05$) between sampling transects or depths (Table 12) in 1978. However, densities of *H. limbata* in T-zone were lowest at T1, the transect immediately downstream of the outfall (Fig. 10). The density of nymphs declined in T-zone and increased

Table 10. Numbers of nymphal *Hexagenia limbata* collected upstream (control-zone) and downstream (treatment-zone) of the Whiteshell Nuclear Research Establishment in 1978. Samples from 1 m and 2 m were pooled.

	Treatment transects						Control transects						
	5	4	3	2	1	$\bar{x}$	1	2	3	4	5	$\bar{x}$	$\bar{x}$
June	11	40	71	44	6	34.4	17	21	19	11	21	17.8	26.1
July	26	79	63	40	13	44.2	3	6	13	20	16	11.6	27.9
Aug.	21	19	22	10	10	16.4	2	4	12	12	13	8.6	12.5
Sept.	51	49	8	33	19	32.0	17	27	48	20	23	27.0	29.9
$\bar{x}$	27.3	46.8	41.0	31.8	12.0	31.8	9.8	14.5	23.0	15.8	18.3	16.3	24.1

Table 11. Numbers of nymphal *Hexagenia limbata* collected upstream (control-zone) and downstream (treatment-zone) of the Whiteshell Nuclear Research Establishment in 1979. Samples from 1 m and 2 m were pooled.

	Treatment transects						Control transects						
	5	4	3	2	1	$\bar{x}$	1	2	3	4	5	$\bar{x}$	$\bar{x}$
June	5	2	1	4	4	3.2	15	9	20	2	24	14.0	8.6
July	34	40	37	62	24	39.4	56	20	42	33	35	37.2	38.3
Aug.	21	25	38	32	2	23.6	36	30	32	26	18	28.4	26.0
Sept.	20	18	23	21	8	18.0	32	57	51	40	41	44.2	31.1
$\bar{x}$	20	21.3	24.8	29.8	9.5	21.1	34.8	29.0	36.3	25.3	29.5	31.0	26.0

Table 12. Analysis of variance for differences between numbers of *Hexagenia limbata* at sampling transects, depths, and periods, in the vicinity of the Whiteshell Nuclear Research Establishment in 1978. T = treatment; C = control; **P = 0.01. Orthogonal contrasts are in parenthesis.

Source of variance	Degrees of freedom	Sum of squares	Mean square	F-statistic
Main effects				
Transect (T5,T4,T3,T2,T1,C5,C4,C3,C2,C1)	9	13.74	1.53	2.15
(T1 vs all other transects)	(1)	(2.54)	(2.54)	(3.58)
(Treatment vs control)	(1)	(4.74)	(4.74)	(6.68)**
Period (June, July, Aug., Sept.)	3	6.25	2.11	2.97**
Depth (1m, 2m)	1	1.30	1.30	1.83
Interaction effects				
Transect x period	27	15.27	.57	.80
Transect x depth	9	5.17	.57	.80
Period x depth	3	4.63	1.54	2.17
Experimental error	27	19.14	.71	
Total	79	65.50		

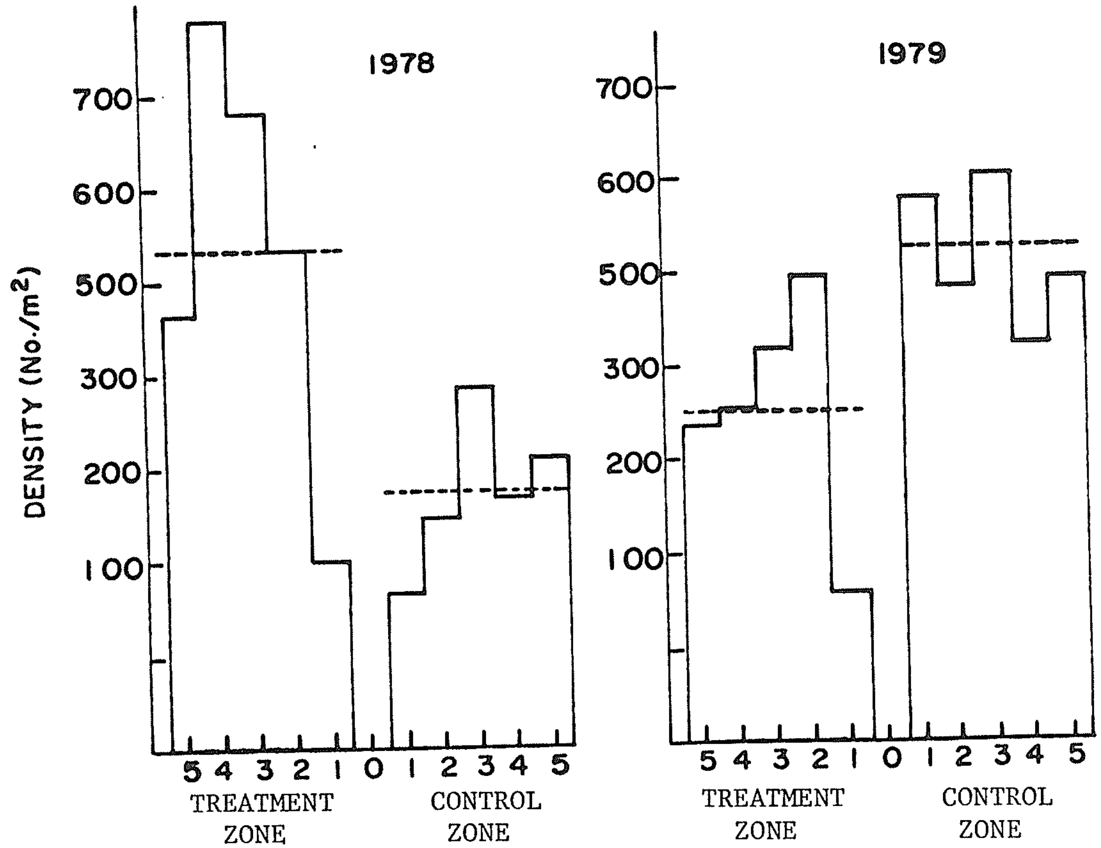


Figure 10. Mean density of Hexagenia limbata nymphs at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

in C-zone by 1979 (Fig. 10) although their density at T1 remained low. H. limbata densities were significantly lower ($P<0.01$) in T-zone than in C-zone (Table 13), and were significantly lower ($P<0.05$) at T1 relative to all the other transects.

Liquid effluents released from WNRE may have detrimentally affected the H. limbata community at T1. There appeared to be a correlation between low numbers of H. limbata and the presence of organic coolant in sediments. T1 was the only transect where organic coolant was found in both years. By 1979, organic coolant was present at all T-zone stations and H. limbata - densities declined, relative to 1978. However, the densities of this species also changed in the C-zone from 1978 to 1979 (Fig. 10), indicating natural fluctuations in H. limbata populations. Nevertheless, the consistently low densities recorded immediately downstream of the outfall suggest that some disturbance has occurred.

Other Ephemeropterans collected, in decreasing order of abundance were: Caenis spp., Ephemera simulans (Walker), Heptagenia spp., Siphlonurus spp., Ephemerella temporalis (McDunnough), and Hexagenia rigida (McDunnough). Only Caenis spp. comprised >1% of the total benthic population. T- and C-zone samples had similar numbers of Caenis spp., except for the relatively large abundances of nymphs found at C4 and C5 (Fig. 11). Unlike H. limbata, Caenis spp. nymphs did not appear to be detrimentally affected by effluent releases. The burrowing mayfly, E. simulans, was more abundant in T- than in C-zone during both years, but the population in T-zone declined between 1978 and 1979. Numbers of other species of Ephemeroptera were too low to analyze.

Table 13. Analysis of variance for differences between numbers of *Hexagenia limbata* at sampling transects, depths, and periods, in the vicinity of the Whiteshell Nuclear Research Establishment in 1979. T - treatment; C = control. *P = 0.05; **P = 0.01. Orthogonal contrasts are in parenthesis.

Source of variance	Degrees of freedom	Sum of squares	Mean square	F-statistic
Main effects				
Transects (T5,T4,T3,T2,T1,C1,C2,C3,C4,C5)	9	12.01	1.33	3.41**
(Treatment vs control)	(1)	(5.27)	(5.27)	(13.51)**
(T1 vs all other transects)	(8)	(7.51)	(.94)	(2.41)*
Period (June, July, Aug., Sept.)	3	31.22	10.41	26.69**
Depth (1m, 2m)	1	0.66	0.66	1.69
Interaction effects				
Transect x period	27	14.72	0.55	1.41
Transect x depth	9	3.72	0.41	1.05
Period x depth	3	0.45	0.15	0.38
Experimental error	27	10.42	0.39	
Total	79	73.20		

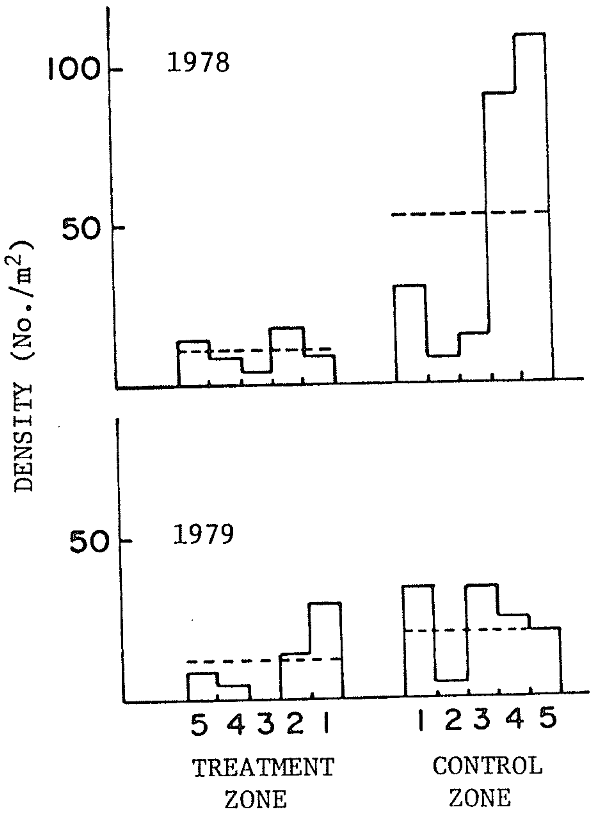


Figure 11. Mean density of <u>Caenis</u> spp. nymphs at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

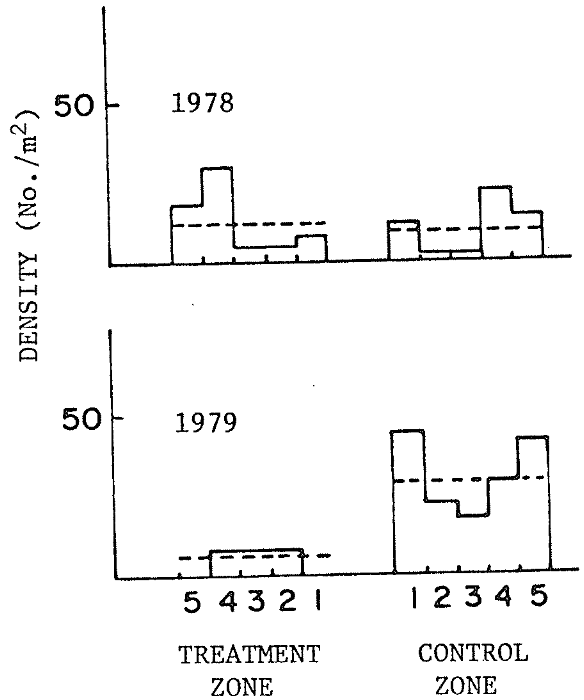


Figure 12. Mean density of <u>Sialis</u> spp. larvae at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

(D) Minor orders

Individuals of Odonata, Plecoptera, Megaloptera, and Trichoptera each comprised <1% of the total number of organisms collected during 1978 (Table 8) and 1979 (Table 9). Realistic numerical estimates for the Odonata *Ophiogomphus columbrinus* (Selys) and *Enallagma* spp. were not obtained because the former species can avoid the sampler and the latter is found in vegetation, a habitat not sampled.

Acroneuria spp. were the only Plecopteran nymphs collected. Numbers probably would have been higher if rocks or fallen logs, their preferred habitats (Frison, 1935), were sampled specifically. Plecopteran nymphs are sensitive to stresses such as organic wastes, low O_2, and biocides (Roback, 1974) so the presence of these nymphs indicates high water quality (Patrick, 1953). Although higher numbers of Plecopterans were collected in T-zone than C-zone, the small numbers of specimens collected preclude any conclusive statements concerning water quality.

Larvae of *Sialis* spp. (Megaloptera) were collected at all sampling transects, and were almost equally abundant in both zones in 1978 (Fig. 12). In 1979, however, their numbers had declined in T-zone, and none were found at T1 or T5. The smaller numbers of *Sialis* spp. in the T-zone in 1979 also suggests the occurrence of an environmental disturbance.

Trichoptera generally are not abundant in the Winnipeg River (Crowe, 1971; Gregory and Loch, 1973). The seven genera collected, in decreasing order of abundance, were: *Phylocentropus* spp., *Hydroptila* spp., *Polycentropus* spp., *Agraylea* spp., *Pycnopsyche* spp., *Agrypnia* spp., and *Rhyacophila* spp. The three most abundant genera prefer soft sediment (Wiggins, 1977), so bottom sediment sampling should provide adequate estimates of their population densities. *Phylocentropus* spp., *Hydroptila* spp., and *Polycentropus* spp. were each equally abundant in both T- and C-zones, however, their low numbers overall preclude conclusive statements concerning water quality.

(E) *Coleoptera*

Larval Coleoptera comprised about 2% of the total number of organisms collected in 1978 and 1979 (Tables 8, 9). The elmid, *Dubiraphia* spp. comprised 79-92% of the beetles collected. There are no previous records of this genus occurring in the Winnipeg River. The density of *Dubiraphia* spp. was low at T1 in 1978 and 1979 (Fig. 13), and declined at the remaining T-zone transects in 1979. Liquid effluent released from WNRE may have detrimentally affected the *Dubiraphia* community at T1. Also the community may have been affected further downstream in 1979, however low numbers at this time may be due to natural fluctuations.

Other beetle taxa collected, in decreasing order of abundance were: *Donacia* spp., *Haliplus* spp., *Galerucella* spp., *Gyrinus* spp., and *Coptotomus* spp. These taxa are swimmers and climbers (Merritt and Cummins, 1978) therefore realistic numerical estimates could not be obtained from samples of bottom sediment.

(F) *Diptera*

Larvae of Chironomidae were the predominant Dipterans collected, comprising 25% of the total number of organisms collected in 1978 and 17% in 1979 (Tables 8, 9). The mean densities of Chironomidae

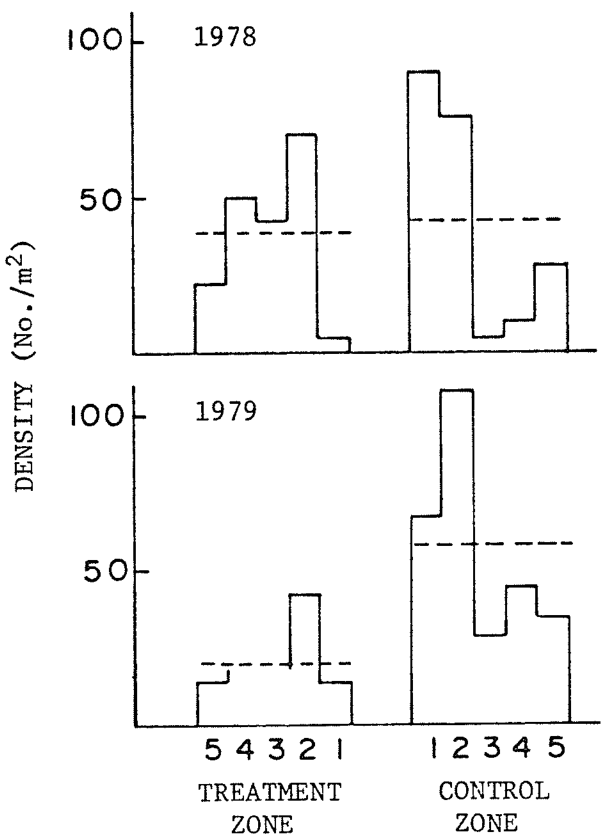


Figure 13. Mean density of <u>Dubiraphia</u> spp. larvae at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

in T- and C-zones were similar in 1978 (Fig. 14), but the density at T1 was lower than at all other transects. Mean larval densities at T1 did not change from 1978 to 1979 but they declined at each of the other T-zone transects in 1979. There was an increase in larval densities at C1, and a decline at the other C-zone stations between 1978 and 1979. Large variations in larval densities were apparent in C-zone stations in 1979. Low densities of Chironomidae at T1 in 1978 may be due to WNRE effluent and the presence of organic coolant at T2-5 in 1979 may have depressed the densities. However, variable chironomid densities in C-zone in 1979 indicate natural fluctuations in the Winnipeg River chironomid population, so declines in T-zone in 1979 may be ascribed to natural changes in their population density.

Twenty-six genera of Chironomidae were identified in 1978 (Table 14). *Procladius* spp., *Microtendipes* spp., *Endochironomus* spp., *Tanytarsus* spp., *Paratanytarsus* spp., *Cryptochironomus* spp., and *Ablabesmyia* spp. were found at all sampling transects and comprised 72% of the total number of chironomid larvae collected. T- and C-zones had similar generic compositions (Table 14), with a slightly higher occurrence of rare genera at the T-zone transects ($\bar{x}$ no. of genera = 18; range 16-20) and C-zone transects ($\bar{x}$ = 16; range = 14-16). In T-zone, the lowest number of genera was found at T1. However, lower numbers of genera were found at most C-zone stations. Although there was no evidence that suggested WNRE effluents had altered the generic composition at T1, the comparatively low abundance of some major taxa (*Procladius* spp., *Tanytarsus* spp., *Paratanytarsus* spp., *Cryptochironomus* spp., *Ablabesmyia* spp.) (Table 14) suggested some disturbance had occurred at this

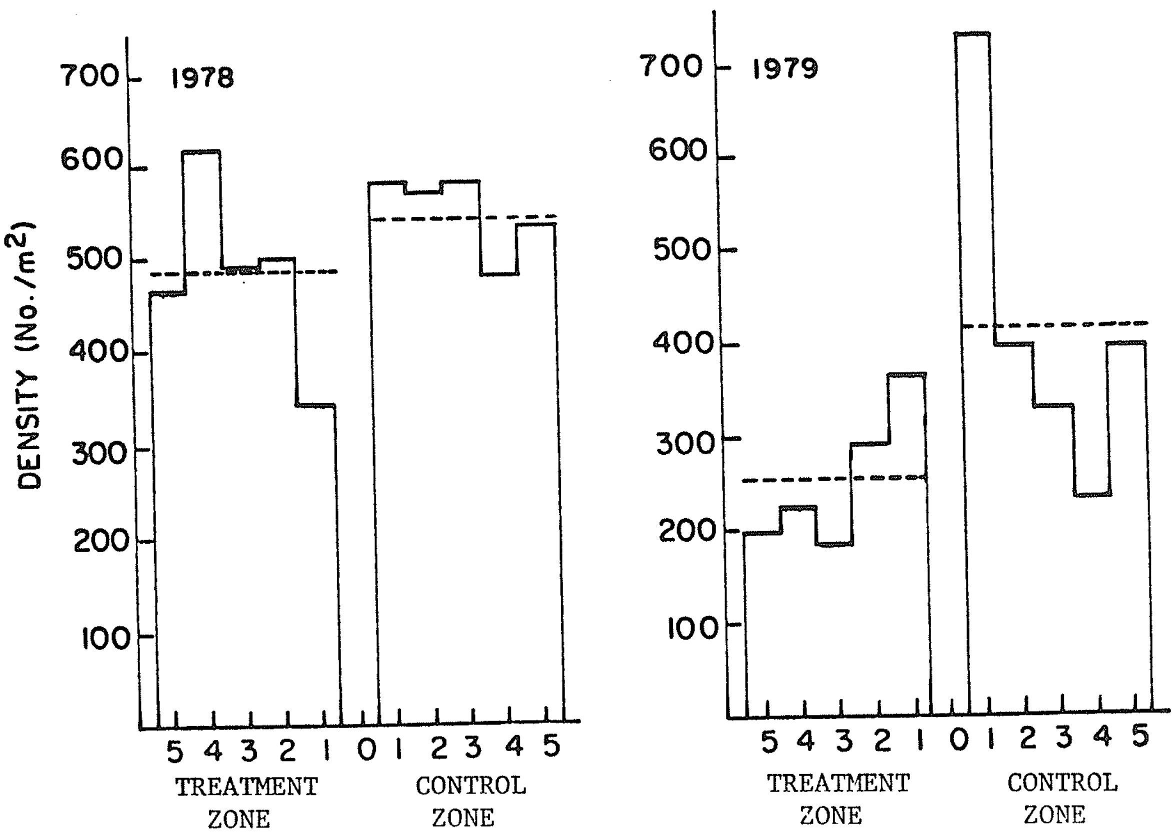


Figure 14. Mean density of larval Chironomidae at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

Table 14. Numbers of larval Chironomidae collected upstream (control) and downstream (treatment) of the Whiteshell Nuclear Research Establishment in 1978.

	Treatment transects					Control transects					Total	
	5	4	3	2	1	1	2	3	4	5	Treatment	Control
Procladius spp.	13	41	11	29	4	24	13	5	26	31	98	99
Microtendipes spp.	10	16	2	6	17	9	5	70	20	5	51	109
Endochironomus spp.	7	11	45	11	27	6	5	2	12	8	101	33
Tanytarsus spp	11	16	14	27	3	5	17	2	6	3	71	33
Paratanytarsus spp.	3	6	2	3	1	26	28	7	8	17	15	86
Cryptochironomus spp.	9	6	7	10	1	14	9	15	3	6	33	47
Ablabesmyia spp.	4	16	15	17	1	1	1	1	5	13	53	21
Orthocladiinae #3	17	14	2	1	2			15	3	3	36	21
Glyptotendipes spp.	3	18		1	8		2	5	13	1	30	21
Stempellina spp.	3	2	1	1		6	13	6		3	7	28
Demicryptochironomus spp.	1	2	2	2	2	3	10	1	1	7	9	22
Orthocladiinae #4		1	1	1		13	15				3	28
Polypedilum spp.	1	2	2	4	4	6	7			3	13	16
Stictochironomus spp.	3		1		1	16		5	1	2	5	24
Orthocladiinae #5	4	4			2	1	2	1			10	4
Paralauterborniella spp.	1		2	1		2			4	1	4	7
Orthocladiinae #2		2	1	1		1	1		1	1	4	4
Dicrotendipes spp.		1	1	1	1					2	4	2
Paratendipes spp.	1					3	1				1	4
Orthocladiinae #1				1			2			1	1	3
Phaenopsectra spp.		1		1	1						3	0
Brillia spp.			1	1	1						3	0
Cricotopus spp.										2	0	2
Conchapelopia spp.							2				0	2
Pagastiella spp.			1								1	0
Total number of Chironomidae	94	160	113	120	76	136	133	139	115	128	563	651
Total number of genera	17	18	19	20	16	16	17	14	14	19	24	23

transect. Species of Procladius, Ablabesmyia, and Tanytarsus are known to be intolerant of organic contaminants (EPA, 1973; Gregory and Loch, 1973; Roback, 1974).

The remaining Dipterans collected, in decreasing order of abundance were, Ceratopogonidae, Chrysops spp., Simulium spp., Tipulidae, and Atherix variegata. Comparatively low densities of ceratopogonid larvae, mostly Probezzia spp., at T1 in 1978 and 1979 (Fig. 15) may indicate a disturbance close to the outfall. The large increase in ceratopogonid density which occurred in C-zone from 1978 to 1979, did not occur in the T-zone population. This may indicate that the presence of organic coolant at T2-5 in 1979 depressed the numbers. Alternatively, natural spatial fluctuation may be responsible. The other Dipterans were relatively rare in both zones (Appendix B1-4).

(G) Crustacea

The amphipod Hyalella azteca (Saussure), comprised about 20% of the total number of invertebrates collected in 1978 (Table 8), and 7% in 1979 (Table 9). In 1978, the average density of H. azteca was similar in both T- and C-zones (Fig. 16), but densities were more variable among the C-zone transects. By 1979, there was a decline in the density of H. azteca in both zones, but the means remained similar. The similarity between T- and C-zone densities indicates H. azteca populations in the T-zone have not been disturbed by WNRE liquid effluents.

Cladocera, Anostraca, Decapoda, and Ostracoda were also collected but non-quantitatively, because of either their small size or mobility.

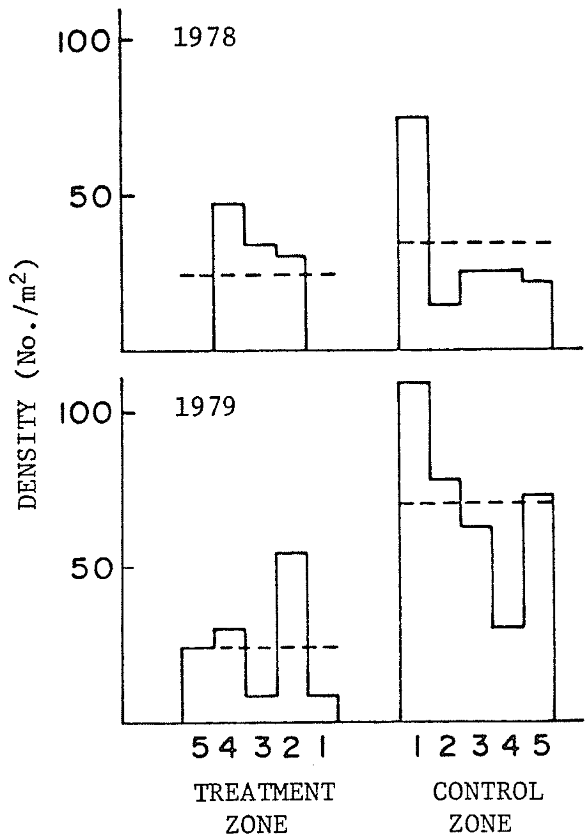


Figure 15. Mean density of larval Ceratopogonidae at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

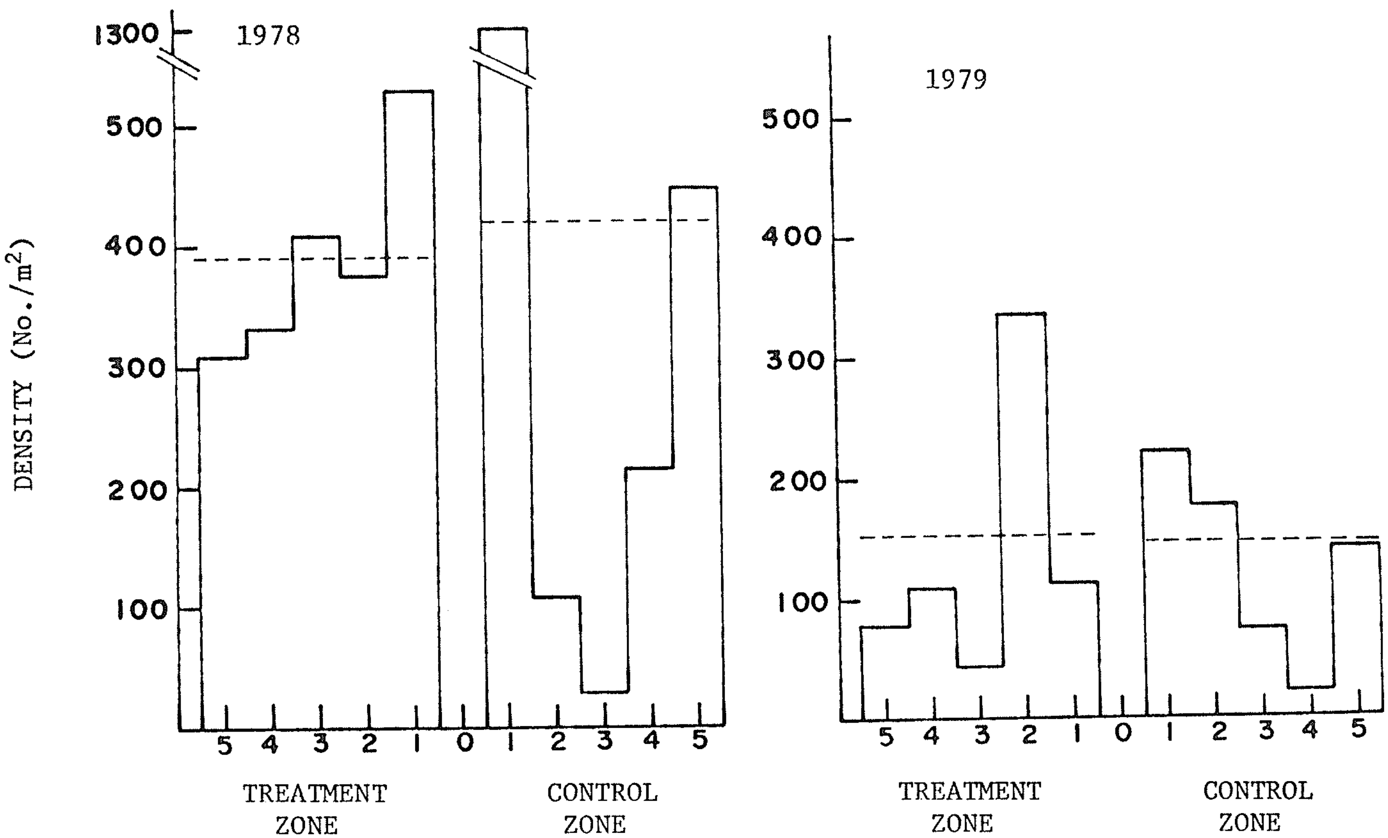


Figure 16. Mean density of Hyalella azteca at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

(H) Mollusca

The molluscs made up a significant portion of the macroinvertebrate community (Table 8, 9). Most of the gastropods collected were *Amnicola* spp., but several specimens of *Campeloma* spp. and *Helisoma* spp. were also found. The mean density of Gastropoda was slightly higher in T-zone than C-zone in 1978 and 1979 (Fig. 17). There were large variations in density between the transects in each zone. The relatively low densities at some T-zone stations may have resulted from the apparent unequal spatial distribution of gastropods in the Winnipeg River. However, the consistently lower densities at T1 than at any other transect during both years may reflect a disturbance to gastropods near the WNRE effluent outfall.

Fingernail clams (*Sphaerium* and *Pisidium*) were the only pelecypods collected. The mean density of Pelecypods was greater in T-zone in 1978 and 1979 (Fig. 18). The pelecypod density at T1 was relatively low at T1 in 1978, but was higher in 1979. There is no indication that Pelecypoda were disturbed by WNRE effluents.

(I) Annelida

Oligochaeta were the most common annelids collected. They tended to break into pieces during sampling and washing. Consequently, density estimates were inaccurate. Densities were higher in the T-zone than C-zone in 1978 and 1979 (Fig. 19). Densities were consistently lower close to the outfall in T-zone, suggesting that WNRE's effluent may be responsible.

(J) Standing crop

There was a significant difference ($P<0.01$) in invertebrate

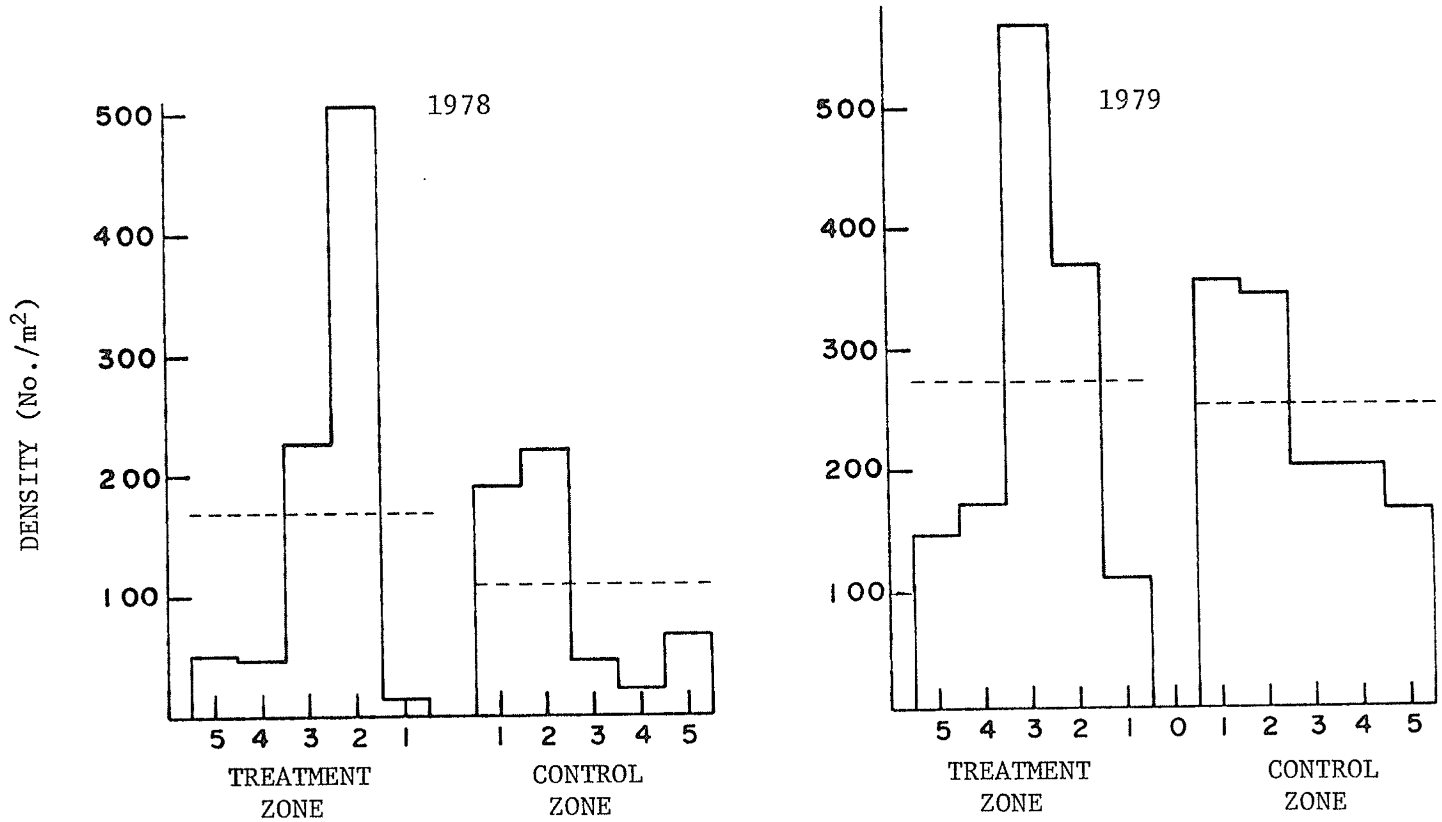


Figure 17. Mean density of Gastropoda at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

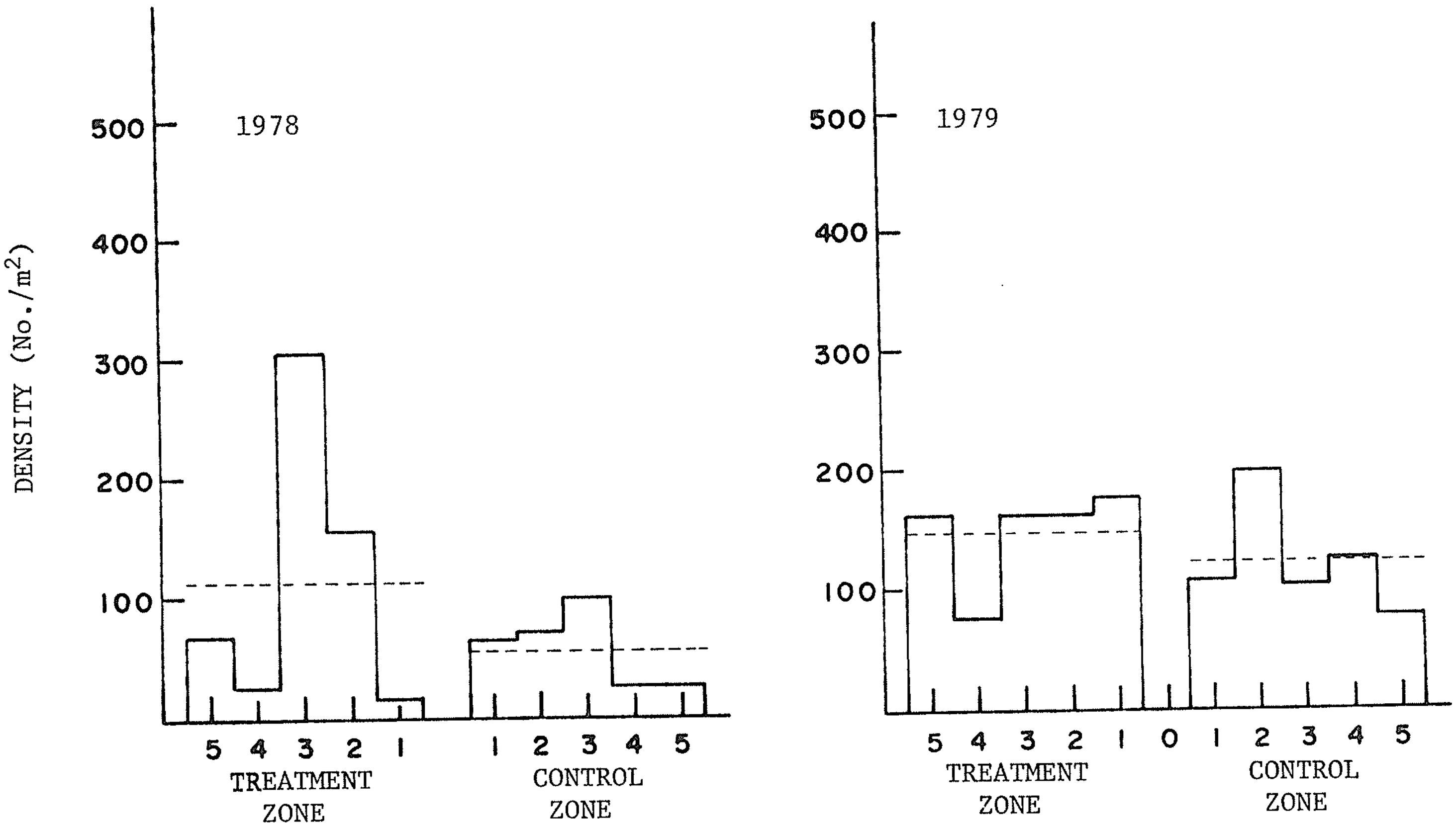


Figure 18. Mean density of Pelecypoda at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

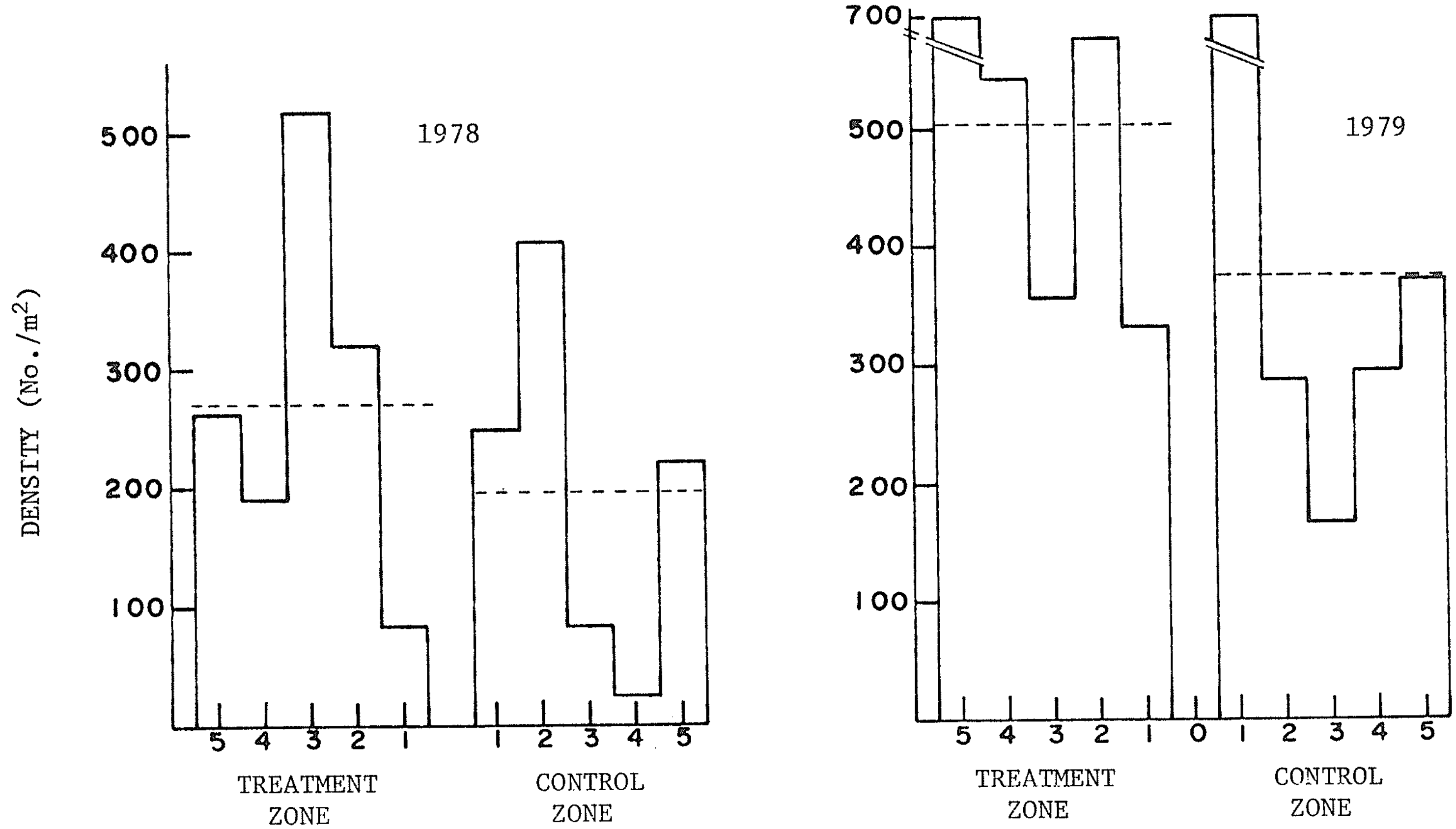


Figure 19. Mean density of Oligochaeta at each transect in 1978 and 1979. Broken lines indicate mean density in each zone.

total standing crops between sampling months in 1978 (Table 15) and again in 1979 (Table 16). Temporal changes in invertebrate standing crop are to be expected because of their life cycle patterns (Hughes, 1978; Hynes, 1971). There was no significant difference in standing crop between sampling depths in 1978 (Table 15) or 1979 (Table 16).

There was a significant ($P<0.01$) difference in standing crop between sampling transects in 1978 (Table 15), but there was no significant difference in standing crops between T- and C-zones. Significant differences in standing crop occurred among the T-zone transects ($P<0.05$), and those of C-zone ($P<0.01$). Since the variations among transects were large (Fig. 20), it was difficult to ascribe low standing crop at any T-zone transect to WNRE effluents and there was no statistical evidence to suggest that WNRE effluents have disturbed the standing crop in T-zone. However, T1 had the lowest standing crop in the T-zone (1300 org/m^2 versus 1700 - 2800 org./m^2, Fig. 20), which indicates that the standing crop immediately downstream from WNRE has been disturbed.

In contrast to 1978, the T-zone standing crop in 1979 was significantly lower ($P<0.05$) than the C-zone crop (Table 16). The largest decreases from 1978 to 1979 occurred at T2, T3 and T4 (Fig. 21), where organic coolant was present. Standing crop at T1 remained low.

Organic coolant released from WNRE may have caused the low standing crop at T1 in 1978 and in 1979 and in T-zone generally in 1979.

(K) Diversity

Diversity ($\bar{d}$) values were calculated only for the 1978 data (Chapter V. 4). Values of $\bar{d}$ varied significantly ($P<0.01$) between sampling months (Table 17). Mean monthly $\bar{d}$ values were high in June

Table 15. Analysis of variance for differences between macroinvertebrate total standing crops at sampling transects, depths, and periods, in the vicinity of the Whiteshell Nuclear Research Establishment in 1978. T = treatment; C = control; *P = 0.05; **P = 0.01. Orthogonal contrasts are in parenthesis.

Source of variance	Degrees of freedom	Sum of squares	Mean square	F-statistic
Main effects				
Transect (T5,T4,T3,T2,T1,C1,C2,C3,C4,C5)	9	10.22	1.14	4.88**
(Control vs treatment)	(1)	(0.81)	(0.81)	(3.48)
(C1 vs C2 vs C3 vs C4 vs C5)	(4)	(5.80)	(1.45)	(6.22)**
(T1 vs T2 vs T3 vs T4 vs T5)	(4)	(3.61)	(0.90)	(3.86)*
Period (June, July, Aug., Sept.)	3	9.45	3.15	13.52**
Depth (1m, 2m)	1	0.57	0.57	2.45
Interaction effects				
Transect x period	27	7.86	0.29	1.25
Transect x depth	9	1.87	0.21	0.89
Period x depth	3	0.86	0.29	1.23
Experimental error	27	6.29	0.23	
Total	79	37.12		

Table 16. Analysis of variance for differences between macroinvertebrate total standing crops at sampling transects, depths, and periods, in the vicinity of the Whiteshell Nuclear Research Establishment in 1979. T = treatment; C = control; *P = 0.05; **P = 0.01. Orthogonal contrasts are in parenthesis.

Source of variance	Degrees of freedom	Sum of squares	Mean square	F-statistic
Main effects				
Transect	9	7.42	0.82	3.57**
(Control vs treatment)	(1)	(1.14)	(1.14)	(4.96)*
(C1 vs C2 vs C3 vs C4 vs C5)	(4)	(3.90)	(0.97)	(4.22)
(T1 vs T2 vs T3 vs T4 vs T5)	(4)	(2.38)	(0.60)	(2.59)
Period (June, July, Aug., Sept.)	3	16.55	5.52	24.00**
Depth (1m, 2m)	1	0.15	0.15	0.65
Interaction effects				
Transect x period	27	5.84	0.22	0.96
Transect x depth	9	1.69	0.19	0.83
Period x depth	3	2.94	0.98	4.26*
Experimental error	27	6.20	0.23	
Total	79	40.79		

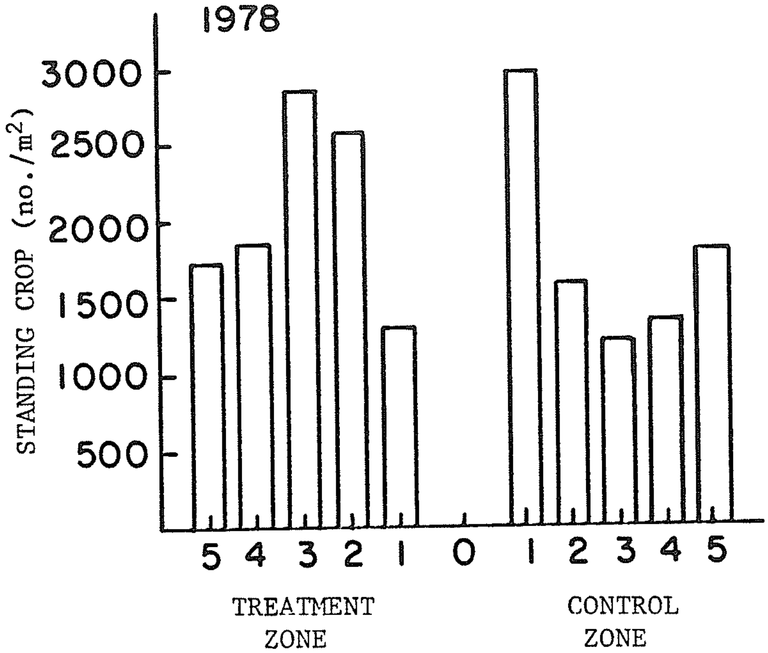


Figure 20. Mean standing crop (total no. of invertebrates/m^2) at each transect in 1978.

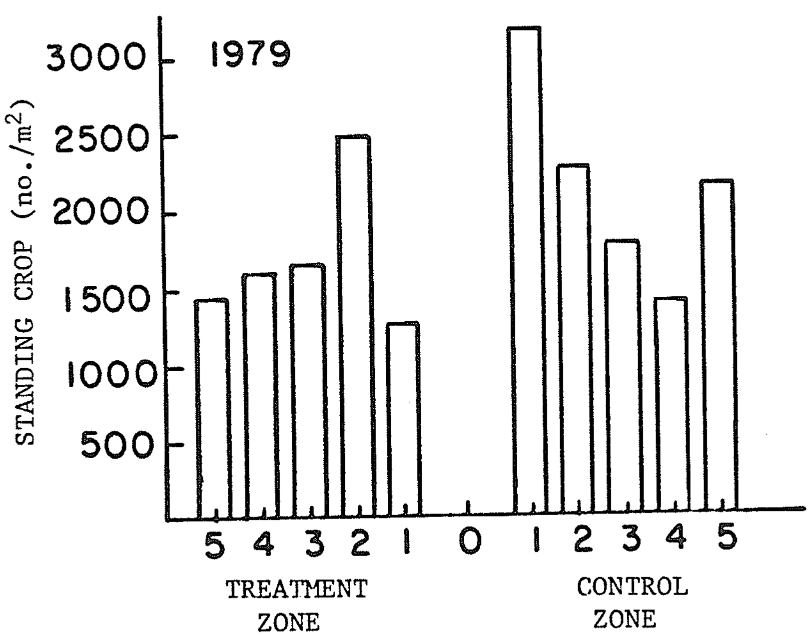


Figure 21. Mean standing crop (total no. of invertebrates/m^2) at each transect in 1979.

Table 17. Analysis of variance for differences between Shannon-Weaver diversity index values calculated for macroinvertebrate communities in the vicinity of the Whiteshell Nuclear Research Establishment in 1978. T = treatment transect; C = control transect; **P = 0.01.

Source of variance	Degrees of freedom	Sum of squares	Mean square	F-statistic
Transect (T5,T4,T3,T2,T1,C1,C2,C3,C4,C5)	9	0.71	0.08	1.33
Period (June, July, Aug., Sept.)	3	1.69	0.56	9.33**
Experimental error	27	1.66	0.06	
Total	39	4.05		

(2.09) and July (2.37), and lower in August (1.85) and September (1.89) (Table 18). Seasonal changes in $\bar{d}$ values are due to life history patterns of the invertebrates (Mackay and Kalff, 1968; Hughes, 1978). Increased diversity from June to July was caused by increased numbers of genera (mostly Chironomidae) (Table 19). Fewer genera were collected in August because of emergence (*Heptagenia* spp., *Cloeon* spp., *Probezzia* spp., *Chrysops* spp., *Ablabesmyia* spp., *Procladius* spp., *Hydroptila* spp., *Agraylea* spp., *Polycentropus* spp.) and overall diversity decreased. The number of genera found in September increased (Table 19) as a result of recently hatched eggs, but $\bar{d}$ increased only slightly because of the decline in evenness in the benthic community (Pielou, 1969) caused by the numerical domination of a few taxa such as *H. limbata* and *H. azteca*.

Mean $\bar{d}$ for C-zone was 2.05 (range: 1.74-2.24) (Fig. 22), and for T-zone was 2.03 (range: 1.95-2.12). There were no significant differences ($P<0.05$) in $\bar{d}$ among sampling transects (Table 17), or between T- and C-zones. No reduction in the value of $\bar{d}$ at T1 was evident. There is no indication from diversity indices that WNRE effluents have deleteriously affected the benthic community.

Use of the Shannon-Weaver diversity index in this study should be qualified. Generic rather than specific level data were used. Although many workers have shown that $\bar{d}$ values calculated using generic identications can reveal significant pollution effects (Hughes, 1979; Kaeslar and Herricks, 1979; Wilhm, 1972), others contend that only calculations using species can satisfactorily reveal an effect on community structure (May, 1976). The use of genus-level data ignores the possibility that a genus may be comprised of many species. The individuals of each

Table 18. Shannon-Weaver diversity index values for macroinvertebrate communities at transects upstream (control) and downstream (treatment) from the Whiteshell Nuclear Research Establishment in 1978. Each value was calculated from 2 pooled bottom samples. Genus level was used.

	Treatment transects						Control transects						Total
	5	4	3	2	1	$\bar{x}$	1	2	3	4	5	$\bar{x}$	$\bar{x}$
June	1.90	1.97	1.69	2.09	2.23	1.98	2.42	1.79	2.06	2.26	2.48	2.20	2.09
July	2.70	2.61	2.40	2.55	2.26	2.50	2.06	2.40	2.04	2.42	2.24	2.23	2.37
Aug.	1.65	1.89	1.58	1.78	1.92	1.76	1.75	2.15	1.56	2.19	2.04	1.94	1.85
Sept.	2.03	1.82	2.18	2.04	1.39	1.89	1.95	2.15	1.29	2.10	1.92	1.88	1.89
$\bar{x}$	2.07	2.07	1.96	2.12	1.95	2.03	2.05	2.12	1.74	2.24	2.17	2.06	2.05

Table 19. Number of taxa at transects upstream (control) and downstream (treatment) from the Whiteshell Nuclear Research Establishment in 1978. Each value represents the number of genera in two pooled samples.

	Treatment transects						Control transects						Total
	5	4	3	1	1	$\bar{x}$	1	2	3	4	5	$\bar{x}$	$\bar{x}$
June	14	17	14	16	15	15.2	18	10	11	15	21	15.0	15.1
July	23	25	25	29	18	24.0	20	21	20	18	19	19.6	21.8
Aug.	8	13	10	12	11	10.8	18	10	8	12	17	13.0	11.9
Sept.	17	14	21	15	18	17.0	19	15	11	14	11	14.0	15.5
$\bar{x}$	15.5	17.3	17.5	18	15.5	16.8	18.8	14	12.5	14.8	17	15.4	16.1

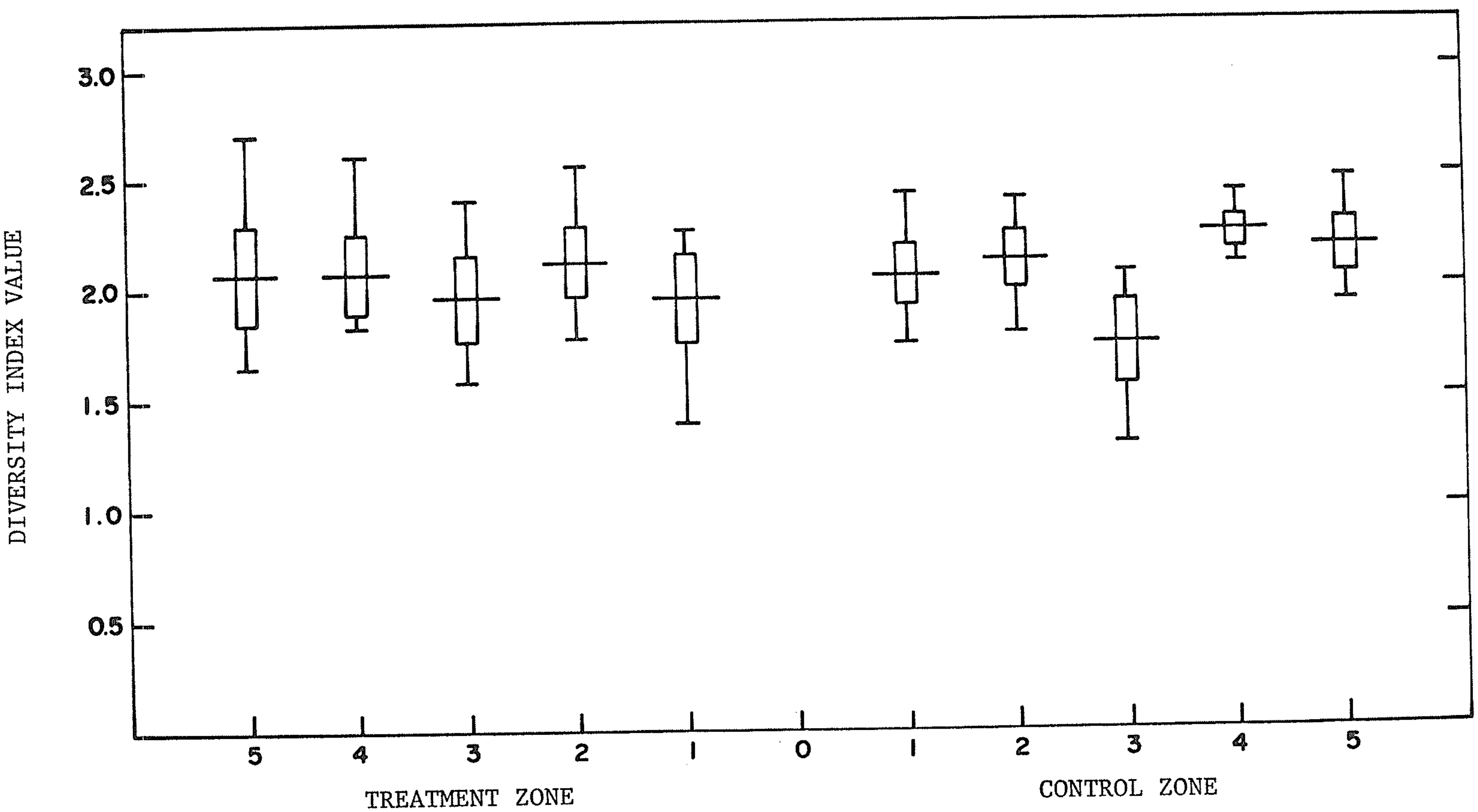


Figure 22. Shannon-Weaver diversity index values for macroinvertebrates at transects upstream (control) and downstream (treatment) of the Whiteshell Nuclear Research Establishment (WNRE). The horizontal line = mean; vertical line = range; bar = one standard error on each side of the mean (n = 4, one value for each month).

species may react differently to environmental stress (Resh and Unzicker, 1975) and may comprise different proportions of the benthic community. Thus the use of species-level rather than generic-level data may yield different $\bar{d}$ values resulting in different conclusions concerning environmental impact.

(L) Comparison with the 1967 study

The invertebrate data from the 1967 study (Ireland, 1968; Ireland *et al.*, 1973) and the present study could not be compared quantitatively. A significantly larger number of invertebrates were removed from the sediment samples during the present study relative to the former study (800 invertebrates/m^2 in 1967 versus 1900/m^2 in 1978 and 1979). The use of a 2x microscope in this study enabled small macroinvertebrates (e.g., early instar insects) to be collected which would otherwise be missed by the unaided eye.

These data, however, were compared to determine whether any gross changes had occurred in the benthic community. There were few changes in the composition of the benthic fauna from 1967 to 1978-79. Benthic communities were numerically dominated by *Hexagenia* spp., chironomids, gastropods, pelecypods, and oligochaetes (Table 20). Odonata, Trichoptera, Megaloptera, and Plecoptera comprised small percentages of the total number of invertebrates collected. There was, however, a large increase in the abundance of *H. azteca* from 1967 to 1978-79 (Table 20). *Elmid* spp. were replaced by *Dubiraphia* spp. These changes were observed in treatment and control zones in 1978-79 and, hence, could not be attributed to WNRE effluents. Acknowledged clean water organisms (e.g.,

Table 20. Major taxa collected downstream from the Whiteshell Nuclear Research Establishment in 1967 (Ireland, 1968), 1978 and 1979. Numbers represent the percent of total number of organisms collected during each year.

	1967	1978	1979
Hexagenia spp.	13.9	24.7	22.2
Ephemera simulans	.8	1.0	.2
Caenis spp.	1.0	.5	.6
Sialis spp.	.1	.4	.3
Chironomidae	26.1	25.5	13.1
Ceratopogonidae	4.6	1.1	1.5
Dubiraphia spp.	0	1.1	1.0
Elmus spp.	5.0	0	0
Hyalella azteca	.1	14.3	8.6
Pelecypoda	3.4	6.6	9.2
Gastropoda	4.4	8.1	11.3
Oligochaeta	30.8	12.6	27.0

Hexagenia spp. and E. simulans) which were present in 1967 were not eliminated from the area receiving effluent. Eleven years of exposure to WNRE liquid effluent has not caused any gross qualitative changes in the benthic fauna of the Winnipeg River.

(M) Summary and conclusions

Similar taxa were collected in T- and C-zones. Burrowers and other invertebrates associated with slow, soft-substratumed rivers dominated the benthic fauna.

Densities of H. limbata, Dubiraphia spp., Chironomidae, Procladius spp., Tanytarsus spp., Paratanytarsus spp., Cryptochironomus spp., Ablabesmyia spp., Ceratopogonidae, Gastropoda, and total standing crops, were low at T1 compared to the other T- and C-zone transects in 1978. Densities of these taxa and total standing crop remained comparatively low at T1 in 1979. Total standing crops and the densities of H. limbata, E. simulans, Dubiraphis spp., and Ceratopogonidae also declined at other T-zone stations (T2, T3, T4, T5) in 1979. Consistently low invertebrate densities at T1 throughout the study period indicated that WNRE effluents have detrimentally affected the benthos in the immediate vicinity of the outfall. The benthos may have also been affected further downstream in 1979. However, the greatest impact occurred close to the outfall, where more taxa were affected and declines in density were most pronounced.

There was no apparent change in densities of H. azteca, Caenis spp., and Pelecypoda between 1978 and 1979. Taxa common in C-zone were also present in T-zone. Pollution sensitive species, such as H. limbata and E. simulans, which probably would not be found in highly contaminated habitats, were not eliminated at any T-zone transect. There was no decrease in generic diversity, as measured by $\bar{d}$, of the benthic community in those regions of the river-bed receiving WNRE effluent. These results suggest any impact on the benthic fauna attributable to WNRE effluent was not severe.

3. Radioactivity of sediment and biota

(A) Beta activity of the sediment

Total beta activities of T-zone sediments were consistently greater than those of C-zone (Fig. 23, 24). The mean beta activity of C-zone sediments was 0.65 Bq/g in 1978 and 0.61 Bq/g in 1979, with little variation amongst sampling locations and sampling times. The highest and most variable beta activities occurred in T1 sediments where activities ranged from 1.7 to 116 Bq/g (i.e., 3-185 times greater than the average level of total activity of C-zone sediments). Beta activities of the remaining sediments in T-zone ranged from 1.2-9.5 Bq/g, 2-15 times greater than C-zone sediments. Sediments sampled at T3, T4, and T5 had higher beta activities than those sampled at T2. Possibly, the more sheltered location of the first three sampling sites allowed water-borne radionuclides to settle out at a greater rate.

Highest beta activities in T-zone sediments were in those sampled during 1979 (Fig. 24) when river discharge was low. Temporal variation in beta activity may, therefore, be related to river discharge. Sedimentation of particulate radionuclides such as Cs-137 increases with lower current speed so the specific activity of the sediment increases (Smedile and Queirazza, 1976).

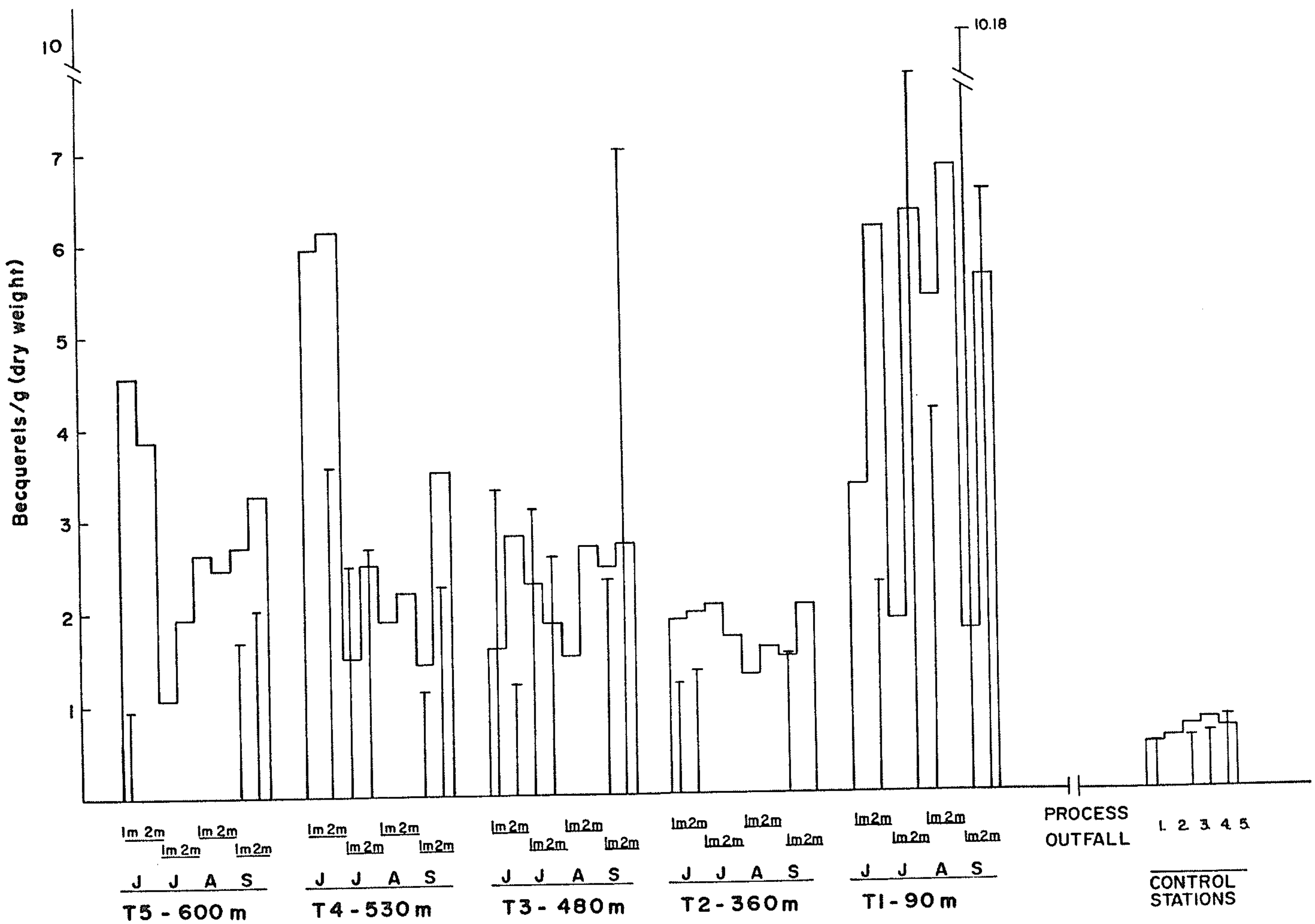


Figure 23. Total beta activity of sediment and Hexagenia limbata nymphs in 1978. Histograms show beta activity of the sediment. Vertical lines show total beta activity of the nymphs. Values for sediment and nymphs in the control zone are yearly means. Distance of treatment (T-) transects downstream of the Whiteshell Nuclear Research Establishment effluent outfall are given. J = June; J = July; A = August; S = September.

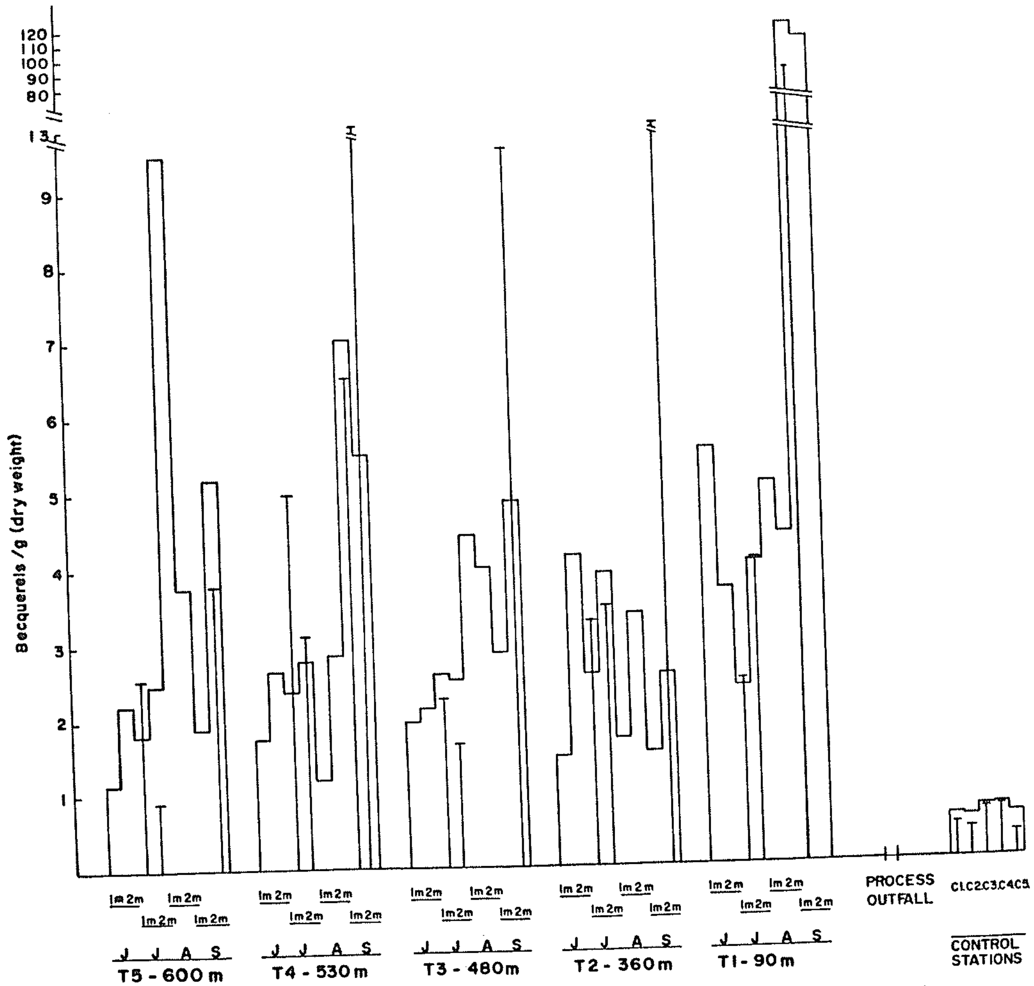


Figure 24. Total beta activity of sediment and Hexagenia limbata nymphs in 1979. Histograms show beta activity of the sediment. Vertical lines show beta activity of the nymphs. Values for sediment and nymphs in the control zone are yearly means. Distance of treatment (T-) transects downstream of the Whiteshell Nuclear Research Establishment effluent outfall are given. J = June; J = July; A = August; S = September.

(B) Radioactivity of the biota

a. Radionuclides present in the biota

Cs-137 was the dominant radionuclide present in *H. limbata*. Trace amounts of Ce-144 and Zr-90 + Nb-90 were also detected. Cs-137 released from WNRE was deposited in greater concentrations in T- than C-zones and was the most abundant radionuclide available to those invertebrates which feed on bottom material.

b. Radioactivity of *Hexagenia limbata*

The concentrations of beta-emitting radionuclides in nymphs of *H. limbata* were greater in T-zone, 0.9-88.0 Bq/g, than in C-zone, 0.3-1.25 Bq/g (Fig. 23, 24). Beta activities at T1 were considerably higher than those of nymphs sampled further downstream.

The beta activity of *H. limbata* and the sediment are correlated ($r = 0.80$; $p<0.05$) (Figs. 23, 24), suggesting that *H. limbata* nymphs accumulate most of their radionuclide body burden from the sediment on which they feed.

c. Accumulation of beta emitting radionuclides by *Hexagenia limbata*

Concentration factor, $\frac{\text{Bq/g wet wt. organism}}{\text{Bq/g water}}$ describes the relationship between the concentration of radionuclides in the organism and its environment when water is the primary source of uptake (Ophel *et al*., 1971; Shure and Gottschalk, 1976). The purpose of this ratio is to determine whether radionuclides are bioaccumulating. This is an important consideration when assessing the radiation dose to benthos, because it is internally deposited radionuclides which make the largest contribution to the organism's total dose (Blaylock, 1972).

When sediment is the primary source of ingested radionuclides, however, it is suggested that a more appropriate ratio would be the Concentration Factor Benthos (CFB). This ratio describes the relationship between the concentration of radionuclides in the organism and the sediment in which it lives:

$$CFB = \frac{Bq/g \text{ dry wt. organism}}{Bq/g \text{ dry wt. sediment}}$$

The mean CFB for H. limbata was 1.15±0.13 (Appendix D). A CFB close to one indicates that the concentration of beta-emitting radionuclides is in equilibrium with the concentration in the sediment, and that no bioaccumulation occurred.

d. Radioactivity of other invertebrates

Total beta activity also was counted in other organisms (Appendix D). Beta activity for all specimens was greater in the T-zone than in the C-zone and beta activity of organisms at T1 was greater than for other T-zone transects. Beta activity of the organisms and the sediment were positively correlated.

The predators, O. colubrinus and Sialis spp., had the lowest total beta activities. The total beta body burden of these invertebrates is probably obtained from their prey (King, 1964; Rice and Baptist, 1974). Since the beta activity of the predators was less than that of the non-predaceous forms, it appears that there is no increase in the concentration of beta-emitting radionuclides between trophic levels. These results agree with Guthrie (1970), Reichle et al. (1970), and Wrenn et al. (1970). The CFB does not apply to these organisms because they do not accumulate radionuclides from the sediment.

The other invertebrates probably obtain their internally deposited radionuclides from ingested sediment, and had higher beta activities than the predators. Chironomids (CFB = 0.73±.16), Chrysops spp. (CFB = 0.49±0.31), pelecypods (CFB = 0.79±.07), gastropods (CFB = 0.57±0.13), and oligochaetes (CFB = 0.78±0.07), concentrated radionuclides in their tissues from the sediment by factors less than one (Appendix D). *H. azteca* and *Dubiraphia* spp. had the highest CFB's, 1.67±.19 and 1.89±.47 respectively. Overall, CFB's indicate no concentration of beta emitting nuclides by benthic macroinvertebrates.

e. Radiation dose received by benthos in the vicinity of WNRE

The radiation dose received by benthic macroinvertebrates was estimated to assess whether it was large enough to cause biological damage. A bioaccumulation factor of one was used in the calculation of estimated dose (See Chapter IV. 3.).

The highest maximum dose was 0.001-0.069 rads/day, calculated for benthos at T1 (Fig. 25). Maximum doses ranged between 0.0006-0.0056 rads/day at the other T-zone transects and 0.0002-0.0006 rads/day at the C-zone transects.

Maximum dose rates received by the benthos in the vicinity of the WNRE outfall were compared with dose rates received by invertebrates in other studies where sub-lethal effects such as chromosomal aberrations and behavioral changes have been observed (Blaylock *et al*., 1978; Emery and McShane, 1980; IAEA, 1976; Polikarpov, 1966) (Fig. 25). Some freshwater invertebrates begin to show damage from radiation at doses ranging from 60-1100 rads/day. The highest maximum dose rate received by

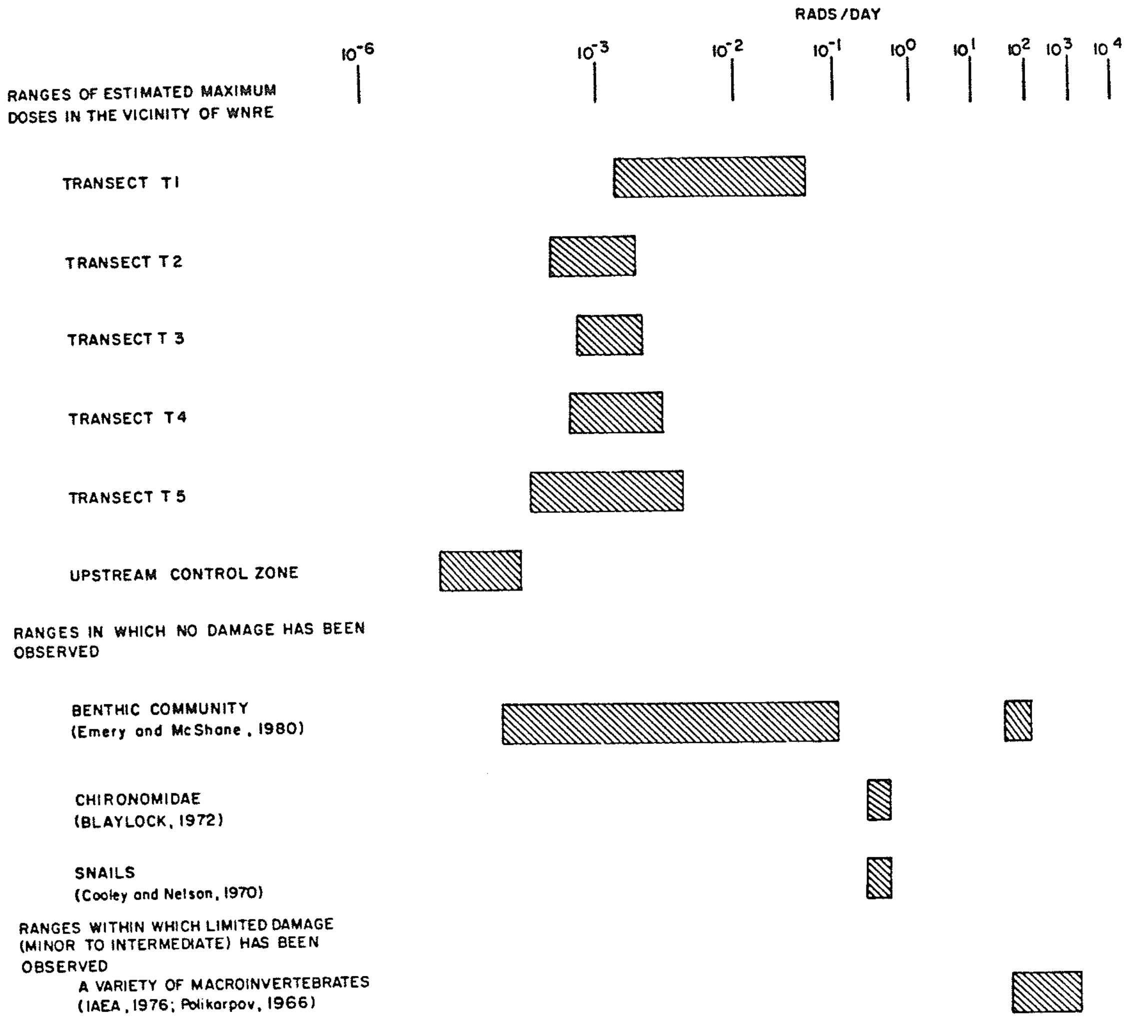


Figure 25. Estimated radiation doses received by benthic macroinvertebrates in the Winnipeg River near the Whiteshell Nuclear Research Establishment (WNRE). The treatment transects (T1-5) are located 90m, 360m, 480m, 530m, and 600m, respectively, downstream of the WNRE outfall. Doses from this study are compared to no-effect and minimal-damage doses reported in other studies.

benthos in the Winnipeg River was 0.069 rads/day, about 3 orders of magnitude lower. The typical maximum dose to the benthos in T-zone (0.0007 and 0.0056 rads/day) was 4 to 5 orders of magnitude lower than the damaging range (Fig. 25). Thus, locations downstream from the WNRE outfall have not been exposed to levels of radiation which detrimentally affect benthic populations.

c. Summary and conclusion

Cs-137 was the most abundant radionuclide present in the benthic habitat downstream from WNRE, and contributed most of the beta activity in the sediment and biota. Highest total beta activities were in the sediments sampled closest to the outfall.

There was a positive correlation between the beta activity of benthic organisms and that of sediment, indicating that benthic invertebrates which feed on bottom sediment accumulate radionuclides from the sediment. Therefore, the ratio 'beta activity of the organism/beta activity of the sediment', affords a more realistic estimate of radionuclide accumulation by benthos, than does the commonly used expression 'activity of the organism/activity of the water'. Beta activity of the benthos achieved a steady state with that of the sediment, and bioaccumulation of beta-emitting radionuclides did not occur.

The maximum dose rate received by the benthos downstream of WNRE (0.0006-0.069 rads/day) was 3 to 5 orders of magnitude less than the range in which sub-lethal damage has been observed. Therefore, it is improbable that radioactive effluent released by WNRE has harmed Winnipeg River benthos.

CHAPTER VI

GENERAL SUMMARY AND CONCLUSIONS

The Whiteshell Nuclear Research Establishment (WNRE) has released the following potential contaminants to the Winnipeg River: (1) process water at above ambient temperature, (2) low-activity liquid radioactive effluent, and (3) organic coolant originating from the primary cooling circuit of the WR-1 nuclear reactor. Communities of Winnipeg River benthic macroinvertebrates upstream and downstream of WNRE were studied during the summer months of 1978 and 1979 to determine if these releases have affected the benthos downstream of WNRE. Species composition, abundance, and diversity of benthic invertebrates downstream were compared with those of upstream. Total beta radioactivities of sediments and organisms in both regions of the river were measured.

Habitat of the benthos in the vicinity of WNRE is characterized by low river current speed, fine-grained clay-sand bottom sediments, and rooted vegetation. Burrowing animals, especially the mayfly nymph *Hexagenia limbata*, and chironomid larvae were predominant. Amphipods (*Hyalella azteca*), coleopterans, gastropods, oligochaetes, and pelecypods also were abundant.

Reduced abundance of some major taxa, most notably *H. limbata* and the chironomids, suggest that some of the benthos downstream of WNRE have been affected by the liquid effluent released. These reductions were apparent 90 m downstream of the outfall in 1978 and 1979, and further downstream (200-600 m) in 1979. Effects were most pronounced in the 90 m area, suggesting that the greatest impact on the benthic population occurred closest to the effluent outfall. However, the high abundances of several taxa, the occurrence of clean water taxa, and no reduction in generic diversity indices, all indicate the impact of the effluent has not been severe.

Heated process water is not responsible for these effects. No increases in ambient water temperatures attributable to WNRE liquid effluent were detected at any of the sampling stations.

Total beta activities of the river-bed sediments downstream of the WNRE outfall ranged from 1.2-116 Bq/gm (dry wt.), or 2-185 times greater than the average level of total beta activity of upstream sediments. Total beta activities of *H. limbata* nymphs downstream of WNRE varied from 0.9-88 Bq/g (dry wt.), or 3-120 times greater than the average activity of nymphs sampled upstream.

Bioaccumulation of beta-emitting radionuclides did not occur in the invertebrates. The ratio 'unit radioactivity of organism/unit radioactivity of sediment' is proposed as a more realistic assessment of nuclide accumulation than the commonly used expression 'unit radioactivity organism/unit radioactivity surrounding water'.

The most abundant gamma emitting nuclide present in the sediments and in the mayfly nymphs was Cs-137. Trace amounts of Ce-144 and Zr-90 + Nb-90 were also detected.

The maximum dose rate of ionizing radiation absorbed by any benthic organism was estimated to be 0.069 rads/day at 90 m downstream from the effluent outfall. At 200 to 600 m downstream from the outfall, the estimated maximum dose-rate was 0.0006-0.0056 rads/day. These dose rates are between three and five orders of magnitude less than the dose-rates at which biological effects due to radiation have been reported (IAEA, 1976). Hence, the observed impact on the benthos in the vicinity of WNRE is not attributed to ionizing radiation.

Largest amounts of organic coolant were present in sediment samples obtained 90 m downstream from the outfall in 1978 and 1979. Organic coolant also occurred further downstream (200-600 m) in 1979. Invertebrate densities were depressed where organic coolant was present. Any adverse impact to the downstream benthos is attributed to the presence of organic coolant on the river bed sediment.

It is recommended that the benthic macroinvertebrate study in the Winnipeg River upstream and downstream of the WNRE effluent outfall be repeated within the next five to ten years. Although the impact on the macroinvertebrate community appeared to be limited to the area immediately downstream of the outfall, it should be determined if this impact zone is increasing or decreasing in size. Since no measurable leaks of organic coolant have occurred since 1979, some change may be expected in the downstream zone.

Both the radiological activity and the organic coolant content of each sediment sample should be measured and correlated with invertebrate data. Organic coolant concentrations in sediments should be determined by chemical analysis (Guthrie and Acres, 1979), rather than by the visual examination used in the present study.

Unless time and funds are available to identify all invertebrates to species, it is recommended that any future study examine only standing crop and several selected species to determine impact on benthos. *Hexagenia limbata* is an ideal test species because it is abundant, easily identified, and sensitive to WNRE effluents.

The use of the Ireland sampler (Ireland, 1968) in future studies of Winnipeg River benthos is not recommended. As the sampler is lowered into the water it creates a compression wave which forces away some organisms at the water-sediment interface. A sampler of known efficiency for estimating macroinvertebrate densities (e.g. Ekman grab) is recommended.

BIBLIOGRAPHY

Aoyama, I. and Y. Inova. 1978. Experimental study on the concentration process of trace element through a food chain from the viewpoint of nutritional ecology. Water Res. 12:831-836.

Armitage, P.D. 1980. The effects of mine drainage and organic enrichment on benthos of the River Nent system, Northern Pennines. Hydrobiologia 74:119.

Ballock, D., C.E. Davies and F.H. Jones. 1976. Biological assessment of water quality in three British rivers: the North Elk (Scotland), the Ivel (England) and the Taff (Wales). Water Pollut. Control, London 75:92-110.

Beirne, B.P. 1963. Collecting, preparing and preserving insects. Can. Dep. Agric. Publ. 932.

Bird, R.D. 1930. Biotic communities of the aspen parkland of central Canada. Ecology 11:356-442.

Blaylock, B.G. 1966. Cytogenetic study of a natural population of Chironomus inhabiting an area contaminated by radioactive waste. Disposal of radioactive wastes into seas, oceans and surface waters (Proc. Symp. Vienna, 1966) IAEA, Vienna, 835.

Blaylock, B.G. 1972. Effects on aquatic biota of radioactive effluents from nuclear power stations. Panel on Radiation Effects on Population Dynamics in Ecosystems. COHF - 721051-2.

Blaylock, B.G. and J.R. Trabalka. 1978. Evaluating the effects of ionizing radiation on aquatic organisms. Adv. Radiat. Biol. 7:103-152.

Blaylock, B.G. and J.P. Witherspoon. 1975. Dose estimations and predictions of radiation effects on aquatic biota resulting from radioactive material released from the nuclear fuel cycle. In: Impacts of nuclear releases into the aquatic environment. (Proc. Symp. Otaniemi, 1975), IAEA, Vienna (1975), 377-93.

Bousfield, E.L. 1958. Freshwater amphid Crustaceans of glaciated North America. Can. Field-Nat., 72:55-113.

Brillouin, L. 1956. Science and information theory. Academic Press, New York, p. 320.

Brinkhurst, R.O. 1965. Observations on the recovery of a British river from gross organic pollution. Hydrobiologia 25:9-51.

Brinkhurst, R.O. and D.G. Cook. 1974. Aquatic earthworms (Annelida: Oligochaeta). In: Pollution ecology of freshwater invertebrates Hart and Fuller (eds.) 1974). Academic Press, New York. pp. 143-156.

Britt, N.W. 1955 (a). Stratification in western Lake Erie in the summer of 1953 and the effects on Hexagenia population. Ecology 35:239-244.

Britt, N.W. 1955 (b). Hexagenia (Ephemeroptera) population recovery in western Lake Erie following the 1953 catastrophe. Ecology 36(3):520-522.

Brown, H.P. 1972. Aquatic Dryopoid beetles (Coleoptera) of the United States. U.S. Environ. Prof. Agency. #18050 ELDO 4/75, U.S.A. p. 83.

Bureau of Radiological Health. 1970. Radiological health handbook - revised edition. U.S. Dep. Health, Educ. Welfare, Maryland, U.S.A.

Burks, B.D. 1953. The mayflies of Ephemeroptera, of Illinois, Ill. Nat. Hist. Surv. Bull. 26:1-216.

Davies, J.F., B.B. Bannatyne, G.S. Barry, and H.R. McCabe. 1962. Manit. Dep. Mines Nat. Resour. p. 190.

Downing, J.A. 1979. Aggregation, transformation, and the design of benthos sampling programs. J. Fish. Res. Board Can. 36:1454-1463.

Cairns, J. Jr. 1967. The use of quality control techniques in management of aquatic ecosystems. Water Resour. Bull. 3, No. 4.

Cairns, J., and K.L. Dickson. 1971. A simple method for the biological assessment of the effects of waste discharges on aquatic bottom-dwelling organisms. J. Water Pollut. Control Fed. 43:755-772.

Carlsson, S. and K. Liden. 1978. Cs-137 and potassium in fish and littoral plants from a humus-rich oligotrophic lake 1961-1976. Oikos 30:126-132.

Clark, P. 1969. Thermal pollution and aquatic life. Scientific America 220:19-27.

Cole, R.A. And J.E. Kelly. 1978. Zoobenthos in thermal discharge to western Lake Erie. J. Water Pollut. Control Fed. 2509-2521.

Comar, C.L. 1965. Movement of fallout radionuclides through the biosphere and man. Annu. Rev. Nucl. Sci. 15:175-205.

Cook, S.E.K. 1976. Quest for an index of community structure sensitive to water pollution. Environ. Pollut. (G.B.). 11:269-288.

Cooley, J.L. and D.J. Nelson. 1970. Effects of chronic irradiation and temperature on populations of aquatic snail Physa heterostrophe. Oak Ridge National Laboratory Report, ORNL - 4612.

Coutant, C.C. 1962. The effect of a heated effluent upon the macroinvertebrate riffle fauna on the Delaware River. Proc. Penn. Acad. Sci. 36:58-71.

Coutant, C.C. 1977. Thermal effects. J. Water Pollut. Control Fed. 63:1369-1425.

Crowe, J.M.E. 1971. Winnipeg River pollution investigations. Manitoba Dep. Mines Nat. Resour. Report 72-12.

Cushing, C.E. 1970. Radiation ecology in freshwater communities. In: Ecological Aspects of the Nuclear Age. (V. Schultz and F.W. Wicker, eds.) U.S. A.E.C. Tech. Inf. TiD-25978.

Dambach, C.A. and J.H. Olive. 1969. Development of biological indices to pollution levels in streams affected by acid mine drainage and oil refinery brine wastes. Natural Resources Inst., Water Resource Center, Ohio State University Resources Project Completion Rpt. A-003.

de March, B.G.E. 1976. Spatial and temporal patterns in macrobenthic stream diversity. J. Fish. Res. Board Can. 33:1261-1270.

Dills, G. and D.T. Rogers. 1974. Macroinvertebrate community structure as an indicator of acid mine pollution. Environ. Pollut. 6:239-261.

Elton, C. 1966. Patterns of animal communities. Wiley, New York.

Emery, R.M. and M.C. McShane. 1980. Nuclear waste ponds and streams on the Hanford Site. An ecological search for radiation effects. Health Phys. 38:738-809.

Environmental Protection Agency (EPA). 1973. Biological field and laboratory methods for measuring the quality of surface waters and effluents. National Research Center, Cincinnati, Ohio, EPA-670/4-73-001.

Federal Department of National Health and Welfare (FDNHW). 1978. Guidelines for Canadian drinking water quality.

Ferguson, V.M. and R.I. Fox. 1978. A comparison of aquatic insects in natural inlets with those in heated effluent from the Oconee Nuclear Station - littoral zone. J. Georgia Entomol. Soc. 13:202-213.

Fetter, S.A. and K. Tsipis. 1981. Catastrophic releases of radioactivity. Sci. Am. 244:41-50.

Flannagan, J.F. 1979. The burrowing mayflies of Lake Winnipeg, Manitoba, Canada. In: Proceedings of the Second Int. Conf. on Ephemeroptera (1975). Krakow, Poland.

Forsyth, M.A. and R.C. Fox. 1976. The influence of substrate and heated effluent from Oconee Nuclear Station on populations of freshwater insects in Lake Keowee - littoral zone. J. Ga. Entomol. Soc. 11:138-150.

Foster, R.F. and J.J. Davis. 1955. The accumulation of radioactive substances in aquatic forms (Columbia River). Proceedings of the International Conference on the Peaceful Uses of Atomic Energy, 13 (p/280):364-367.

Fremling, C.R. 1960. Biology of a large Mayfly, Hexagenia bilineata (Say), of the upper Mississippi River. Iowa State University of Science and Technology. Res. Bull. 482.

Fremling, C.R. 1964. Mayfly distribution indicates water quality on the upper Mississippi River. Science 146:1164-1166.

Fremling, C.R. 1970. Mayfly distribution as a water quality index. U.S. Environmental Prot. Agency. Water Pollut. Control Res. Ser. 16030 DOH. pp. 1-39.

Fremling, C.R. 1975. Acute toxicity of the lampricide 3-trifluoromethyl (TFM) to nymphs of mayflies (Hexagenia sp.). U.S. Dep. Inter. Investigation in Fish Control 58.

Frison, T.H. 1935. The Stoneflies, or Plecoptera, of Illinois. Ill. Nat. Hist. Surv. Bull. 20:281-471.

Fuller, S.L.H. 1974. Clams and Mussels (Molluska: Bivalvia). In: Pollution ecology of freshwater invertebrates. Hart and Fuller (eds.). 1924. Academic Press, New York. pp. 215-273.

Gaufin, A.R. and C.M. Tarzell. 1956. Aquatic macroinvertebrate communities as indicators or organic pollution in Lytle Creek. Sew. Ind. Wastes 28:906-924.

Gerking, S.D., A. Berg and P. Gerard, 1976. Rate of sediment intake by midge larvae (Chironomus plumosus: Diptera) using Cs-134 tracer. In: Radioecology and Energy Resources. (C.E. Cushing, ed.). Edol. Soc. America. Pub. 76-8348.

Good, I.J. 1953. The population frequencies of species and the estimation of population parameters. Biometrika 40:237-64.

Gregory, L.A. and J.S. Loch. 1973. Benthos studies (1971 and 1973) on the Winnipeg River in the vicinity of the Abitibi Manitoba Paper Company, Pine Falls, Manitoba. Tech. Rpt. Series. CENT-73-3.

Gustafson, P.F. 1967. Cesium - 137 in freshwater fish during 1954-65. In: Symposium on Radioecology. Second National Symposium, Ann Arbour, Michigan. (D.J. Nelson and F.C. Evans, Eds.). CONF-670530.

Gustafson, P.F. 1967. Comments on radionuclides in aquatic ecosystems. In: Radioecological Concentration Processes. (B. Aberg and F.P. Hungate, eds.). Pergamon Press, London.

Guthrie, J.E. 1964. Radioactivity in Winnipeg River water. Atomic Energy of Canada Limited Report. AECL-1874.

Guthrie, J.E. 1970. Radioecology of insects in aquatic environments. Manitoba Entomol. 4:33-45.

Guthrie, J.E. 1978. Routine environmental radiation monitoring program at WNRE. Atomic Energy of Canada Limited Report, AECL-5984.

Guthrie, J.E. and O.E. Acres. 1979. Organic coolant in Winnipeg Riverbed sediments. Atomic Energy of Canada Limited Report, AECL-6317.

Guthrie, J.E., O.E. Acres and G.I. Paul. 1974. Environmental surveillance around nuclear installations. I.A.E.A. Vienna, 1974. Report SM-180/3.

Guthrie, J.E. and R.A. Brust. 1969. Elimination rate of radiocesium by aquatic hemiptera Lethocerus americanus. Can. Entomol. 101:856-862.

Guthrie, J.E. and A.Z. Burznyski. 1972. Accumulation of Cesium-137 by Aedes aegypti larvae and the effect of potassium concentration. Can. Entomol. 104:411-416.

Guthrie, J.E. and W.E. Grummitt. 1963. Manual of low activity environmental analyses. Atomic Energy of Canada Limited Report, AECL-1745.

Harman, N.W. 1974. Snails (Mollusca: Gastropoda). In: Pollution ecology of freshwater invertebrates. Hart and Fuller (eds.). 1974. Academic Press, New York. pp. 215-275.

Hart, C.W. and S.L.H. Fuller (eds.). 1974. Pollution ecology of freshwater invertebrates. Academic Press, New York.

Hasanen, E., S. Kolehmainen and J.K. Miettine. 1967. Biological half-time of Cs-137 in three species of freshwater fish: perch, roach and rainbow trout. In: Radioecological Concentration Processes. (B. Aberg and F.P. Hungate, eds.). Pergamon Press, London.

Hewett, C.J. and D.F. Jefferies. 1978. The accumulation of radioactive caesium from food by the plaice (Pleuronectes platessa) and the brown trout (Salmo trutta). J. Fish. Biol. 13:143-153.

Hill, M.O. 1973. Diversity and eveness: a unifying notation and its consequences. Ecology 54:427-32.

Holland, G.A., J.E. Laster, E.D. Neumann and W.E. Eldridge. 1960. Toxic effects of organic and inorganic pollutants on young salmon and trout. Wash. Dept. Fish. Res. Bull. 5:278.

Howell, G.H. and J.B. Gentry. 1975. Effects of thermal effluents from nuclear reactors on species diversity of aquatic insects. In: Thermal Ecology. Georgia Symposium (1975). CONF-730505.

Hubbard, M.D. and W.L. Peters. 1975. Environmental requirements and pollution tolerance of Ephemeroptera. U.S. Environ. Prot. Agency. EPA-600/4-78-061.

Hughes, B.D. 1978. The influence of factors other than pollution on the values of Shannon's diversity index for benthic macro-invertebrates in streams. Water Res. 12:359-364.

Hunt, B.P. 1953. The life history and economic importance of a burrowing mayfly, Hexagenia limbata in Southern Michigan lakes. Michigan Conservation Department. Bull. Inst. Fish. Res. 4.

Hurlbert, S.H. 1971. The nonconcept of species diversity: a critique and alternative parameters. Ecology 52:577-86.

Hynes, H.B.N. 1970. The ecology of running water. Univ. Toronto Press, Toronto, p. 555.

International Atomic Energy Agency (IAEA). 1976. Effects of ionizing radiation on aquatic organisms and ecosystems. IAEC. Rpt. No. 172.

Ireland, R.E. 1968. A survey of benthic insects in the Winnipeg River. M.Sc. Thesis - Dept. of Entomology, University of Manitoba.

Ireland, R.E., J.E. Guthrie and A.G. Robinson. 1973. Occurrence, radioactivity, and diversity, of Winnipeg River benthic organisms in the vicinity of Whiteshell Nuclear Research Establishment. Atomic Energy of Canada Limited Report, AECL-4221.

Johnson, M.G. and R.O. Brinkhurst. 1971. Associations and species diversity in benthic macroinvertebrates of the Bay of Quinte and Lake Ontario. J. Fish. Res. Board Can. 28:1683-1697.

Kaesler, R.L. and H.E. Herricks. 1979. Hierarchical diversity of communities of aquatic insects and fishes. Water Resour. Bull. 15:1117-1125.

Killough, G.G. and L.R. McKay. 1976. A methodology for calculating radiation doses from radioactivity released to the environment. Oak Ridge National Laboratory Report No. 4992.

King, S.F. 1964. Uptake and transfer of Cs-137 by Chlamydomas, Daphnia, and bluegill fingerlings. Ecology 45:852-858.

Kolohmainen, S., E. Hasanen and J.K. Miettinen. 1967. Cs-137 in fish, plankton and plants in Finnish lakes during 1964-65. In: Radioecological Concentration Processes. (B. Aberg and F.P. Hungate, eds.). Pergamon Press, London.

Langham, W.H. and E.C. Anderson. 1959. Cs-137 biospheric contamination from nuclear weapons tests. Health Phys. 2:30-48.

Lenat, D.R. 1979. Effects of power plant operation on the littoral benthos of Belews Lake, North Carolina. In: Energy and Environmental Stress in Aquatic Systems, Selected Papers from a Symposium, Augusta, Ga., November 2-4, 1977. CONF-771114 (1978).

Lerman, A. and H. Taniguichi. 1971. Strontium-90 and Cesium-137 in water and deep sediments of the Great Lakes. In: Third National Symposium on Radioecology. Oak Ridge, Tennessee. May 10-12, 1971.

Lewis, P.A. 1978. Mayfly communities as indicators of water quality. North Amer. Benth. Soc. 26th Ann. Meeting. Winnipeg, Canada.

Loch, J.S. and L.A. Gregory. 1973. A benthic survey of the Red River in vicinity of an oil refinery. Dept. of the Envir. Fish. and Marine Serv. Tech. Rpt. Series No.: CEN/T-73-11.

Mason, W.T. 1973. An introduction to the identification of Chironomid Larvae. U.S. Environ. Prot. Agency. U.S.A. p. 90.

May, R.M. 1975. Patterns of species abundance and diversity. In: Ecology and Evolution of Communities. M.L. Cody and J.M. Diamond, eds.

May, R.M. (ed.). 1976. Theoretical ecology. W.B. Saunders Company, Toronto. p. 317.

McKenzie, D.H., L.D. Kannberg, K.L. Gore, E.M. Arnold and D.G. Watson. 1977. Design and analysis of aquatic monitoring programs at nuclear power plants. Nuclear Regulatory Commission. PNL-2423.

Meritt, R.W. and K.W. Cummins. 1978. An introduction to aquatic insects of North America. Kendal/Hunt Publishing Co. Dubuque, Iowa. p. 441.

Meritt, W.F. 1965. A study of dilution in the Winnipeg River near WNRE. Atomic Energy of Canada Limited Report, AECL-2295.

Nutall, P.M. 1972. The effect of sand deposit on the macroinvertebrate fauna of the River Camel, Cornwall. Freshw. Biol. 2:181-186.

Odum, E.P. 1971. Fundamentals of ecology. W.B. Saunders Company, Toronto, p. 574.

Oliver, D.R., D. McClymont and M.E. Roussel. 1978. A key to some larvae of Chironomidae (Diptera) from the Mackenzie and Porcupine River watersheds. Fisheries and Environment Canada Technical Report. #791.

Ophel, I.L. 1962. The fate of Radiostrontium in a freshwater community. Atomic Energy of Canada Limited Report, AECL-1642.

Ophel, I.L. 1979. Methodology for assessing impacts of radioactivity on aquatic ecosystems. IAEA-AG-134/9.

Ophel, I.L., C.C. Fraser and J.M. Judd. 1971. Strontium concentration factors in biota and bottom sediments of a freshwater lake. Proc. International Symposium on radioecology applied to the protection of man and his environment Rome, Italy. September 7, 1971.

Patrick, R. 1953. Aquatic organisms as aid in solving waste disposal problems. Sewage Ind. Wastes 25:210-219.

Peet, R.K. 1974. The measurement of species diversity. Annu. Rev. Ecol. Syst. 5:285-307.

Pennak, R.W. 1953. Freshwater invertebrates of the United States. Ronald Press Company. New York.

Pielou, E.C. 1966. The measurement of diversity in different types of biological collections. J. Theor. Biol. 13:131-144.

Pielou, E.C. 1969. An introduction to mathematical ecology. Wiley and Sons, New York.

Pielou, E.C. 1975. Ecological diversity. Wiley and Sons. New York.

Polikarpov, G.G. 1966. Radioecology of aquatic organisms. North-Holland Press, New York.

Polikarpov, G.G. 1967. Regularities of uptake and accumulation of radionuclides in aquatic organisms. In: Radioecological Concentration Processes. (B. Aberg and F.P. Hungate, eds.). Pergamon Press, London.

Poole, R.W. 1974. An introduction to quantitative ecology. McGraw-Hill, Inc., New York.

Reichle, D.E., P.B. Dunaway and D.J. Nelson. 1970. Turnover and concentration of radionuclides in food chains. Nucl. Saf. 11(1):43-55.

Resh, V.H. and J.D. Unzicker. 1975. Water quality monitoring and aquatic organisms: the importance of species identification. J. Water Pollut. Control Fed. 47:9-19.

Rice, T.R. and J.P. Baptist. 1974. Ecological effects of radioactive emissions from nuclear power plants. pp. 373-439. In: Human and ecological effects of nuclear power plants. A.L. Sagan (ed.). Thomas Publications, Springfield, U.S.A. p. 536.

Roback, S.S. 1974. Pollution ecology of insects. In: Pollution ecology of freshwater invertebrates. (C.W. Hart and S.L.H. Fuller, eds.). Academic Press, New York. pp. 313-376.

Ruggiero, M.A. and H.C. Merchant. 1979. Water quality, subtrate and distribution of macroinvertebrates in the Patuxent River, Maryland. Hydro. 64:183-189.

Saether, O.A. 1971. Notes on general morphology and terminology of the Chironomidae (Diptera). Can. Entomol. 103:1237-1260.

Saether, O.A. 1977. Taxonomic studies on Chironomidae: Nanocladius, Pseudochironomus, and the Harnischia complex. Fisheries and Environment Canada. Bulletin 1960.

Sagan, L.A. (ed.). 1974. Human and ecological effects of nuclear power plants. Charles C. Thomas Publication, Springfield, U.S.A. p. 536.

Scullion, J. and R.W. Edwards. 1980. The effects of coal industry pollutants on the macroinvertebrate fauna of a small river in the South Wales coalfield. Fresh. Biol. 10:141-162.

Shannon, C.E. and W. Weaver. 1949. The mathematical theory of communication. Univ. Illinois Press, Urbana. p. 117.

Shiomoto, G.T. and B.T. Olson. 1978. Thermal pollution impact upon aquatic life. J. Environmental Health 41:132-139.

Shure, D.J. and M.R. Gottschalk. 1976. Cs-137 dynamics within a reactor effluent stream in South Carolina. In: Radioecology and Energy Resources (C.E. Cushing, ed.). Ecological Society of America.

Simpson, E.H. 1949. Measurement of diversity. Nature 163-68.

Sladecek, V. 1973. System of water quality from the biological point of view. Ergeb. Limnol. 7:1.

Smedile, E. and G. Queirazza. 1976. Uptake of Co-60 and Cs-137 in different components of a river ecosystem connected with discharges of a nuclear power station. In: Radioecology and Energy Resources. (C.E. Cushing, ed.). Ecological Society of America. PUN-76-8348.

Snedecor, G.W. and W.C. Cochran. 1967. Statistical methods. The Iowa State University Press, Ames, Iowa. p. 593.

Stewart, P.L. and J.S. Loch. 1973. A guide for the identification of two subfamilies of Larval Chironomidae: The Chironominae and Tanypodinae found in benthic studies in the Winnipeg River in the vicinity of Pine Falls, Manitoba in 1971 and 1972. Environment Canada Fisheries and Marine Service. #CEN/T-73-12.

Tarzwell, C.M. and A.R. Gaufin. 1967. Some important biological effects of pollution often disregarded in stream surveys. In: Biology of water pollution. W.M. Ingram and K.M. Mackenthun (eds.). pp. 21-31. Fed. Water Poll. Contr. Adm. Cincinnatti, Ohio, CWA-3.

Truelle, M.A. 1958. Effect of waste water from synthetic India rubber manufacture on fish. Sbornik Askoslov, adak. zemedel. ved. Zivocisna vyroba 3, 333-340.

Turner, F.B. 1975. Effects of continuous irradiation on animal populations. Adv. Radiat. Biol. 5:83-144.

Usinger, R.L. (ed.). 1963. Aquatic insects of California. University of California Press. U.S.A. p. 508.

Walker, E.M. 1958. The Odonata of Canada and Alaska. Anisoptera, Vol. 2. University of Toronto Press, Canada. p. 318.

Welch, P.S. 1948. Limnological methods. McGraw-Hill Book Co., Philadelphia, U.S.A.

Wiggins, G.B. 1972. Larvae of the North American caddisfly genera (Trichoptera). University of Toronto Press, Canada. p. 401.

Wilhm, J. 1970. Range of diversity index in benthic macroinvertebrate populations. J. Wat. Pollut. Control Fed. 42:221-224.

Wilhm, J. 1972. Graphic and mathematical analysis of biotic communities in polluted streams. J. Wat. Pollut. Control Fed. 223-252.

Wilhm, J.L. and T.C. Dorris. 1966. Species diversity of benthic macroinvertebrates in a stream receiving domestic and oil refinery effluents. Am. Midl. Nat. 76:427-449.

Wilhm, J.L. and T.C. Dorris. 1968. Biological parameters for water quality criteria. Bioscience 18:477-481.

Williams, L.G. and Q. Pickering. 1961. Direct and food-chain uptake of Cs-137 and strontium 85 in bluegill fingerlings. Ecology 42:205-206.

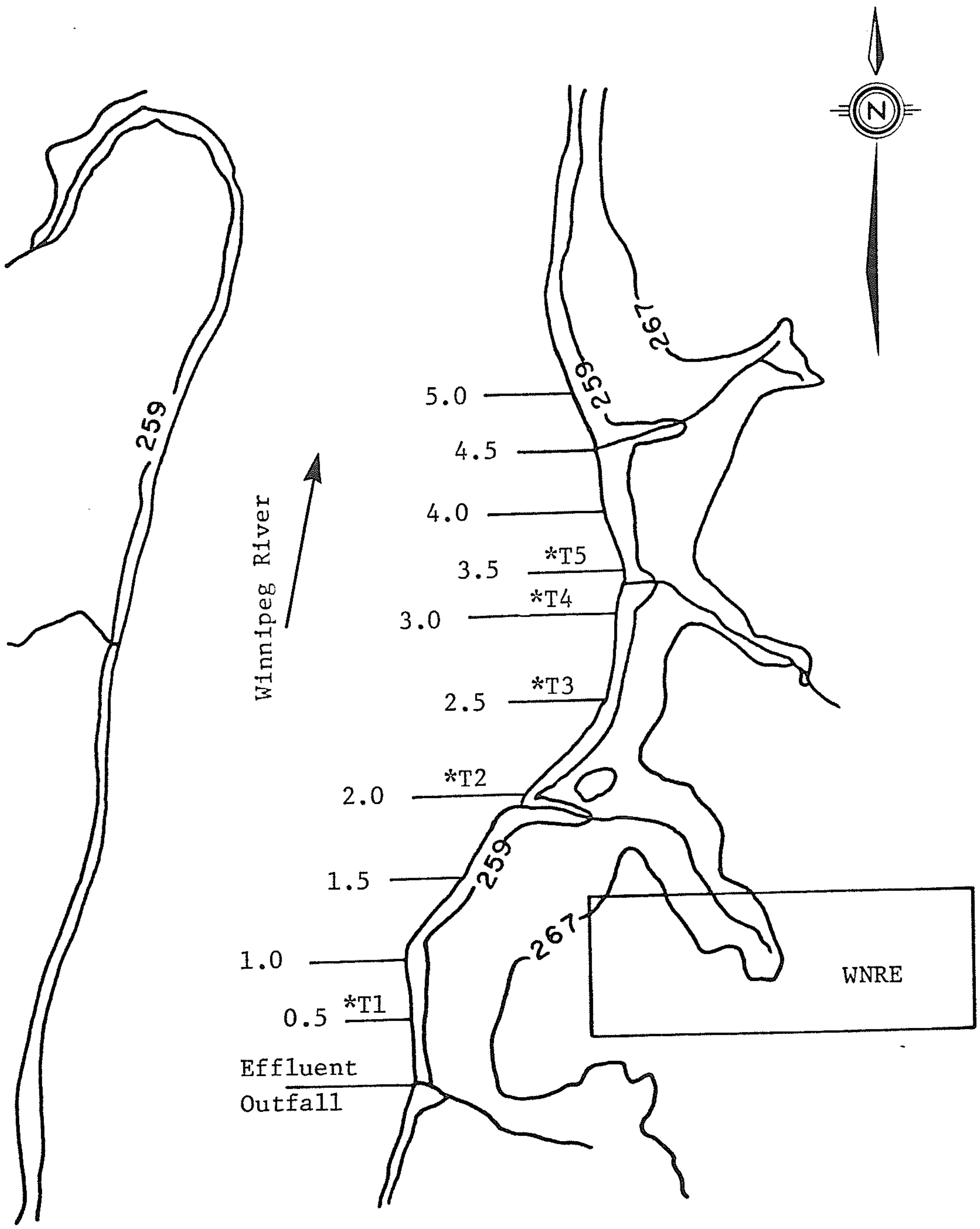
Wrenn, M.E., J.W. Lentsch, M. Eisenbud, G.J. Laver and G.W. Howells. 1971. Radiocesium distribution in water, sediment, and biota in the Hudson River Estuary from 1964 through 1970. In: Third Sym. Rad. Ecology. Oak Ridge, Tennessee. May 10-12, 1971.

Zand, S.M. 1976. Indexes associated with information theory in water quality. J. Water Pollut. Control Fed. 48:2026-2031.

APPENDIX A(1)

Location of treatment zone transects downstream of the Whiteshell Nuclear Research Establishment (WNRE) established by Ireland (1968). Co-ordinates (based on the WNRE co-ordinate system) for each transect are: 0.5 = 67+25N; 1.0 = 69+95N, 1.5 = 72+00N; 2.0 = 76+10N; 2.5 = 80+00N; 3.0 = 81+90N; 3.5 = 85+10N; 4.0 = 86+60N; 4.5 = 89+07N; and 5.0 = 90+90N.

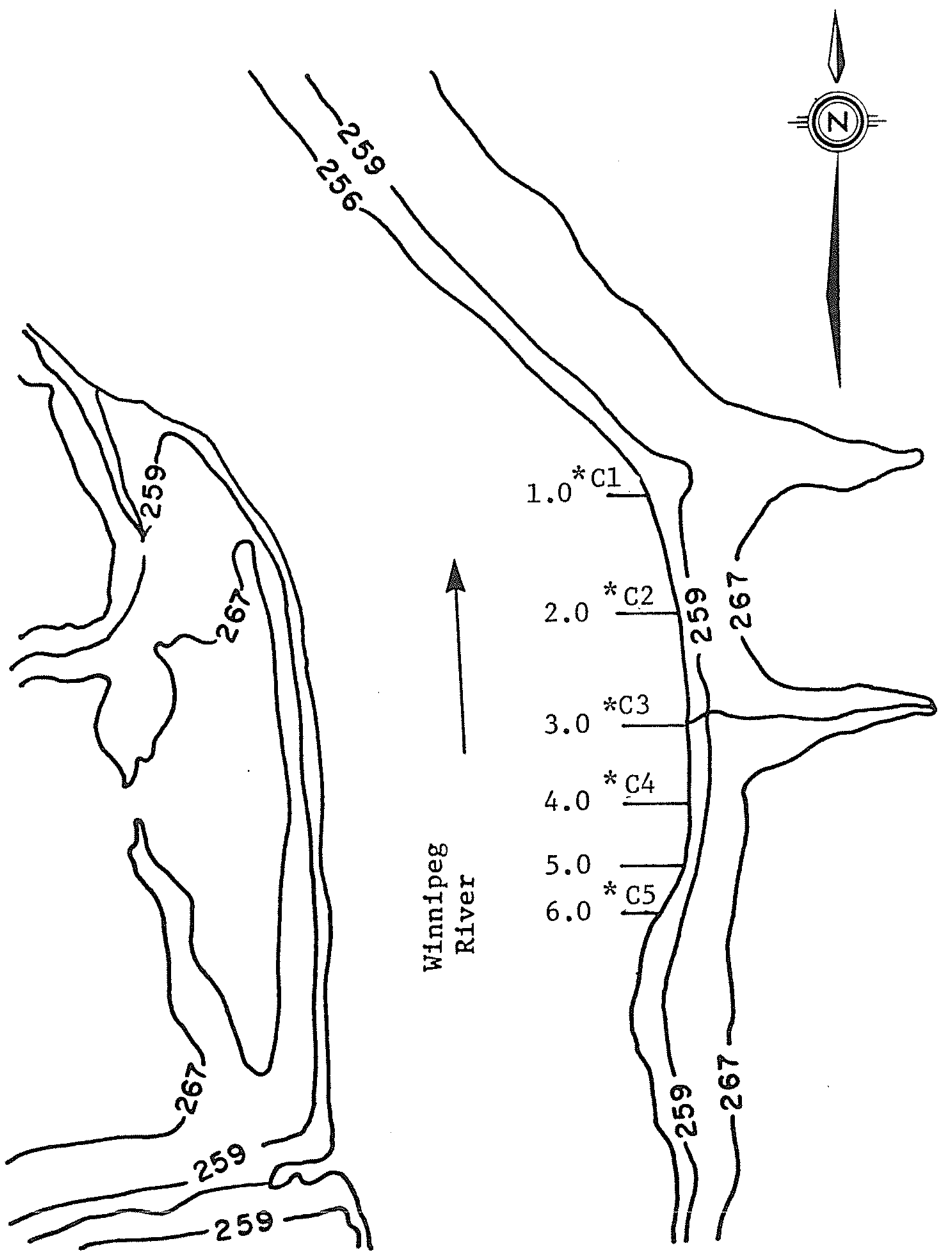
*Transects used for the present study



APPENDIX A(2)

Location of control zone transects upstream of the Whiteshell Nuclear Research Establishment (WNRE) established by Ireland (1968). Co-ordinates (based on the WNRE co-ordinate system) for each transect are: 1.0 = -26+55N; 2.0 = -30+25N; 3.0 = -34+15N; 4.0 = -36+80N, 5.0 = -40+00N, and 6.0 = 42+40N.

*Transects used for the present study



Appendix B(1). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in June, 1978.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	*Hexagenia limbata*	13	40	71	44	6	174	37.7	17	21	19	11	21	89	15.8
	Hexagenia rigida		1				1	.2							
	Ephemera simulans	2	2	5	6	2	17	3.7				1		1	.2
	Caenis spp.	2	2	1	1	1	7	1.5	3	1		4	11	19	3.4
	Ephemerella temporalis														
	Siphlonurus spp.														
	Baetisca spp.														
	Heptagenia spp.														
	Cloeon spp.														
Odonata	*Ophiogomphus colubrinus*					4	4	.9				1		1	.2
	Gomphus spp.														
	Enallagma spp.														
Plecoptera	*Acroneuria* spp.														
Megaloptera	*Sialis* spp.												1	1	.2
Trichoptera	*Agrypnia* spp.														
	Hydroptila spp.														
	Agraylea spp.														
	Pycnopsyche spp.					1	1	.2							
	Phylocentropus spp.														
	Polycentropus spp.														
	Unidentified Trichoptera														
Coleoptera	*Haliplus* spp.														
	Donacia spp.														
	Hydroporus spp.														
	Dubiraphia spp.		1	5	3		9	2.0	4				6	10	1.8
	Gyrinus spp.														
	Galerucella spp.														
	Coptotomus spp.														
Diptera	Ceratopogonidae		8	6	3		17	3.7	13		5	2	3	23	4.1
	Simulium spp.														
	Tipulidae														
	Chrysops spp.														
	Atherix variegata														
Diptera (Chironomidae)	*Chironomus* spp.											1	17	18	3.2
	Pagastiella spp.														
	Cryptochironomus spp.	3	2	3	5	4	17	3.2	9	4	9	1	3	26	4.6
	Demicryptochironomus spp.								1					1	
	Stictochironomus spp.										1			1	.2
	Paralauterborniella spp.			1			1	.2							
	Microtendipes spp.														
	Paratendipes spp.	1					1	.2							
	Dicrotendipes spp.		1				1	.2					2	2	.4
	Glyptotendipes spp.	1	3			4	8	1.7					1	1	.2
	Polypedilum spp.	3	1	1	4		9	2.0	5	6				11	
	Endochironomus spp.	12	1	1	3	4	21	4.6		4	2	1	6	13	2.0
	Phaenospectra spp.														
	Stempellina spp.												1	1	.2
	Paratanytarsus spp.	1			2		3	.7	7	7	3	2	6	25	4.4
	Tanytarsus spp.				1	1	2	.4	1			4		5	.9
	Brillia spp.														
	Cricotopus spp.												2	2	.4
	Orthocladiinae #1												1	1	.2
	Orthocladiinae #2				1		1	.2	1				1	2	.4
	Orthocladiinae #3														
	Orthocladiinae #4								13					13	2.3
	Orthocladiinae #5								1					1	.2
	Ablabesmyia spp.	2	1	1			4	.9					2	2	.4
	Procladius spp.		8	5			13	2.9	4	1		2	26	33	5.9
	Conchapelopia spp.														
Acarina	Acarina								1					1	.2
Crustacea	Cladocera														
	Hyalella azteca	6	1		10	21	38	8.2	32		1	14	49	96	17.1
	Anostraca					1	1	.2							
	Decapoda														
	Ostracoda											1		1	.2
Mollusca	Pelecypoda		2	6	5	1	14	3.3	1	11	14	2	5	33	5.9
	Gastropoda		2	7	4	1	14	3.3	15	7	5	4	10	41	7.3
Annelida	Oligochaeta	35	6	19	17	2	79	17.1	16	43	9		17	85	15.1
	Hirudinea														
Nematoda	Nematoda	1					1	.2			3			3	.5
	Others (unidentified)	1			1	1	3	.7							
	Total numbers	83	82	132	110	54	461	100%	144	105	71	51	191	562	100%
	Number of taxa	14	17	14	16	15	26		18	10	11	15	21	32	

Appendix B(2). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in July, 1978.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	*Hexagenia limbata*	26	79	63	40	13	221	22.9	3	6	13	20	16	58	8.4
	Hexagenia rigida														
	Ephemera simulans														
	Caenis spp.	1			3		4	.4	1		3	2	5	11	1.6
	Ephemerella temporalis					1	1	.1							
	Siphlonurus spp.														
	Baetisca spp.														
	Heptagenia spp.		1				1	.1							
	Cloeon spp.			2			2	.2							
Odonata	*Ophiogomphus colubrinus*				1		1	.1							
	Gomphus spp.														
	Enallagma spp.														
Plecoptera	*Acroneuria* spp.														
Megaloptera	*Sialis* spp.	2	4		1	1	8	.8			1		1	2	.3
Trichoptera	*Agrypnia* spp.														
	Hydroptila spp.				8		8	.8	3		3			6	.9
	Agraylea spp.			3			3	.8							
	Pycnopsyche spp.		1				1	.1							
	Phylocentropus spp.				1		1	.1							
	Polycentropus spp.	1					1	.1	1					1	.1
	Unidentified Trichoptera														
Coleoptera	*Haliplus* spp.		2				2	.2							
	Donacia spp.														
	Hydroporus spp.														
	Dubiraphia spp.	2	5	4	6		17	1.8	9	11	1			21	3.0
	Gyrinus spp.									1				1	.1
	Galerucella spp.														
	Coptotomus spp.														
Diptera	Ceratopogonidae		4	1	4		9	.9		1				1	.1
	Simulium spp.	2					2	.2							
	Tipulidae														
	Chrysops spp.				1	1	2	.2							
	Atherix variegata												1	1	.1
Diptera (Chironomidae)	*Chironomus* spp.	3		2			5	.5			3	9	2	14	2.0
	Pagastiella spp.			1			1	.1							
	Cryptochironomus spp.	6	4	3	4	1	18	1.9	2		6	2	1	11	1.6
	Demicryptochironomus spp.	1	2	1	1	2	2	.7	2	1		1	3	7	1.0
	Stictochironomus spp.											1		1	.1
	Paralauterborniella spp.	1		1	1		3	.3							
	Microtendipes spp.	10	16	2	6	17	51	5.3	4	1	70	10		85	12.3
	Paratendipes spp.													1	.1
	Dicrotendipes spp.														
	Glyptotendipes spp.		12		1		13	1.3		2		1		3	.4
	Polypedilum spp.		1	1		4	6	.6	1				1	2	.3
	Endochironomus spp.	4	8	31	8	19	70	7.2		1		11	2	14	2.0
	Phaenospectra spp.				1	1	2	.2							
	Stempellina spp.	3	2	1			6	.6	6	7	6		2	21	3.0
	Paratanytarsus spp.	1	3	1	1		7	.7	12	9	1	6	4	32	4.6
	Tanytarsus spp.	11	15	14	25		65	6.7	1	8	2	2	2	15	2.2
	Brillia spp.					1	1	.1							
	Cricotopus spp.														
	Orthocladiinae #1				1		1	.1							
	Orthocladiinae #2		2	1			3	.3				1		1	.1
	Orthocladiinae #3	7	5		1	2	15	1.6			13		2	15	2.2
	Orthocladiinae #4									1				1	.1
	Orthocladiinae #5										1			1	.1
	Ablabesmyia spp.	2	14	14	12		42	4.4	1	1	1	3		6	.9
	Procladius spp.	5	19	4	24	3	55	5.7		6	3	16	2	27	3.9
	Conchapelopia spp.														
Acarina	Acarina				1		1	.1					1	1	.1
Crustacea	Cladocera														
	Hyalella azteca	16	12	37	56	4	125	13.0	82	18	6	14	28	148	21.4
	Anostraca														
	Decapoda										1		1	2	.3
	Ostracoda														
Mollusca	Pelecypoda	2		7	8	1	18	1.9	5	5	3	1		14	2.0
	Gastropoda	2	4	12	67	1	86	8.9	12	29	3	1	4	49	7.1
Annelida	Oligochaeta	8	12	10	16	15	61	6.3	29	37	7	4	22	99	14.3
	Hirudinea			1			1	.1		1				1	.1
Nematoda	Nematoda	4	2	1	4	1	12	1.3	13	5				18	2.6
	Others								1	1				2	.3
	Total numbers	120	233	218	305	88	964	100%	189	152	147	105	100	693	100%
	Number of taxa	23	25	25	29	18	44		20	21	20	18	19	36	

Appendix B(3). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in August, 1978.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	Hexagenia limbata	21	19	21	10	10	81	25.2	2	4	12	12	13	43	12.1
	Hexagenia rigida														
	Ephemera simulans				1	1	2	.6					1	1	.3
	Caenis spp.					1	1	.3		1			1	2	.6
	Ephemerella temporalis														
	Siphlonurus spp.														
	Baetisca spp.														
	Heptagenia spp.														
	Cloeon spp.														
Odonata	Ophiogomphus colubrinus		1			1	2	.6							
	Gomphus spp.														
	Enallagma spp.														
Plecoptera	Acroneuria spp.														
Megaloptera	Sialis spp.	2				1	3	.9							
Trichoptera	Agrypnia spp.											4		4	1.1
	Hydroptila spp.														
	Agraylea spp.														
	Pycnopsyche spp.														
	Phylocentropus spp.			2	2		4	1.2						1	.3
	Polycentropus spp.														
	Unidentified Trichoptera														
Coleoptera	Haliplus spp.														
	Donacia spp.														
	Hydroporus spp.														
	Dubiraphia spp.		2		1	1	4	1.2	8	4			1	13	3.7
	Gyrinus spp.														
	Galerucella spp.														
	Coptotomus spp.														
Diptera	Ceratopogonidae									2		1		3	.8
	Simulium spp.														
	Tipulidae														
	Chrysops spp.														
	Atherix variegata														
Diptera (Chironomidae)	Chironomus spp.														
	Pagastiella spp.														
	Cryptochironomus spp.			1			1	.3	3				1	4	1.1
	Demicryptochironomus spp.			1			1	.3		7	1		4	12	3.4
	Stictochironomus spp.														
	Paralauterborniella spp.								2			4	1	7	2.0
	Microtendipes spp.								4			10	5	19	5.3
	Paratendipes spp.								2					2	.6
	Dicrotendipes spp.			1	1		2	.6							
	Glyptotendipes spp.		1				1	.3				9		9	2.5
	Polypedilum spp.														
	Endochironomus spp.	1				3	4	1.2							
	Phaenospectra spp.		1				1	.3							
	Stempellina spp.				1		1	.3							
	Paratanytarsus spp.	1	2				3	.9	7		1		1	9	2.5
	Tanytarsus spp.		1		1	1	3	.9	3	6			1	10	2.8
	Brillia spp.														
	Cricotopus spp.														
	Orthocladiinae #1														
	Orthocladiinae #2														
	Orthocladiinae #3	8	1	1	1		11	3.4			2	3	1	6	1.7
	Orthocladiinae #4		1				1	.3							
	Orthocladiinae #5		4			2	6	1.9							
	Ablabesmyia spp.					1	1	.3	1			2	2	5	1.4
	Procladius spp.	2					2	.6	2			2	3	7	2.0
	Conchapelopia spp.														
Acarina	Acarina								2					2	.6
Crustacea	Cladocera														
	Hyalella azteca	12	17	18	12	8	67	20.8	84	1		19	30	134	37.6
	Anostraca														
	Decapoda														
	Ostracoda								3					3	.8
Mollusca	Pelecypoda			2	3		5	1.6	1	1	4	2	1	9	2.5
	Gastropoda			12	22		34	10.6	10	8	2		1	21	5.9
Annelida	Oligochaeta	13	8	58	1		20	24.8	6	11	3	2	7	29	8.1
	Hirudinea														
Nematoda	Nematoda														
	Others		1				1	.3	1					1	.3
	Total numbers	60	59	117	56	30	322	100%	142	45	25	70	74	356	100%
	Number of taxa	8	13	10	12	11	26		18	10	8	12	17	25	

Appendix B(4). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in September, 1978.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	Hexagenia limbata	51	49	8	33	19	160	19.7	17	27	48	20	23	135	26.8
	Hexagenia rigida	1				1	2	.2							
	Ephemera simulans								3			15	11	29	5.8
	Caenis spp.														
	Ephemerella temporalis														
	Siphlonurus spp.														
	Baetisca spp.			4		4	8	1.0				1		1	.2
	Heptagenia spp.	1					1	.1							
	Cloeon spp.														
Odonata	Ophiogomphus colubrinus					1	1	.1							
	Gomphus spp.								2					2	.4
	Enallagma spp.														
Plecoptera	Acroneuria spp.														
Megaloptera	Sialis spp.		3	1			4	.5	3	1		1	1	6	1.2
Trichoptera	Agrypnia spp.								1					1	.2
	Hydroptila spp.														
	Agraylea spp.														
	Pycnopsyche spp.					2	2	.2							
	Phylocentropus spp.			1			1	.1	1	2				3	.6
	Polycentropus spp.														
	Unidentified Trichoptera														
Coleoptera	Haliplus spp.			1			1	.1							
	Donacia spp.														
	Hydroporus spp.	3	4	1	6		14	1.7		3		2		5	1.0
	Dubiraphia spp.														
	Gyrinus spp.			4	1		5	.6							
	Galerucella spp.														
	Coptotomus spp.														
Diptera	Ceratopogonidae			1			1	.1	4		1	3	2	10	2.0
	Simulium spp.		1				1	.1							
	Tipulidae			2	2		4	.5	1	3		2		6	1.2
	Chrysops spp.														
	Atherix variegata														
Diptera (Chironomidae)	Chironomus spp.		1		1		2	.2			1	2		3	.6
	Pagastiella spp.									3			1	4	.8
	Cryptochironomus spp.				1		1	.1							
	Demicryptochironomus spp.				1		1	.1	16		4		2	22	4.4
	Stictochironomus spp.	3		1		1	5	.6							
	Paralauterborniella spp.								1					1	.2
	Microtendipes spp.														
	Paratendipes spp.														
	Dicrotendipes spp.					1	1	.1			5	3		8	1.6
	Glyptotendipes spp.	2	2			4	8						2	2	.4
	Polypedilum spp.	1					1	.1	6					6	1.2
	Endochironomus spp.	2	1	13		1	17	2.1							
	Phaenospectra spp.														
	Stempellina spp.										2		6	8	1.6
	Paratanytarsus spp.			1		1	2	.2		2				2	.4
	Tanytarsus spp.					1	1	.1							
	Brillia spp.														
	Cricotopus spp.									2				2	.4
	Orthocladiinae #1														
	Orthocladiinae #2														
	Orthocladiinae #3	2		1			3	.4		1				1	.2
	Orthocladiinae #4			1	1		3	.1		1				1	.2
	Orthocladiinae #5												9	9	1.8
	Ablabesmyia spp.		1		5	2	8	1.0	18	2	2	6		28	5.6
	Procladius spp.	6	12	6	5	1	30	3.7		2				2	.4
	Conchapelopia spp.														
Crustacea	Cladocera	2					2	.1	19					19	3.8
	Hyalella azteca	40	50	43	12	93	238	29.2	114	7		5		126	25.0
	Anostraca														
	Decapoda														
	Ostracoda														
Mollusca	Pelecypoda	14	4	58	22	2	100	12.3	8		3	1		12	2.4
	Gastropoda	10	5	23	28	1	67	8.2	9	9	1		1	20	4.0
Annelida	Oligochaeta	7	15	37	43	3	105	12.9	9	7	1		7	24	4.8
	Hirudinea								1					1	.2
Nematoda	Nematoda	1	1	2	2	2	8	1.0							
	Others (unidentified)	6		1			7	.9	1		1	1		3	.6
	Total numbers	152	149	210	163	140	814	100%	234	72	69	64	65	504	100%
	Number of taxa	17	14	21	15	18	35		19	15	11	14	11	33	

Appendix C(1). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in June, 1979.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	Hexagenia limbata	5	2	1	4	4	16	7.3	15	9	20	2	24	70	21.5
	Hexagenia rigida														
	Ephemera simulans														
	Caenis spp.					1	1	.5	1					1	.3
	Ephemerella temporalis														
	Siphlonurus spp.								1					1	.3
	Baetisca spp.														
	Heptagenia spp.														
	Cloeon spp.														
Odonata	Ophiogomphus colubrinus				1		1	.5							
	Gomphus spp.														
	Enallagma spp.														
Plecoptera	Acroneuria spp.					2	2	.9	1		1			2	.6
Megaloptera	Sialis spp.														
Trichoptera	Agrypnia spp.														
	Hydroptila spp.														
	Agraylea spp.														
	Pycnopsyche spp.														
	Phylocentropus spp.														
	Polycentropus spp.		1				1	.5							
Coleoptera	Haliplus spp.								1					1	.3
	Donacia spp.				2		2	.9	2					2	.6
	Hydroporus spp.														
	Dubiraphia spp.		2				2	.9	6	4				10	3.1
	Gyrinus spp.														
	Galerucella spp.														
	Coptotomus spp.														
Diptera	Ceratopogonidae		3				3	1.4	5	3	4		2	14	4.3
	Simulium spp.														
	Tipulidae														
	Chrysops spp.	2			1		3	1.4		5	4		3	12	3.7
	Atherix variegata														
	Chironomidae	5	18		9	7	39	17.7	45	12	12		13	82	25.7
Crustacea	Cladocera														
	Hyalella azteca	4	18	1	28		51	23.2	16	4	16			36	11.0
	Anostraca														
	Decapoda														
	Ostracoda														
Mollusca	Pelecypoda		5	6	8	2	21	1.5	4	7			4	15	4.6
	Gastropoda	6	8	12	9		35	15.9	17	5		1	2	25	7.7
Annelida	Oligochaeta	3	19	6	7	5	40	18.2	26	5	1		14	46	14.1
	Hirudinea								3					3	.9
Nematoda	Nematoda		2		1		3	1.4	2	2			1	5	1.5
	Others								1					1	.3
	Total	25	78	26	70	21	220	100%	146	56	58	3	63	326	100%
	Number of taxa	6	10	5	10	6	15		16	10	7	2	8	17	

Appendix C(2). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in July, 1979.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	*Hexagenia limbata*	34	40	38	62	24	198	21.4	56	20	42	33	35	186	22.1
	Hexagenia rigida														
	Ephemera simulans	1			2		3	.3							
	Caenis spp.	2	1		3	5	11	1.2	3	1	3	5	2	14	1.7
	Ephemerella temporalis														
	Siphlonurus spp.														
	Baetisca spp.														
	Heptagenia spp.				1	1	2	.2							
	Cloeon spp.														
Odonata	*Ophiogomphus colubrinus*														
	Gomphus spp.														
	Enallagma spp.														
Plecoptera	*Acroneuria* spp.														
Megaloptera	*Sialis* spp.				1		1	.1							
Trichoptera	*Agrypnia* spp.														
	Hydroptila spp.														
	Agraylea spp.														
	Pycnopsyche spp.					1	1	.1			2			2	.2
	Phylocentropus spp.														
	Polycentropus spp.					3	3	.3							
	Hydropsyche spp.					7	7								
Coleoptera	*Haliplus* spp.			2						2				2	.2
	Donacia spp.						2	.2	2					2	.2
	Hydroporus spp.														
	Dubiraphia spp.	2			1	1	4	.4	4	7		5	1	17	2.0
	Gyrinus spp.														
	Galerucella spp.														
	Coptotomus spp.														
Diptera	Ceratopogonidae	5	3	1	3		12	1.3	20	15	4	7	7	53	6.3
	Simulium spp.														
	Tipulidae														
	Chrysops spp.										1		1	2	.2
	Atherix variegata					1	1	.1				1		1	.1
	Chironomidae	18	21	23	37	77	176	19.0	82	43	41	41	37	244	28.9
Crustacea	Cladocera														
	Hyalella azteca	5	2	1	12	26	46	5.0	2					2	.2
	Anostraca														
	Decapoda														
	Ostracoda														
Mollusca	Pelecypoda	16	7	21	14	14	72	7.8	10	5	5	2	1	23	2.7
	Gastropoda	15	16	61	32	7	131	14.1	27	16	3	2	11	59	7.0
Annelida	Oligochaeta	39	74	62	54	21	250	27.0	79	54	10	37	29	209	24.8
	Hirudinea														
Nematoda	Nematoda			1		3	4	.4	4	7	1	1	1	23	2.7
	Others		1			2	3	.3			2	2		4	.5
	Total	137	165	210	222	193	927	100%	289	170	114	136	134	843	100%
	Number of taxa	10	9	9	12	15	19		11	10	11	11	10	16	

Appendix C(3). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in August, 1979.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	*Hexagenia limbata*	21	25	38	32	2	118	20.8	36	30	32	26	18	142	22.7
	Hexagenia rigida														
	Ephemera simulans		1				1	.2							
	Caenis spp.										1	1	1	3	.5
	Ephemerella temporalis														
	Siphlonurus spp.												1	1	.2
	Baetisca spp.														
	Heptagenia spp.				1		1	.2							
	Cloeon spp.														
Odonata	*Ophiogomphus colubrinus*								1					1	.2
	Gomphus spp.														
	Enallagma spp.		1		1		2	.4							
Plecoptera	*Acroneuria* spp.														
Megaloptera	*Sialis* spp.			1	1		2	.4	7	2	2	2	6	19	3.0
Trichoptera	*Agrypnia* spp.														
	Hydroptila spp.														
	Agraylea spp.														
	Pycnopsyche spp.														
	Phylocentropus spp.	2			1		3	.5	4	1			2	7	1.1
	Polycentropus spp.									1				1	.2
	Rhyacophila spp.				2		2	.4							
Coleoptera	*Haliplus* spp.		2	1			3	.5							
	Donacia spp.														
	Hydroporus spp.														
	Dubiraphia spp.		2	2	5	1	10	1.8	4	7	3	6	3	23	3.7
	Gyrinus spp.														
	Galerucella spp.														
	Coptotomus spp.				1		1	.2							
Diptera	Ceratopogonidae	1			7		8	1.4			4		6	10	1.6
	Simulium spp.														
	Tipulidae														
	Chrysops spp.			1	2	5	8	1.4	1		2	5	4	12	1.9
	Atherix variegata														
	Chironomidae	19	4	7	16	7	53	9.3	31	37	21	9	36	134	21.4
Crustacea	Cladocera														
	Hyalella azteca	5	3	7	11		26	4.6	6	16			34	56	8.9
	Anostraca														
	Decapoda														
	Ostracoda														
Mollusca	Pelecypoda	11	4	8	12	8	43	7.6	1	6	11	2	9	29	4.6
	Gastropoda	5	9	51	37	14	116	20.4	26	18	24	4	14	86	13.7
Annelida	Oligochaeta	86	7	8	44	18	163	28.7	33	10	23	7	19	92	14.7
	Hirudinea			2			2	.4	2					2	.3
Nematoda	Nematoda		3		1	1	5	.9	1		1	3	1	6	1.0
	Others				1		1	.2		1		1		2	.3
	Total	150	61	126	173	58	568	100%	153	129	124	66	154	626	100%
	Number of taxa	8	11	11	16	9	20		13	11	11	11	14	18	

Appendix C(4). Invertebrate taxa collected downstream (treatment zone) and upstream (control zone) from the Whiteshell Nuclear Research Establishment in September, 1979.

		Treatment transects							Control transects						
	Taxa	T5	T4	T3	T2	T1	Total	% Total organisms	C1	C2	C3	C4	C5	Total	% Total organisms
Ephemeroptera	*Hexagenia limbata*	20	18	23	21	8	90	18.4	32	57	51	40	98	278	36.9
	Hexagenia rigida														
	Ephemera simulans														
	Caenis spp.					1	1	.2					2	2	.3
	Ephemerella temporalis														
	Siphlonurus spp.														
	Baetisca spp.														
	Heptagenia spp.	1		1	1		3	.6	1					1	.1
	Cloeon spp.														
Odonata	*Ophiogomphus colubrinus*														
	Gomphus spp.														
	Enallagma spp.				1		1	.2	1					1	.1
Plecoptera	*Acroneuria* spp.														
Megaloptera	*Sialis* spp.		2	1			3	.6	4	3		5	4	16	2.1
Trichoptera	*Agrypnia* spp.								2					2	.3
	Hydroptila spp.														
	Agraylea spp.														
	Pycnopsyche spp.														
	Phylocentropus spp.			1	1		2	.4	8			1		9	1.2
	Polycentropus spp.			1			1	.2	1					1	.2
Coleoptera	*Haliplus* spp.	2		1	2		5	1.0	1	3		1		5	.7
	Donacia spp.				4		4								
	Hydroporus spp.														
	Dubiraphia spp.	1		2	4	1	8	.8	2	8	4		4	18	2.4
	Gyrinus spp.														
	Galerucella spp.														
	Coptotomus spp.														
Diptera	Ceratopogonidae		1	1	3	2	7	1.4	1	1	3		2	7	.9
	Simulium spp.														
	Tipulidae														
	Chrysops spp.	2	2	4	2		10	2.0	3		1	1	1	6	.8
	Atherix variegata														
	Chironomidae	5	10	14	8	1	38	7.8	17	3	5	6	9	40	5.3
Crustacea	Cladocera								3					3	.4
	Hyalella azteca	5	3	22	30	1	61	12.5	29	23	2	5		59	7.8
	Anostraca														
	Decapoda	1					1	.2							
	Ostracoda								1	18				19	2.5
Mollusca	Pelecypoda	12	3	4	5	19	43	8.8	11	30	14	9	5	69	9.2
	Gastropoda	9	8	12	12	5	46	9.4	16	44	22	16	13	111	14.7
Annelida	Oligochaeta	42	30	10	34	36	152	31.1	41		6	27	27	101	13.4
	Hirudinea			2	1		3	.6							
Nematoda	Nematoda	1	1	1	5	1	9	1.8	1		1		2	4	.5
	Others								1					1	.1
	Totals	101	78	100	134	75	488	100%	176	190	109	111	167	753	100%
	Number of taxa	12	10	16	16	10	20		20	10	10	10	11	21	

Appendix D. Total beta activities of invertebrates and sediment in the Winnipeg River near the Whiteshell Nuclear Research Establishment in 1978-79. Concentration factors for benthos (CFB; see text for definition) are calculated for each invertebrate sample. J = June; Jy = July; A = August; S = September; P = all months pooled.

Date	Organism	Sampling station	Number of organisms	Total beta activity: Bq./g dry wt. in organisms	Total beta activity: Bq./g dry wt. in sediment	CFB	Mean CFB for each group (± one standard error)
1978-J	*H. limbata*	T1-2m	2	2.25	6.11	.37	$\bar{X}$ = 1.15 ± .13
	"	T2-2m	2	1.19	1.92	.62	
	"	T2-2m	37	1.43	1.92	.74	
	"	T3-1m	15	3.30	1.60	2.06	
	"	T3-2m	40	1.20	2.80	.43	
	"	T4-2m	36	3.57	6.15	.58	
	"	T5-1m	6	.95	4.57	.21	
	"	C1-1m	1	.05	.48	.10	
	"	C3-2m	2	.19	.82	.23	
	"	C4-2m	9	.12	.73	.16	
	"	C5-1m	9	.69	.68	1.01	
1978-Jy	"	T3-1m	13	3.09	2.24	1.38	
	"	T3-2m	50	2.59	1.82	1.42	
	"	T4-1m	32	2.48	1.47	1.69	
	"	T4-2m	47	2.68	2.50	1.07	
1978-S	"	T1-1m	4	10.18	1.76	5.72	
	"	T1-2m	4	6.50	5.58	1.16	
	"	T2-1m	12	1.49	1.45	1.03	
	"	T3-1m	4	2.33	2.42	.96	
	"	T3-2m	1	6.97	2.70	2.58	
	"	T4-1m	7	1.14	1.42	.80	
	"	T4-2m	3	2.24	3.51	.64	
	"	T5-1m	11	1.66	2.71	.61	
	"	T5-2m	5	2.01	3.24	.62	
	"	C1-1m	3	.80	.52	1.53	

Continued ....

Appendix D (Continued)

Date	Organism	Sampling station	Number of organisms	Total beta activity: Bq./g dry wt. in organisms	Total beta activity: Bq./g dry wt. in sediment	CFB	Mean CFB for each group (± one standard error)
	H. limbata	C3-2m	3	.88	.62	1.42	
	"	C4-1m	5	.43	.70	.61	
	"	C4-2m	3	1.25	.71	1.76	
	"	C5-1m	2	1.15	.69	1.66	
	"	C5-2m	10	1.08	.52	2.07	
1979-Jy	"	T1-1m	4	2.45	2.36	1.04	
	"	T1-2m	3	4.09	4.00	1.02	
	"	T2-1m	24	3.23	2.54	1.27	
	"	T2-2m	4	3.08	3.87	.80	
	"	T2-2m	34	3.80	3.87	.98	
	"	T3-1m	2	2.23	2.59	.86	
	"	T3-2m	10	1.65	2.49	.66	
	"	T4-1m	1	4.96	2.38	2.08	
	"	T4-2m	4	3.13	2.78	1.23	
	"	T5-1m	3	2.54	1.80	1.41	
	"	T5-2m	2	.91	2.46	.40	
	"	C1-1m	2	.45	.56	.80	
	"	C1-1m	35	.38	.56	.68	
	"	C3-2m	4	.47	.61	.77	
	"	C4-1m	14	.45	.73	.62	
	"	C4-2m	1	.77	.71	1.08	
	"	C5-2m	21	.34	.56	.61	
1979-A	"	C2-1m	17	.39	.54	.72	
	"	T1-1m	2	87.71	116.72	.75	
	"	T2-2m	17	13.85	2.54	5.45	
	"	T3-2m	6	9.51	4.82	1.97	
	"	T4-1m	10	6.50	7.01	.93	
	"	T4-2m	8	12.96	5.48	2.36	

Continued ....

Appendix D (Continued)

Date	Organism	Sampling station	Number of organisms	Total beta activity: Bq./g dry wt. in organisms	Total beta activity: Bq./g dry wt. in sediment	CFB	Mean CFB for each group (± one standard error)
	H. limbata	T5-2m	11	3.78	5.19	.73	
	"	C1-1m	8	.55	.62	.89	
	"	C1-2m	9	.40	.53	.75	
	"	C3-1m	6	.80	1.00	.80	
	"	C4-2m	5	.80	.66	1.21	
	"	C5-1m	10	.26	.51	.51	
1978-J	O. colubrinus	T1-1m	2	1.15	3.34	.34	$\bar{X}$ = 0.25 ± .04
	"	T1-2m	1	1.39	6.11	.21	
	"	C1-1m	2	.04	.48	.08	
	"	C4-1m	2	.22	.88	.25	
1978-Jy	"	T2-2m	1	.66	1.66	.40	
1978-A	"	T4-1m	1	.60	1.88	.32	
1979-Jy	"	T2-1m	1	.70	2.54	.28	
	"	T4-1m	1	.35	2.38	.15	
1978-P	Dubiraphia spp.	T1+2	17	3.57	3.18($\bar{x}$)	1.12	$\bar{X}$ = 1.89 ± .47
	"	T3	15	3.31	2.22 "	1.49	
	"	T4+5	21	3.77	2.97 "	1.27	
	"	C-all	24	1.79	.63 "	2.84	
1979-P	"	T1+2	21	6.68	18.79 "	.35	
	"	T3+4+5	25	2.35	3.29 "	.71	
	"	C1	34	2.11	.58 "	3.63	
	"	C2	22	2.08	.56 "	3.71	
1978-P	Sialis spp.	T-all	14	1.24	2.90	.43	$\bar{X}$ = 0.22 ± .11
1979-P	"	T-all	15	.78	9.47	.08	
	"	C-all	34	.09	.61	.15	

Continued ....

Appendix D (Continued)

Date	Organism	Sampling station	Number of organisms	Total beta activity: Bq./g dry wt. in organisms	Total beta activity: Bq./g dry wt. in sediment	CFB	Mean CFB for each group (± one standard error)
1979-P	Chironomidae	T1	95	7.08	34.94	.20	$\bar{X} = 0.73 \pm .16$
	"	T2	73	1.63	2.63	.62	
	"	T3+4	110	1.65	3.19	.52	
	"	T5	73	2.78	3.49	.80	
	"	C1	174	.43	.58	.74	
	"	C2	85	.84	.56	1.50	
	"	C3	82	.95	.67	1.42	
	"	C4	62	.10	.68	.15	
	"	C5	103	.37	.58	.64	
1979-P	*Chrysops* spp.	T-all	20	1.67	9.47	.18	$\bar{X} = 0.49 \pm .31$
	"	C-all	20	.49	.61	.80	
1978-P	*H. azteca*	T1-1m	72	6.14	3.07	2.00	$\bar{X} = 1.67 \pm .19$
	"	T1-2m	54	3.91	6.19	.63	
	"	T2	90	2.81	1.72	1.63	
	"	T3	98	2.14	2.22	.96	
	"	T4	97	3.22	3.13	1.03	
	"	C1-1m	136	1.30	.76	1.71	
	"	C1-2m	176	1.67	.54	3.09	
	"	C2+3	67	1.06	.63	1.68	
	"	C4	67	1.29	.75	1.72	
	"	C6	107	.94	.66	1.42	
1979-P	"	T3+4	140	5.56	3.19	1.74	
	"	C-all	101	1.46	.61	2.39	
1978-P	Pelecypoda	T1	10	4.86	4.63	1.05	$\bar{X} = 0.79 \pm .07$
	"	T2	12	2.01	1.72	1.17	

Continued ....

Appendix D (Continued)

Date	Organism	Sampling station	Number of organisms	Total beta activity: Bq./g dry wt. in organisms	Total beta activity: Bq./g dry wt. in sediment	CFB	Mean CFB for each group (± one standard error)
	Pelecypoda	T3+4+5	13	1.92	2.72	.71	
	"	T3+4+5	7	2.09	2.72	.77	
	"	C1+2	22	.52	.55	.95	
	"	C3	19	.63	.68	.97	
	"	C4+5	12	.49	.71	.69	
1979-P	"	T1	16	6.97	34.91	.20	
	"	T2+3+4+5	36	2.82	3.12	.90	
	"	T2+3+4+5	26	1.75	3.12	.56	
	"	C1+2+3	16	.46	.60	.77	
	"	C4+5	9	.47	.63	.75	
1978-Jy	Gastropoda	T2-2m	1	.33	1.66	.20	$\bar{X}$ = 0.57 ± .13
1978-A	"	C2-1m	1	.37	.58	.64	
1979-A	"	T1-1m	1	4.06	5.02	.81	
	"	C1-1m	1	.35	.56	.63	
1978	Oligochaeta	T1	70	4.02	4.63	.87	$\bar{X}$ = 0.78 ± .07
	"	T2	149	1.67	1.72	.97	
	"	T3+4	192	2.71	2.68	1.01	
	"	T5	109	2.49	2.80	.89	
	"	C1+2+3	164	.30	.59	.51	
	"	C4+5	149	.38	.71	.54	
1979-P	"	T1	176	10.24	34.94	.29	
	"	T2	294	2.87	2.63	1.09	
	"	T3+4	270	2.95	3.19	.92	
	"	T5	349	1.71	3.49	.49	
	"	C1	315	.48	.58	.83	
	"	C2+3+4	226	.49	.64	.77	
	"	C5	175	.57	.58	.98	