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Canada

GUIDANCE DOCUMENT
ENVIRONMENTAL EFFECTS ASSESSMENT OF FRESHWATER THERMAL DISCHARGE

Environmental Protection Operations Division – Ontario
Environmental Stewardship Branch
Environment and Climate Change Canada

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LIST OF ACRONYMS, ABBREVIATIONS AND MEASUREMENT UNITS

Acronyms and Abbreviations

1Q10	The minimum 1-day flow that occurs with a return frequency of once in each 10 years.
3-D	Three-dimensional
7Q10	The minimum 7-day average flow that occurs with a return frequency of once in each 10 years
ACCC	Abitibi Consolidated Company of Canada
accl	Acclimation
ANOVA	Analysis of variance
BACI	Before-after-control-impact
BANRP	Bruce A Nuclear Restart Project
BBPS	Brighton Beach Power Station
BioMAP	Biological Monitoring and Assessment Program
CCME	Canadian Council of Ministers of the Environment
CCREM	Canadian Council of Resource and Environment Ministers
CEAA	*Canadian Environmental Assessment Act*
C of A	Certificate of Approval
CNSC	Canadian Nuclear Safety Commission
CORMIX	Cornell Mixing Zone Expert System
CPUE	Catch-per-unit-effort
CTMax	Critical thermal maximum (at which point locomotory movement becomes disorganized and the animal loses its ability to escape from conditions that may ultimately lead to its death)
CWC	Cold weather conditions
CWQG	Canadian Water Quality Guideline
D.O.	Dissolved oxygen
Diff.	Difference
EA	Environmental assessment
EPI	Ephemeroptera, Plecoptera, Trichoptera
GS	Generating station

HBI	Hilsenhoff biotic index
IDH	Incremental degree-hour
IJC	International Joint Commission
ILL	Incipient lethal temperature
LC50	Lethal concentration of a substance tolerated by 50% of the test population
LHT	Lethal high temperature
LT50	Incipient lethal temperature tolerated by 50% of the test population for a sustained period
MDDEP	Ministère du Développement durable, de l'Environnement et des Parcs du Québec (Quebec's Ministry of Sustainable Development, Environment and Parks)
MEF	Ministère de l'Environnement et de la Faune (Quebec's former Ministry of Environment and Wildlife)
MOE	Ontario Ministry of the Environment (formerly MOEE)
MOEE	Ontario Ministry of Environment and Energy
MRNF	Ministère des Ressources naturelles et de la Faune (Quebec's Ministry of Natural Resources and Wildlife)
MWAT	Maximum weekly average temperature
MWMT	Mean weekly maximum temperature
NGS	Nuclear generating station
O.Reg.	Ontario Regulation
PEC	Portlands Energy Centre
PWQO	Provincial Water Quality Objective
RMA	Resource Management Associates
SLSMC	St. Lawrence Seaway Management Corporation
SPP	Sundance Power Plant
TCP	Thorold Cogeneration Project
TLm	Median tolerance limit at which 50% of the organisms are dead
TRCA	Toronto and Region Conservation Authority
USACOE	United States Army Corps of Engineers
U.S. EPA	United States Environmental Protection Agency
ULT	Upper lethal temperature
VEC	Valued ecosystem component

WPP	Wabamun Power Plant
WQI	Water quality index
WSP	Water Supply Plant
WWC	Warm weather conditions
YOY	Young-of-the-year
ΔT	Change in temperature

Measurement Units

°C	degree Celsius (0°C = 32°F)
°F	degree Fahrenheit
/mL	per millilitre
/m^2	per square metre
μ	micron (micrometre)
μg/L	microgram per litre
cm	centimetre
d	day
ft	foot
g/m^2	gram per square metre
h	hour
h/y	hour per year
ha	hectare
km	kilometre
km^2	square kilometre
m	metre
m/s	metre per second
m^2	square metre
m^3/s	cubic metre per second
min	minute
mm	millimetre
MW	megawatt
y	year

1.0 INTRODUCTION

This guidance document on freshwater thermal discharge has been prepared for use by Environment Canada staff involved in environmental assessment (EA) reviews and by proponents required to conduct an EA. The information and recommendations presented in this document are in no way to be interpreted as any type of acknowledgement, compliance, permission, approval, authorization, or release of liability related to applicable federal or provincial statutes and regulations. Responsibility for achieving regulatory compliance and cost-effective risk and liability reduction lies solely with the project proponent.

Facilities proponents and operators should evaluate and regulate their thermal discharges on the basis of site-specific environmental impact assessments rather than using fixed, detailed numerical standards for broad areas, such as water quality standards for whole provinces. This type of assessment is likely to result in numerical limits for the facility (e.g., maximum discharge temperature), with the benefit that the numerical limit would be based on the site-specific aquatic water environment and its biological community. This is in contrast to making decisions based almost exclusively on a single most thermally sensitive species or life stage.

This guidance document presents the following:

- environmental quality guidelines for thermal effluents, based on federal and provincial legislation, policies and guidelines;
- environmental baseline and facility design/operation information requirements to undertake thermal plume modelling;
- guidance on the selection of the appropriate thermal plume modelling approach(es);
- guidance on thermal plume effects assessment;
- identification of common mitigative strategies/technologies for thermal discharges; and
- recommended follow-up monitoring approaches to confirm predicted thermal effects.

This guidance document provides a more comprehensive elaboration of a draft interim document “Best Practices Guidance Document for Assessing Environmental Effects of Thermal Effluents including Data Sources and Requirements” (Barker and Noakes, 2004). In addition, it presents a series of case studies to illustrate how the environmental assessment process takes site-specific factors into account, from the spatial layout, thermal plume configurations and biology of the area.

2.0 FEDERAL AND PROVINCIAL LEGISLATION, POLICIES AND GUIDELINES

2.1 Canada

Subsection 36(3) of the *Fisheries Act* specifies that, unless authorized by federal regulation, no person shall deposit or permit the deposit of deleterious substances of any type in water frequented by fish, or in any place under any conditions where the deleterious substance, or any other deleterious substance that results from the deposit of the deleterious substance, may enter any such water. In the definition of deleterious, the *Fisheries Act* includes “any water … treated, processed or changed, by heat or other means, from a natural state that it would, if added to any other water, degrade or alter or form part of a process of degradation or alteration of the quality of that water so that it is rendered or is likely to be rendered deleterious to fish or fish habitat”. Subsection 36(3) makes no allowance for a mixing or dilution zone.

In the application of the *Fisheries Act*, court cases have accepted that a discharge or effluent that is acutely lethal to fish is deleterious. In other words, results of tests designed to determine whether fish will die in an effluent or discharge within a specified time period will determine one aspect of deleteriousness. However, any substance with a potentially harmful chemical, physical or biological effect on fish or fish habitat would be deleterious. For example, substances that smother rearing areas or spawning grounds, or interfere with reproduction, feeding or respiration of fish, at any point in their life cycle would also be deleterious. Therefore, the discharge of a thermal effluent would be considered a deleterious substance if it had a potentially harmful chemical, physical or biological effect on fish or fish habitat.

Based on a special meeting of the Detroit River Canadian Cleanup Habitat Subcommittee at the University of Windsor on 30 January 2002 involving representatives of the proposed Brighton Beach Power Station (BBPS), federal and provincial government agencies, non-government organizations and university researchers, it was agreed that thermal discharge does not directly alter fish habitat. However, there may be some localized alteration of the use of this habitat by fish, for example, when the heated discharge promotes or changes the growth of habitat formers such as benthic algae and macrophytes.

National guidelines for water temperatures are provided by the Canadian Council of Ministers of the Environment (CCME). The Canadian Water Quality Guideline (CWQG) for temperature (as well as other parameters) for the protection of aquatic life has also been adopted by four provinces: New Brunswick, Nova Scotia, Prince Edward Island and Newfoundland and Labrador, as well as two territories, Yukon and Nunavut.

The CWQG for temperature is provided below (CCREM, 1987).

1. Thermal Stratification

Thermal additions to receiving waters should be such that thermal stratification and subsequent turn-over dates are not altered significantly from those existing prior to the addition of heat from artificial origins.

2. Maximum Weekly Average Temperature

Thermal additions to receiving waters should be such that the allowable maximum weekly average temperature (MWAT) is not exceeded.

a) In the warmer months, the MWAT is determined by adding to the physiological optimum temperature (usually for growth) a factor calculated as one-third of the difference between the ultimate upper incipient lethal temperature and the optimum temperature for the most appropriate life stage of the sensitive important species that normally is found at that location and time. Some MWAT values are shown in U.S. EPA (1976). (Note: MWATs for many fish species are also available in Wismer and Christie, 1987).

b) In the colder months, the MWAT is an elevated temperature that would still ensure that important species would survive if the temperature suddenly dropped to the normal ambient temperature. The limit is the acclimation temperature minus 2°C when the lower lethal threshold temperature equals the ambient water temperature.

c) During reproductive seasons, the MWAT meets specific site requirements for successful migration, spawning, egg incubation, fry rearing and other reproductive functions of important species.

d) At a specific site, the MWAT preserves normal species diversity or prevents undesirable growths of nuisance organisms.

3. Short-term Exposure to Extreme Temperature

Thermal additions to receiving waters should be such that the short-term exposures to maximum temperatures as calculated in a) and b) are not exceeded. Exposures should not be so lengthy or frequent as to adversely affect the important species.

a) For growth, the short-term maximum temperature is the 24-h median tolerance limit, minus two degrees, at an acclimation temperature approximating the MWAT for that month.

b) The short-term maximum temperature for the season of reproduction should not exceed the maximum incubation temperature for successful embryo survival, or the maximum temperature for spawning.

The CWQG for temperature is summarized as follows in CCME (1999) with an indication to consult CCREM (1987) for more information:

- **Thermal Stratification**: Thermal additions to receiving waters should be such that thermal stratification and subsequent turnover dates are not altered significantly from those existing prior to the addition of heat from artificial origins;
- **MWAT**: Thermal additions to receiving waters should be such that the MWAT is not exceeded; and
- **Short-term Exposure to Extreme Temperature**: Thermal additions to receiving waters should be such that the short-term exposures to maximum temperatures are not exceeded. Exposures should not be so lengthy or frequent as to adversely affect the important species.

2.2 British Columbia

The British Columbia (BC) water quality guidelines for temperature for the protection of freshwater aquatic life are based on a report prepared by Oliver and Fidler (2001), which presents a background literature review, recommends guidelines and provides supporting documentation. These guidelines are presented below.

Streams with Bull Trout (*Salvelinus confluentus*) and/or Dolly Varden (*S. malma*)

- Maximum daily temperature is 15ºC.
- Maximum incubation temperature is 10ºC.
- Minimum incubation temperature is 2ºC.
- Maximum spawning temperature is 10ºC.

These two species have been demonstrated to have the highest thermal sensitivity of native fish species tested.

Streams with Known Fish Distribution

- ±One degree change beyond optimum temperature range as shown in Table 1 for each life history phase of the most sensitive salmonid species present.
- Hourly rate of change not to exceed one degree.

Streams with Unknown Fish Distribution

- Mean weekly maximum temperature (MWMT) = 18ºC (maximum daily temperature = 19ºC).
- Hourly rate of change not to exceed one degree.
- Maximum daily incubation temperature = 12ºC (in the spring and fall).

The MWMT is defined as the average of the warmest daily maximum temperatures for seven consecutive days.

Lakes and Impoundments

- ±One degree change from natural ambient background.

Table 1: Optimum[1] Temperature (ºC) Ranges of Specific Life History Stages of Salmonids and Other Coldwater Species for BC Guideline Application

Species	Incubation	Rearing	Migration	Spawning
Salmon (*Oncorhynchus*)				
Chinook (*O. tshawytscha*)	5.0-14.0	10.0-15.5	3.3-19.0	5.6-13.9
Chum (*O. keta*)	4.0-13.0	12.0-14.0	8.3-15.6	7.2-12.8
Coho (*O. kisutch*)	4.0-13.0	9.0-16.0	7.2-15.6	4.4-12.8
Pink (*O. gorbuscha*)	4.0-13.0	9.3-15.5	7.2-5.6	7.2-12.8
Sockeye (*O. nerka*)	4.0-13.0	10.0-15.0	7.2-15.6	10.6-12.8
Trout (*Salmo*)				
Brown (*S. trutta*)	1.0-10.0	6.0-17.6	-	7.2-12.8
Cutthroat (*S. clarki*)	9.0-12.0	7.0-16.0	-	9.0-12.0
Rainbow (*S. gairdneri*)	10.0-12.0	16.0-18.0	-	10.0-15.5
Char (*Salvelinus*)				
Arctic char (*S. alpinus*)	1.5-5.0	5.0-16.0	-	4.0
Brook trout (*S. fontinalis*)	1.5-9.0	12.0-18.0	-	7.1-12.8
Bull trout (*S. confluentus*)	2.0-6.0	6.0-14.0	-	5.0-9.0
Dolly varden (*S. malma*)	-	8.0-16.0	-	-
Lake trout (*S. namaycush*)	5.0	6.0-17.0	-	10.0
Grayling (*Thymallus*)				
Arctic grayling (*T.arcticus*)	7.0-11.0	10.0-12.0	-	4.0-9.0
Whitefish				
Lake (*Coregonus clupeaformis*)	4.0-6.0	12.0-16.0	-	>8.0
Mountain (*Prosopium williamsoni*)	<6.0	9.0-12.0	-	<6.0
Other Species				
Burbot (*Lota lota*)	4.0-7.0	15.6-18.3	-	0.6-1.7
White sturgeon (*Acipenser transmontanus*)	14.0-17.0	-	-	14.0

Oliver and Fidler (2001) indicated that further definition is required on the size of acceptable mixing zones associated with point-source discharges to lakes and rivers before

[1] Oliver and Fidler defined optimum temperatures indirectly by stating that criteria exceeding optimum temperatures would have serious implications on growth/development, disease resistance, reproduction and species interactions.

temperature guidelines are formalized. Mixing zone boundary considerations should be resolved with future deliberations aimed at water use classifications.

2.3 Alberta

The Alberta water quality guideline for the protection of freshwater aquatic life is “not to be increased by more than three degrees above ambient water temperature” (Alberta Environment, 1999).

In addition, Alberta has adopted the CWQG for temperature as its guideline (CCME, 1999) as paraphrased below:

- “Thermal additions should not alter thermal stratification or turnover dates, exceed maximum weekly average temperatures, nor exceed maximum short-term temperatures.”

2.4 Saskatchewan

Saskatchewan Environment (2006) has also adopted the CWQG for temperature (CCME, 1999) as its surface water quality objective as paraphrased below:

- “Thermal additions should not alter thermal stratification or turnover dates, exceed maximum weekly average temperatures, nor exceed maximum short-term temperatures.”

In addition, specific effluent mixing zone guidelines have been developed:

- Size or shape should not cause or contribute to the impairment of existing or likely water uses.
- The existing General Objectives for Effluent Discharges (Saskatchewan Environment, 2006) should be achieved at all sites within the limited use zone.
- The limited use zone in streams and rivers should be apportioned no more than 25% of the cross-sectional area or volume of flow, nor more than one-third of the river width at any transect in the receiving water during all flow regimes that equal or exceed the 7Q10 flow for the area. Surface water quality objectives **applicable to the area** must be achieved at all points along a transect at a distance downstream of the effluent outfall to be determined on a case-by-case basis.
- In lakes and other surface impoundments, surface water quality objectives applicable to that waterbody must be achieved at all points beyond a radius of 100 m from the effluent outfall. The volume of limited use zones in lakes should not exceed 10% of that part of the receiving waters available for mixing.

- The mixing zone should be designed to allow an adequate zone of passage for the movement or drift of all stages of aquatic life; specific portions of a cross-section of flow or volume may be arbitrarily allocated for this purpose.
- Mixing zones should not interfere with the migratory routes, natural movements, survival, reproduction, growth, or increase the vulnerability to predation, of any representative aquatic species or endangered species.
- Mixing zones should not interfere with fish spawning and nursery areas.
- When two or more mixing zones are in close proximity, they should be so defined that a continuous passageway for aquatic life is available.
- When two or more mixing zones overlap the combination of the effluent, plumes should not result in unacceptable synergistic or antagonistic effects on aquatic life or other water uses downstream of the mixing zone(s).
- Mixing zones should not cause an irreversible organism response or attract fish or other organisms and thereby increase their exposure period within the zone.
- The 96-h LC50 toxicity criteria for indigenous fish species and other important aquatic species should not be exceeded at any point in the mixing zones.
- Mixing zones should not result in contamination of natural sediments so as to cause or contribute to excursions of the water quality objectives outside the mixing zone.
- Mixing zones should not intersect domestic water supply intakes, bathing areas or other sensitive designated use areas.
- Specific numerical water quality objectives may be established by the Department for such variables or constituents thought to be of significance within the effluent mixing zone.
- Defining the effluent mixing zone may need to be done on a case-by-case basis, in consultation with the Department, particularly where effluent is discharged into smaller waterbodies (i.e., streams and small lakes).

2.5 Manitoba

The Manitoba water quality objective for temperature (Williamson, 2002) is based on the U.S. EPA (1976, 1999) quality criteria for water, as indicated in Table 2.

Table 2: Manitoba Surface Water Quality Objective for Temperature for the Protection of Aquatic Life and Wildlife

Water Quality Objectives	Applicable Period	Averaging Duration	Allowable Exceedance Frequency	Design Flow
Site-specific objectives will be developed considering the following:				
(1) Thermal additions should be such that thermal stratification and subsequent turnover dates are not altered from those existing prior to the addition of heat from artificial origin.	All periods			
(2) One limit which consists of a MWAT that:				
(a) In the warmer months is determined by adding to the physiological optimum temperature (usually for growth) a factor calculated as one-third of the difference between the ultimate upper incipient lethal temperature and the optimum temperature for the most sensitive important species (and appropriate life stages) that normally is found at that location and time; and	Warmer months	7 days	Not more than once each 3 years, on average	4-day, 3-year or 7Q10
(b) In the colder months is an elevated temperature that would still ensure that important species would survive if the temperature suddenly dropped to the normal ambient temperature; or	Colder months			
(c) During reproduction seasons meets specific site requirements for successful migration, spawning, egg incubation and other reproductive functions of important species; or	Reproduction season			
(d) At a specific site is found necessary to preserve normal species diversity or prevent undesirable growths of nuisance organisms.	All periods			
(3) A second limit which is the time-dependent maximum temperature for short exposures.	All periods	Site-specific or regional-specific		1-day, 3-year or 1Q10
(4) Maximum limits may be specified for incremental temperature fluctuations necessary to protect aquatic life from sudden temperature changes.	All periods	Site-specific or regional-specific		

In addition, Williamson (2002) provides guidance on mixing zones as follows:

Mixing zones should be determined on a case-by-case basis utilizing a thorough knowledge of local conditions. Normally, geometric size constraints will not be assigned due to the complex nature of the mixing properties of liquids. The following guidelines should apply to mixing zones, where applicable, in order to minimize the loss of value such that water uses are not unacceptably impaired (U.S. EPA 1994a,b, with modifications):

(a) The mixing zone should be as small as practicable and should not be of such size or shape as to cause or contribute to the impairment of water uses outside the zone;

(b) The mixing zone should be designed to allow an adequate zone of passage for the movement or drift of all stages of aquatic life:

  (i) For those materials that elicit an avoidance response from aquatic life, the mixing zone should contain not more than 25% of the cross-sectional area or volume of flow at any transect in the receiving water. Should a proportion of the stream width greater than 25% be selected for these materials, the mixing zone could act similar to a physical barrier and could effectively preclude the passage of aquatic life;

  (ii) The mixing zone should not be acutely lethal to aquatic life passing through the mixing zone. Thus, for toxic materials, acute lethality within the mixing zone is a function of concentration (i.e., temperature) and the duration of exposure. Whole effluents should not be acutely lethal to aquatic life, as demonstrated by 96-h LC50 tests done on appropriate species, unless it can be shown either through mixing zone modelling that mixing of the effluent with the receiving water will be achieved in a relatively rapid and complete manner (e.g., no more than a 10% difference in bank-to-bank concentrations within a longitudinal distance of not more than two stream or river widths) or through other scientifically rigorous methods that acute lethality will not occur within the mixing zone;

  (iii) Mixing zones should not interfere with the migratory routes essential to the reproduction, growth or survival of aquatic species;

  (iv) Mixing zones should not cause an irreversible organism response, or increase the vulnerability to predation;

  (v) When two or more mixing zones are in close proximity, they should be so defined that a continuous passageway for aquatic life is available; and

  (vi) Mixing zones should not intersect the mouths of rivers.

(c) Mixing zones should not interfere with spawning and nursery areas;

(d) In lakes and other surface impoundments, the volume of mixing zones should not exceed 10% of the volume of those portions of the receiving waters available for mixing or 100 m in radius, whichever is less;

(e) Mixing zones should not contaminate natural sediments so as to cause or contribute to exceedances of the water quality standards, objectives and guidelines outside the mixing zone;

(f) Mixing zones should not intersect domestic water supply intakes or bathing areas;

(g) Mixing zones generally do not apply to ground water; and

(h) The applicable narrative *Tier III – Water Quality Guidelines* should apply at all points within mixing zones to avoid objectionable nuisance conditions and to protect uses outside mixing zones from unacceptable effects.

2.6 Ontario

The Ontario Provincial Water Quality Objective (PWQO) for temperature (MOEE, 1994a) is provided below.

1. General

The natural thermal regime of any body of water shall not be altered so as to impair the quality of the natural environment. In particular, the diversity, distribution and abundance of plant and animal life shall not be significantly changed.

2. Waste Heat Discharge

(a) Ambient Temperature Changes

The temperature at the edge of a mixing zone shall not exceed the natural ambient water temperature at a representative control location by more than 10 Celsius degrees. However, in special circumstances, local conditions may require a significantly lower temperature difference than 10 degrees. Potential dischargers are to apply to the Ontario Ministry of the Environment (MOE) for guidance as to the allowable temperature rise for each thermal discharge. This Ministry will also specify the nature of the mixing zone and the procedure for the establishment of a representative control location for temperature recording on a case-by-case basis.

(b) Discharge Temperature Permitted

The maximum temperature of the receiving body of water, at any point in the thermal plume outside a mixing zone, shall not exceed 30ºC or the temperature of a representative control location plus 10 degrees or the allowed temperature difference, whichever is the lesser temperature. These maximum temperatures are to be measured on a mean daily basis from continuous records.

(c) Taking and Discharging of Cooling Water

Users of cooling water shall meet both the PWQO for temperature outlined above and the Procedures for the Taking and Discharge of Cooling Water as outlined in the MOE publication *Deriving Receiving-Water Based, Point-Source Effluent Requirements for Ontario Waters* (MOEE 1994b).

Water Management Policy 5 of the MOE states that: “mixing zones should be as small as possible and not interfere with beneficial uses. Mixing zones are not to be used as an alternative to reasonable and practical treatment” (MOEE, 1994a).

The MOE defined a mixing zone as an area of water contiguous to a point source or definable non-point source where the water quality does not comply with one or more of the PWQOs (MOEE 1994b). A mixing zone is, under no circumstances, to be used as an alternative to reasonable and practical treatment. It must be designed to be as small as possible. It is one factor to be considered in establishing effluent requirements.

The concept of mixing zones recognizes that the release of the aqueous component of adequately treated municipal and industrial wastes to rivers, streams and lakes does occur. As a general principle, the dilution of such effluents, and thus the use of mixing zones, should be minimized and limited to conventional pollutants. The mixing zone principle does not apply to hazardous substances identified in Tables 1.7 and 1.8 of the MOE report *Candidate Substances for Bans, Phase-Outs or Reductions – Multimedia Revision* (MOEE 1993a).

Conditions within a mixing zone must not result in irreversible environmental damage, risk to ecosystem integrity or risk to human health. Mixing zones cannot interfere with other water uses such as drinking water supply or recreation.

Terms and conditions related to a mixing zone will be designated on a case-by-case basis and may be specified in Certificates of Approval, Control Orders, Requirements and Directions, or approvals to proceed under the *Environmental Assessment Act*.

As effluent loading requirements are based on careful design, so too should mixing zones be carefully planned on a site-specific basis including consideration of water quality, seasonal streamflow and current patterns, physical factors, biotic communities and spawning areas in and adjacent to the mixing zone, nearby water uses such as bathing beaches and drinking water intakes as well as other wastewater discharges.

As general guidance in the design of mixing zones, the following range of concerns must be adequately addressed:

1) In order to protect important aquatic communities (fish, invertebrates and plants) in the vicinity of mixing zones, no conditions within the mixing zone will be permitted which:
   a) are acutely lethal to aquatic life;
   b) cause irreversible responses which could result in detrimental post-exposure effects;
   c) result in bioconcentration of toxic materials which are harmful to the organism or its consumer;
   d) attract organisms to the mixing zones, resulting in a prolonged exposure; and
   e) create a barrier to the migration of fish or other aquatic life.

2) To ensure the protection of acceptable aesthetic conditions, mixing zones should not contain:
   a) materials which form objectionable deposits (e.g., scums, oil or floating debris);
   b) substances producing objectionable colour, odour, taste or turbidity;
   c) substances which produce or contribute to the production of objectionable growths of nuisance plants and animals; and
   d) substances that render the mixing zone aesthetically unacceptable.

3) Mixing zones should not impinge upon existing municipal and other water supply intakes, bathing beaches or important fish spawning areas. Conversely, new intakes or aquatic recreation areas should not be constructed within the boundaries of existing mixing zones. However, with detailed knowledge of the effluent and receiver characteristics and the resulting conditions within the mixing zone, some mixture of uses might be permissible.

4) Mixing zones may overlap unless the combined effects exceed the conditions specified in this section.

5) When background water quality conditions at a proposed mixing zone site do not meet one or more of the PWQOs, effluent requirements should be established that ensure, at the very least, that background water quality is not further degraded.

Procedures for the discharging of cooling water are also provided by the MOE, as follows (MOEE 1994b):

Method of Disposal

Discharge of waste heat shall be into such areas and locations and in such quantities as may be allowed by the MOE on a case-by-case basis. The method of discharge shall be such that rapid mixing occurs with the receiving water, thus minimizing the area affected.

Scientific studies must demonstrate to the satisfaction of the MOE that the discharge design and location are optimal for minimizing the area affected by the discharge and the environmental impact of the discharge. These studies will be required for each discharge with a capacity greater than 10 m^3/s. The MOE may require these studies for discharges of lower capacity, particularly for discharges to inland waters. A discharge of waste heat should not affect the water temperature of any water intake or fish spawning area.

These studies will contribute towards the EA process as well.

Alternative Cooling Facilities

In those instances where significant effects can be clearly predicted, or demonstrated for an existing facility, alternative cooling facilities are to be employed. In those instances where potential harmful effects may arise but cannot be clearly predicted, generating stations should be initially designed so that alternative cooling facilities can be added at such time as evidence indicates sufficient adverse effects.

Circulation Patterns

The discharge of cooling water and structures built for this purpose shall not alter the local existing circulation patterns such that other water uses, sedimentation, spawning or fishing grounds are adversely affected.

Beneficial Uses

Wherever possible, all or part of the waste heat should be used in a beneficial way, ideally through redistribution rather than direct discharge into a water body or wastewater system.

2.7 Quebec

The Quebec surface water quality criterion for temperature for the protection of aquatic life (MDDEP, 2007; MDDEP, 2009) is provided below, followed by a translation.

> "Toute augmentation artificielle de la température ne doit pas:
>
> - modifier la température de l'eau sur tout un tronçon de rivière ou une portion de lac avec pour résultat le déplacement prévisible ou la modification des populations aquatiques présentes ou potentielles;
> - altérer certaines zones sensibles localisées, telle une frayère; et
> - tuer les organismes vivants à proximité d'un rejet.
>
> De plus, le milieu ne doit pas subir de changements brusques de température occasionnés, par exemple, par un arrêt subit d'un rejet thermique en saison froide."

Translation:

> Any artificial increase in temperature should not:

- modify water temperature in the entire section of river or portion of a lake with the anticipated result of displacing or modifying the present or potential aquatic populations;
- result in the deterioration of specific localized sensitive areas, such as a spawning ground; and
- result in organism mortality near the discharge.

Moreover, the receiving water should not undergo abrupt changes in temperature due, for example, to the sudden cessation of a thermal discharge during the cold season.

The Quebec Ministère des Ressources naturelles et de la Faune provides the following information on tolerated mixing zones (MEF 1996):

> "Since the goal is to ensure maximum protection of all receiving waters, any potential exceedance must be confined to a very limited mixing zone located immediately downstream of the pollutant discharge point."

The principle underlying this approach is that, on occasion, a very restricted mixing zone may be damaging without harming the watercourse as a whole. However:

- the mixing zone must be as small as possible;
- a passageway for fish and other mobile aquatic life must be maintained;
- the mixing zone must not interfere with spawning grounds, areas for raising young, or migration routes;
- the mixing zone must not be located at the mouth of a tributary (to avoid creating a wall which might hinder fish movement);
- mixing zones must not overlap to prevent water quality criteria exceedances beyond the mixing zone;
- the mixing zone must not permit sedimentation or accumulation to toxic levels; and
- the mixing zone must be eliminated or minimized if the effluent that contains toxics is likely to attract fish.

To respect these clauses, the notion of a tolerable mixing zone has been defined, using physical constraints for stream design flow conditions.

In general, the limits of the tolerable mixing zone correspond to the more restrictive of the following:

- maximum length: 300 m;
- width: less than half of the width of the watercourse, to a maximum of 50 m;
- minimal dilution factor: 0.01; and

- when a specific use, whether actual or potential, occurs in the mixing zone, the above-mentioned limits are modified to protect this use where it occurs.

For very small watercourses, the second limit may, for operational purposes, correspond to half (for toxics) or all (for conventional pollutants) of the low-water stream design flow.

Used in conjunction with flows corresponding to the critical low-water level period, this definition of mixing zone generally ensures that no section of a watercourse deteriorates to any significant degree.

2.8 International Joint Commission

The International Joint Commission (IJC, 1978, 1987) has established a Specific Objective for water temperature for the boundary waters of the Great Lakes System, as follows:

- "There should be no change in temperature that would adversely affect any local or general use of the waters."

2.9 Regulatory Requirements

Discharges of cooling water to waterbodies require federal or provincial regulatory approval as part of a permit or Certificate of Approval (C of A), which delineates an objective and/or maximum limit for temperature in the discharge, as well as temperature and any other monitoring requirements. The Terms and Conditions for thermal effluent stipulated in the Cs of A for the Brighton Beach Power Station (BBPS), the Portlands Energy Centre (PEC), the Thorold Cogeneration Project (TCP) and the Bruce A Nuclear Restart Project (BANRP) are discussed below.

2.9.1 Brighton Beach Power Station

The C of A (Industrial Sewage Works Number 2256-569K5Y) for the BBPS in Windsor stipulates that cooling water discharge must meet an effluent temperature objective of 32ºC, with a maximum effluent temperature of 34°C and a maximum temperature rise (ΔT) between the intake and the outfall of 10 degrees. Temperature at the intake and outfall must be continuously monitored.

Whenever the cooling water discharge temperature exceeds the effluent objective but is less than or equal to 34ºC, BBPS must determine for each 24-h period values of the parameter, incremental degree-hours (IDH) for the effluent. IDH values are calculated by multiplying the calculated temperature difference in ºC between the effluent temperature and effluent temperature objective, by the number of hours over which the specific calculated increment is recorded.

When an IDH parameter value equals or exceeds 1.0 in any 24-h period, BBPS must:

- notify the District Office as soon as possible;

- no later than 30 days following the day on which the first IDH value equal to or greater than 1.0 was determined, conduct a macroinvertebrate study upstream and downstream of the effluent discharge point, near the respective former J.C. Keith Generating Station (GS) water lot property lines;
- continue to calculate and record, for all effluent temperatures greater than the effluent temperature objective but less than or equal to 34ºC, IDH parameter values, until such time as the field collection for the benthic macroinvertebrate study has been completed; and
- within 60 days of the field collection, prepare and submit the results of the benthic macroinvertebrate study together with a listing of dates, effluent temperatures and their durations for which the calculated IDH parameter values were equal to or greater than 1.0 and which occurred prior to the field collection.

Once the benthic macroinvertebrate field collection has been completed, another benthic macroinvertebrate study would be required if a new IDH parameter value equal or greater than 1.0 has been determined in the period following the completion of the field collection, whereby the requirements listed above would once again apply.

In addition, BBPS "shall establish and carry out prior to the start-up of the works and periodically during operation of the works, a macroinvertebrate study of the areas immediately downstream and upstream of the Site Outfall to monitor the impact of chlorination/dechlorination operations". The study should provide the following information:

- an initial background survey of benthic organisms upstream and downstream immediately prior to the works coming into service;
- a benthic survey immediately after completion of any seasonal continuous chlorination for zebra mussel control and general frequency of sampling;
- the number and location of sites ensuring that the locations are inside and outside the chlorination/thermal plumes;
- the need for separate sites and seasonal monitoring to study separately the chlorination and thermal effects;
- the number of samples at each site, the size of the sampling area and the mesh sampling size;
- methods of analysis, i.e., taxonomical and multivariate to determine impacts and comparisons with other relevant study/literature data; and
- duration of the study.

The BBPS commenced commercial operations on 16 July 2004 and operates on a peak demand basis, generally during the day on most weekdays (not on weekends). Because the BBPS cooling water discharge occurs at the shoreline of the Detroit River and the thermal plume was predicted to extend along the Canadian shoreline downstream, the monitoring program as outlined above required by the MOE was based on nearshore benthic macroinvertebrate communities. No important fish spawning habitat was observed in the nearshore area adjacent to the BBPS and up to 1 km downstream (Eakins and Fitchko, 2001, 2003).

2.9.2 Portlands Energy Centre

In contrast, the C of A (Industrial Sewage Works Number 8449-5SMQX4) for the PEC in Toronto stipulates that cooling water discharge must meet an effluent temperature objective of 26.0ºC, with a maximum effluent temperature of 30ºC and a maximum ΔT of 5.5Cº. Temperature at the intake and outfall must be continuously monitored.

The PEC came into operation in 2009. Cooling water is taken from the Ship Channel and discharged via the former R.L. Hearn GS Discharge Channel into the Outer Harbour (Lake Ontario).

The lower temperature limits (compared with those for BBPS) reflect the lower facility design ΔT as well as the location of the intake within the cooler waters of the hypolimnion due to thermal stratification of the Ship Channel during the summer. An additional monitoring requirement with respect to cooling water discharge was for *Escherichia coli* during the summer due to concern that stormwater and combined sewer overflows to the Ship Channel would be entrained during and after rainfall events and potentially affect a public bathing beach (Cherry Beach) approximately 1.5 km from the outfall.

PEC had also committed to a monitoring program involving continuous temperature monitoring over the spawning period in nearshore spawning habitat along Cherry Beach and within the embayments of the Outer Headland. In addition, Environment Canada requested that additional fish surveys be undertaken to confirm that the Outer Harbour nearshore in the area of the Discharge Channel provides only foraging and limited nursery habitat for warmwater fish.

No benthic macroinvertebrate community monitoring was recommended, since cooling water discharge from the Discharge Channel was predicted to only affect the surface waters of the uppermost basin of the Outer Harbour.

2.9.3 Thorold Cogeneration Project

For the TCP, the C of A (Industrial Sewage Works Number 7868-6URPLM) stipulates that cooling water discharge must meet a maximum effluent temperature of 32ºC and a maximum ΔT of 10Cº. Temperature at the intake and outfall must be continuously monitored. The C of A also stipulates Special Operating Requirements as follows:

General

(1) The Owner shall establish, prior to the commencement of operation of the works, the procedures and instrumentation to allow the following flow information to be obtained and recorded by the Owner:

(a) real time, continuous flow through the Welland Canal Power House;

(b) an estimate of the daily average flow at Weir 7 from the St. Lawrence Seaway Management Corporation (SLSMC), which may be on a one-day delayed basis: and

(c) an estimate of the daily average flow through Lock 7 from the SLSMC which may be on a one-day delayed basis and which may be estimated on the basis of the number of vessels passed through the Lock.

(2) The Owner shall establish and operate, prior to the commencement of operation of the works, continuous temperature monitors/detectors, in the Welland Canal, each with a minimum accuracy of ± four degrees, at the following locations, as approved by the District Manager:

(a) upstream of the Plant intake and outside of any potential zone of influence from the Plant, which will be deemed to be the Upstream Canal Water Temperature;

(b) at the Plant cooling water intake which will be deemed to be the Local Intake Water Temperature; and

(c) at two locations which are representative of the temperature in the Canal in the area of Lock 7.

The temperature information should be continuously recorded in the Plant Control Room.

During the Welland Canal Navigation Season

(3) Despite meeting the maximum effluent and ΔT temperature limitations at all times, whenever the Welland Canal Power House flow is less than 6 m^3/s, based on a 60-minute rolling average, during the Canal navigation season, the Owner shall operate the Plant in such a way as to ensure that there is no thermal impact caused by the Plant spent cooling water discharge on the aquatic life in the Canal.

(4) As a minimum, to comply with the requirements of Subsection (3), the Owner shall stop the discharge of all spent cooling water from the Plant to the Canal, whenever the temperature of the spent cooling water exceeds 33°C for any continuous period of six hours, at any time.

(5) The Owner may not restart the discharge of Plant spent cooling water to the Canal following a shutdown under Subsection (4), until either:

(a) the flow to the Welland Canal Power House is 6 or more m^3/s, based on a 60-minute rolling average and the temperature difference between the Upstream Canal Water Temperature monitoring point and the average of the two downstream Lock 7 area monitoring point temperatures is within two degrees based on a 60-minute average, or

(b) Lock 7 has been put through a minimum of three complete fill and empty cycles since the shutdown, should the Welland Canal Power House flow be less than 6 m^3/s, and the temperature difference between the Upstream

Canal Water Temperature monitoring point and the average of the two downstream Lock 7 area monitoring point temperatures is within two degrees based on a 60-minute average.

(6) The Owner shall report to the District Manager, each occurrence of a shutdown and restart resulting from the limiting conditions under Subsections (4) and (5), orally as soon as reasonably possible, and in writing within seven days of the first oral reporting of the information.

During the Welland Canal Non-navigation (Winter) Season

(9) On any given calendar day, during the non-navigation (winter) season, when the Plant is discharging spent cooling water to the Welland Canal and the flow at the Welland Canal Power House is less than 6 m^3/s, based on a 60-minute rolling average, during the discharge period, the Owner shall restrict the discharge of spent cooling water from the Plant to the Canal to the lesser of the following two time periods:

(a) 12 hours from the time the Welland Canal Power House flow first becomes less than 6 m^3/s, while the Plant is discharging spent cooling water, or

(b) the period of time until the calculated 60-minute rolling average temperature difference between the Local Intake Water Temperature and the Upstream Canal Water Temperature first exceeds two Celsius degrees, indicating entrainment of re-circulated heated water

and shall provide a minimum 12-h separation period between successive spent cooling water discharge periods arising from (a) or (b), above;

(10) The minimum 12-h separation period under Subsection (9) may be shortened at any time to allow the Plant to restart the discharge of spent cooling water to the Canal at any time when the Welland Canal Power House flow is increased to a minimum of 12 m^3/s, based on a 60-minute rolling average, and maintained at that level.

(11) The Owner shall report to the District Manager, each occurrence of a shutdown resulting from the limiting condition under Subsection (9)(b), orally as soon as reasonably possible, and in writing within seven days of the first oral reporting of the information.

These Special Operating Requirements were included in the C of A due to potential non-compliance with the maximum effluent temperature as a result of:

- high ambient water temperature in the Welland Canal during the summer; and
- upstream migration and entrainment of the thermal plume in the TCP intake during low flow conditions in the winter.

No biological monitoring is required due to the paucity of benthic macroinvertebrates in the Welland Canal (Fitchko, 2006a) and the poor fish habitat primarily providing for the migration of adult fish (Fitchko, 2007). The TCP, currently under construction, will operate in cogeneration mode for approximately 3500 h/y (40% of the year) as dispatched in accordance with electricity market conditions and will be capable of starting up and shutting down up to two times daily.

2.9.4 Bruce A Nuclear Restart Project

For the BANRP, the C of A specifies a ΔT limit of 11.1 degrees for the period of 15 April to 15 December and 13 degrees for the period of 15 December to 15 April, with a maximum temperature discharge limit of 32.0°C. This maximum temperature discharge limit may require occasional de-rating of the Bruce A units to comply with the C of A (Golder, 2005a). Temperature at the intake and outfall must be continuously monitored. Bruce Power has committed to the following monitoring programs (Golder, 2007):

- smallmouth bass (*Micropterus dolomieu*) recruitment and nesting in the Bruce A discharge channel;
- dissolved gases in the Bruce A discharge duct and discharge channel to assess the potential of gas bubble disease due to supersaturated effluent; and
- water temperature at the substrate.

In addition, Bruce Power will continue its involvement with the Whitefish Technical Working Group to investigate spawning activity of lake whitefish (*Coregonus clupeaformis*).

2.9.5 Case Studies

In the following, three of the aforementioned projects are presented as case studies with respect to baseline environmental data requirements, thermal plume modelling approaches, thermal plume effects assessment, mitigation strategies and/or monitoring requirements.

The BBPS discharges cooling water into a lotic system (Detroit River), whereas the PEC and TCP will be discharging into a lentic (Lake Ontario) and lotic/lentic (Welland Canal) system, respectively. These three projects were subject to the Environmental Screening Process for electricity projects as outlined in the MOE (2001) *Guide to Environmental Assessment Requirements for Electricity Projects* under the *Electricity Projects Regulation* (O.Reg. 116/01) of the Ontario *Environmental Assessment Act*.

In addition to these three natural-gas fired power plants, the BANRP discharging to Lake Huron is also provided as a case study. The Canadian Nuclear Safety Commission (CNSC) determined, pursuant to paragraph 5(1)(*d*) of the *Canadian Environmental Assessment Act* (CEAA), that a federal EA was required before the CNSC could amend the current Bruce A operating license to provide Bruce Power with authorization to refuel and restart Bruce A Units 1 and 2. Amendment of the operating license for the Bruce A facility is a trigger for the CEAA under the *Law List Regulations*. The CNSC's Screening Report concluded that the BANRP is not likely to cause significant adverse effects on the environment, thereby enabling the CNSC to issue Bruce Power an amended license to operate.

Available information on other Canadian facilities is also provided in this guidance document under various other topics. These include Selkirk GS discharging to Cooks Creek in Manitoba, Wabamun Power Plant (WPP) and Sundance Power Plant (SPP) discharging to Lake Wabamun in Alberta, Iroquois Falls GS discharging to the Abitibi River in Ontario, among others.

3.0 ENVIRONMENTAL BASELINE DATA REQUIREMENTS

Environmental baseline data are required to undertake the assessment of thermal discharge effects (see Chapter 5.0). Some of these data, together with facility design/ operation information, are required to undertake thermal plume modelling (see Chapter 4.0), which provides the basis for thermal plume effects assessment. Other data are required to establish the pre-operational biological communities with which predicted or measured biological changes will be compared.

Requisite environmental baseline information generally includes shoreline configuration/morphology, bathymetry, hydrology (for lotic systems), physical limnology (e.g., currents, waves, water levels, seiches), thermal regime (e.g., stratification, turnover), ice conditions, phytoplankton, zooplankton, aquatic vegetation, benthic macroinvertebrates, fish (including habitat type and utilization), aquatic avifauna and water uses. Site-specific considerations may require additional information or may make some of these topics inapplicable.

Considerable published baseline information from governmental agencies, universities, or other nearby installations is often available for these environmental study components, particularly for larger waterbodies. Other sources of information include topographic maps, hydrographic charts, as well as government and non-government organization websites (e.g., Water Survey of Canada, Bird Studies Canada and the Committee on the Status of Endangered Wildlife in Canada). In addition to these, it is recommended that confirmatory studies be conducted by the project proponent prior to construction and/or facility operation in order to acquire accurate and up-to-date baseline information for the site.

Field studies may be required to provide site-specific thermal and biological data. Thermal regime information, e.g., seasonal water temperatures in lotic systems or thermal stratification conditions in lentic systems, would be used for thermal plume modelling. The field studies can also include surveys of total bathymetry, currents and other hydrological/limnological parameters required for thermal plume modelling.

Site-specific studies should be undertaken for benthic macroinvertebrate communities (for shoreline discharges only) and fish species composition (including habitat type and utilization). Fish habitat determination would include site-specific information on shoreline configuration, bathymetry, substrate type, aquatic vegetation and nearshore structures. Fish habitat utilization would include spawning, nursery, foraging and migration. The delineation of the benthic macroinvertebrate and fish study areas would be dependent upon the results of thermal plume modelling. Field studies of phytoplankton and zooplankton are generally not needed because thermal discharges rarely have significant effects on these assemblages (yet some site-specific situations may warrant them).

The benthic macroinvertebrate survey plan for BBPS on the Detroit River and the fish survey plan for PEC on Lake Ontario are described below as examples of pre-operational field studies recently completed to provide baseline information and a basis for subsequent effects monitoring (see Chapter 7.0).

3.1 BBPS Macroinvertebrate Survey Plan

The BBPS site provides an example of a detailed macroinvertebrate study. As indicated in Section 2.9.1, the C of A for the BBPS stipulated a requirement for pre-operational and operational benthic macroinvertebrate community surveys. The pre-operational benthic survey plan was based on a thermal plume, predicted by the CORMIX model (see Section 4.1), that extended along the Canadian shoreline of the Detroit River (BEAK, 2001). The temperature difference was predicted to be reduced to within two degrees of ambient within approximately 1000 m under all conditions modeled. The width of the thermal plume at two degrees was estimated to be less than 140 m with the plume contacting the river bed up to a water depth of 1.5 m.

As a result, the sampling program consisted of four sampling locations: upstream control, at the water lot limit (approximately 75 m downstream of the proposed outfall), 500 m downstream and 1000 m downstream (Fitchko, 2006b,c). At each of the four locations, three sites were sampled at three water depths, i.e., 0.5, 0.9 and 1.5 m, to encompass the anticipated thermal plume's contact with the river bed. For subsequent operational surveys, the actual sampling locations were to be confirmed by temperature readings at the water-sediment interface (see Section 7.2).

At each station, three replicate samples were collected by mini-Ponar grab (grab sampling area of 0.023 m^2) and composited for taxonomic analysis. A total of 12 composite samples were collected from the four transects. Each composite sample was screened through a 500-μ sieve and preserved in 10% formalin. The sediment samples were characterized according to texture, odour and presence of oily material. Water quality at each location was assessed by on-site measurements of water temperature, dissolved oxygen, conductivity and pH. Observations were also made on water clarity and colour, as well as the presence of surface film, aquatic macrophytes and attached filamentous algae. All relevant information was documented on Benthic Field Sheets.

The organisms were identified by a qualified taxonomist to the lowest practical taxonomic level, which in most cases was to genus or species level. Table 3 indicates the level of identification used for major taxonomic groups as well as the taxonomic reference used to identify each particular group.

Table 3: Taxonomic Level and Primary Taxonomic References used in the Identification of Benthic Macroinvertebrates

Taxon	Level[2]	Reference
FLATWORMS	Class/Family/Species	Pennak (1989)
NEMATODES	Phylum	Pennak (1989)
NEMERTEANS	Phylum	Pennak (1989)
Hydra	Genus	Pennak (1989)
ANNELIDS		
Oligochaeta	Species	Wetzel *et al*. (2000); Kathman and Brinkhurst (1999); Brinkhurst (1986)
Leeches	Species	Klemm (1991)
ARTHROPODS		
Mites	Order	Thorp and Covich (1991); Pennak (1989)
Crustaceans		Thorp and Covich (1991); Pennak (1989)
Isopods	Genus	
Harpacticoids	Order	
Ostracods	Class	
Crayfish	Species	Thorp and Covich (1991); Crocker and Barr (1968)
Amphipods	Genus/Species	Thorp and Covich (1991); Bousfield (1967)
Gammarus	Species	Holsinger (1976); Bousfield (1967)
Hyalella	Genus	Bousfield (1967)
INSECTS	Genus	Merritt and Cummins (1996); Hilsenhoff (1995)
Beetles	Genus	Merritt and Cummins (1996)
Elmidae	Species	Hilsenhoff and Schmude (1992); Brown (1972)
Dytiscidae	Genus/Species	Larson *et al*. (2000)
Caddisflies	Genus	Merritt and Cummins (1996); Wiggins (1996)
Hydropsyche	Species	Schefter and Wiggins (1986); Schuster and Etnier (1978)
Dragonflies/Damselflies	Genus/Species	Westfall and May (1996); Hilsenhoff (1995); Walker and Corbet (1978); Walker (1953, 1958)
Mayflies	Genus	Merritt and Cummins (1996); Edmunds *et al*. (1976)
Baetidae	Genus/Species	McCafferty (2000); Waltz (1994); McCafferty and Waltz (1990); Morihara and McCafferty (1979); Ide (1937)
Ephemerella	Species	Allen and Edmunds (1965)
Ephemeridae	Species	McCafferty (1976)
Stenonema	Species	Bednarik and McCafferty (1979)
Stoneflies	Genus	Stewart and Stark (1988)
Isoperla	Species	Hitchcock (1974)
Leuctridae	Species	Harper and Hynes (1971a)
Nemouridae	Species	Harper and Hynes (1971b)
Perlidae	Species	Hitchcock (1974)
Taeniopterygidae	Species	Fullington and Stewart (1980); Harper and Hynes (1971c)
True Flies	Genus	Merritt and Cummins (1996); Hilsenhoff (1995)
Chironomidae	Genus/Species	Epler (2001); Maschwitz and Cook (2000); Oliver and Dillon (1990); Oliver and Roussel (1983); Simpson *et al*. (1983); Wiederholm (1983); Simpson and Bode (1980); Jackson (1977)
Simuliidae	Family	Merritt and Cummins (1996)

2 Only mature organisms can be identified to species.

Taxon	Level[2]	Reference
MOLLUSCS		
Snails	Genus/Species	Frest and Johannes (1999); Jokinen (1992); Burch (1989); Clarke (1981)
Clams		
Pisidium	Genus	Clarke (1981)
Sphaerium	Species	Mackie and Huggins (1983); Clarke (1981)
Musculium	Species	Mackie and Huggins (1983); Clarke (1981)

Organism abundance for each composite sample was calculated on a number/m^2 basis and the number of taxa recorded. In addition, the Shannon-Wiener diversity index, evenness (equitability) and richness values were calculated for each station. High values are generally indicative of a diverse, non-impacted site, whereas low values generally indicate stressed environmental conditions. The Shannon-Wiener diversity index is a measure of the number of species and individuals present at a given location as well as the distribution of those individuals among the various species. Wilhm and Dorris (1968) proposed that benthic macroinvertebrate communities with diversity index values greater than 3 are generally found in unpolluted conditions, whereas communities with values less than 1 are generally found in organically enriched (polluted) conditions. Evenness (equitability) is a component of the diversity index and measures how organisms are distributed among the species present. Richness is another component of diversity, and is dependent on the number of taxa present, e.g., a sample with 50 different taxa would score a much higher richness value than a sample with 5 taxa.

The percent abundance of the key taxonomic groups, e.g., Oligochaeta (worms), Chironomidae (midgefly larvae), Ephemeroptera (mayfly nymphs), Trichoptera (caddisfly larvae), Bivalva (clams), was also determined to provide an indication of the percentage of the benthic macroinvertebrate population at each station that was composed of sensitive (if any) and tolerant groups. For example, high numbers and percentages of oligochaetes and chironomids usually indicate organic enrichment. The percentage of Hydropschidae to total Trichoptera was also determined to indicate the proportion of the trichopteran group that is comprised of the more tolerant hydropsychids, e.g., *Cheumatopsyche* and *Hydropsyche*.

The EPT index, a measure of the diversity of the relatively more sensitive benthic macroinvertebrate groups, Ephemeroptera, Plecoptera (stonefly nymphs) and Trichoptera, was also calculated. This index is the sum of all taxa within these three orders, which generally are the most sensitive to anthropogenic stressors. The EPT index is low to zero when the aquatic environment is moderately to severely impacted, whether due to organic enrichment or other pollutants. An EPT index of 5 to 8 is indicative of acceptable water quality, whereas a higher value reflects good water quality. Although some sites may lack any EPT taxa, it should be noted that this may be the result of habitat type rather than anthropogenic stress.

Two additional biotic indices were computed in order to measure the biological integrity of the benthic communities according to established tolerance values assigned to individual taxa. These are the Hilsenhoff biotic index (HBI) and the BioMAP Water Quality Index (WQI).

The modified HBI was developed to summarize overall pollution tolerance of the benthic macroinvertebrate community (Hilsenhoff, 1982, 1987). It is calculated as the mean tolerance value for all of the individuals collected at each site (i.e., by multiplying the density of each taxa by its tolerance value, summing the values for all taxa and then dividing by the total density).

The HBI values range from 0 to 10, increasing with increasing community tolerance to disturbance, as listed below.

Index Value	**Water Quality**	**Degree of Organic Pollution**
0.00–3.50	Excellent	No apparent organic pollution
3.51–4.50	Very Good	Possible slight organic pollution
4.51–5.50	Good	Some organic pollution
5.51–6.50	Fair	Fairly significant organic pollution
6.51–7.50	Fairly Poor	Significant organic pollution
7.51–8.50	Poor	Very significant organic pollution
8.51–10.0	Very Poor	Severe organic pollution

These categories and associated levels of organic pollution are meant as rough guidelines for thermal impacts. Site-specific evaluations concerning the relative degree of organic pollution may not follow these criteria.

The **Bio**logical **M**onitoring and **A**ssessment **P**rogram (BioMAP) was developed by the MOE to provide a direct measure of water quality for riverine systems throughout southern Ontario (Griffiths, 1993, 1996, 1998; MOEE, 1993b). The calculation is similar to that of the HBI, with some exceptions. The tolerance values run from 0 to 4, with 0 the most tolerant. These values are converted to a function of e (natural log). The ln (x+1) density of each animal is then multiplied by its specific tolerance value. These values are summed for each station and then divided by the sum of the ln (x+1) densities. Water quality is thus expressed as an abundance weighted sensitivity value. All benthic macroinvertebrates in the sample (which have assigned tolerance values) contribute to the estimate of water quality. Unlike the HBI, the sensitivity values contribute proportionally, not arithmetically, to the measure of water quality. Consequently, rare taxa, which are most sensitive to environmental stresses, contribute 54.6 times more to the measure of water quality than do the most tolerant taxa (Griffiths, 1998). Sites with WQI values less than 8 are considered impaired, whereas those greater than 10 are considered unimpaired. Values between 8 and 10 are considered unstable. In unstable watercourses, stress may cause sensitive species to disappear, and the recovery of the benthic macroinvertebrate community may not be possible.

Within-year differences between sites included a one-way ANOVA with a post-hoc test involving comparisons to the reference site (Dunnett Test). The benthic data were also analyzed according to a before-after-control-impact (BACI) analysis (Smith *et al.*, 1993).

The study methodology was reviewed and approved by the MOE, Sarnia/Windsor District Office. A detailed macroinvertebrate sampling survey and analysis was conducted commensurate with the anticipated likelihood of effects. In this case, the thermal plume was predicted to extend along the shore and to be in direct contact with the bottom of the water body over a considerable area. Such a detailed benthic invertebrate study would not be necessary for study scenarios where thermal impacts to the benthic invertebrate community were less likely.

A summary of the comparative assessment of the findings of the pre-operational and operational benthic macroinvertebrate surveys for the BBPS thermal discharge is provided in Section 7.2.

3.2 PEC Fish Survey Plan

The PEC site is an example of emphasis on impacts on important fish resources. The fish survey consisted of initial field studies and assessments and subsequent, more detailed field studies and evaluations based on the initial evaluations. Initially, the PEC fish survey plan involved the collection of fish by gillnetting in the former R.L. Hearn GS Discharge Channel and in the Outer Harbour offshore of the Discharge Channel outlet. Larval tows were also undertaken in the Ship Channel and Outer Harbour (Stantec and SENES, 2003).

As concluded by SENES *et al.* (2003), beyond thermal plume fish attraction, thermal discharges from the proposed PEC were not expected to affect fish community structure or function in the Outer Harbour, except possibly in the case of plant upset conditions during the spring resulting in an increase in cooling water temperatures of up to 10C° (see Section 4.2). This temperature increase may potentially have a positive effect on fish growth and negative effect on spawning activities and embryo survival (see Section 5.4).

Although short-term unusual conditions are unlikely and are expected to have a negligible effect on fish populations, a baseline-monitoring program was recommended. This monitoring program would involve continuous temperature monitoring over the spawning period in nearshore spawning habitat along Cherry Beach and within the embayments of the Outer Headland. This monitoring program, to be developed in consultation with the Toronto and Region Conservation Authority (TRCA), would provide a baseline for post-operational monitoring, particularly during any plant upset conditions.

Moreover, Environment Canada requested that additional fish surveys be undertaken to confirm that the Outer Harbour nearshore in the area of the Discharge Channel provides only foraging and limited nursery habitat for warmwater fish. These surveys were to be conducted during both daylight and nighttime periods during each of the four seasons to determine life-stage fish species composition and density distribution at different water depths and nearby embayment locations.

Initially, a fish habitat survey was undertaken to identify and map areas of similar habitat type (Eakins and Fitchko, 2004). Based on the fish habitat survey findings, the fish survey design was finalized in consultation with the TRCA. After review of the final survey design, Environment Canada suggested that larval and/or embryo surveys be conducted in the spring, including periodic larval tows in the vicinity of the proposed discharge point, identification of gravid females and documentation of spawning or nest building/guarding activities. Environment Canada also suggested that three spring and summer surveys be undertaken, i.e., in early May, mid-June and late July. These suggestions were incorporated into the fish survey design. Due to the timing of the fall fisheries resources survey (mid-December), a winter survey was deemed by the TRCA to be unnecessary.

In addition to mid-December, fish sampling was conducted in early May, mid-June and early August using a variety of gear, including gillnet, minnow trap, seine, larval tow net and/or backpack electrofisher. Experimental gillnets (225 ft x 6 ft) with mesh sizes ranging from 1 to 6 inches were set overnight, as well as during the day, in three different locations within the Outer Harbour for all four surveys. Minnow traps were set at six locations overnight and during the day for all four surveys.

Larval tows using a paired 50-cm diameter Bongo net array with 505-µ mesh were conducted across seven or eight inshore and offshore transects at about 0.5 to 1 m below the water surface during the early May, mid-June and early August surveys.

As part of the four surveys, electrofishing was undertaken with a Smith-Root Model 12B-POW battery-powered backpack electrofisher and dip net at various nearshore locations during the day. An additional electrofishing survey was undertaken at four locations in mid-October when the continuous temperature monitoring loggers were removed for final data downloading (see below).

Seining was undertaken at one location during the early May survey, with the area sampled approximately 80 m long and 6 m wide. The seine dimensions were 54 ft by 4 ft with ¼-inch mesh. This area was also electrofished in mid-December, as well as in August and October.

Captured fish were identified to species, enumerated and released after sampling. Larger adult fish captured by gillnetting were also measured and weighed. Age classes were assigned, i.e., young-of-the-year, juvenile, adult, to each fish based on examination of relative size by an experienced fisheries biologist. Catch-per-unit-effort (CPUE) was calculated for each method of fishing (except larval tows).

As indicated above, continuous water temperature monitoring was undertaken over the warmwater fish spawning period in potential nearshore spawning habitat along Cherry Beach and within the Outer Headland embayments. Temperature monitoring loggers (Onset StowAway TidbiT) were installed in early April at seven locations with temperature measurements every two hours. All seven loggers were relocated, downloaded and reset during the early May fisheries survey and again during the mid-June and early August surveys. The TidbiTs were removed for downloading in mid-October.

Eakins and Fitchko (2004) provide a detailed description of the fish survey findings. An operational fish survey will be undertaken after initiation of PEC operation, anticipated in June 2008.

3.3 Summary

The design of pre-operational survey plans for future proposed facilities with thermal discharges would be dependent on the receiving water baseline conditions, the thermal plume modelling findings, the need for site-specific information, the severity of likely effects and the regulatory requirements. A proponent of a proposed facility with thermal discharges would rely on a qualified fisheries and/or aquatic biology contractor to develop and implement the site-specific field study endorsed by the appropriate government agency.

4.0 THERMAL PLUME MODELLING APPROACHES

When a thermal effluent is discharged to a waterbody, temperature reduction along the plume occurs by dilution and heat loss to the atmosphere (MacLaren, 1975). Dilution occurs by plume entrainment, mixing through jet momentum and natural diffusion. Not to be confused with *intake* entrainment, which is the pumping of water and organisms through the cooling circuit, plume entrainment occurs in the immediate vicinity of a high velocity outfall, where the jet has a significant velocity excess compared to the ambient water. Mixing by momentum is important in the near field to a point where the slowed discharge still has velocity in excess of the ambient water and the temperature is reduced by approximately 20%. In this transition region, both momentum entrainment and natural diffusion contribute to dispersion and dilution. In the far field, natural diffusion is the dominant process together with surface cooling, i.e., heat loss to the atmosphere.

Increasing discharge velocity results in an increase in entrainment of surrounding water (plume entrainment) and promotion of dilution, as well as increase in vertical mixing. If discharge velocity is low, i.e., not characterized as a jet, the effluent would be buoyed directly to the water surface due to thermally induced density gradients.

The pressure of a longshore current flowing transverse to the initial direction of a shore-based discharge results in thermal plume entrainment becoming parallel to the shoreline. This often results in the plume contacting the shore and thus being deprived of dilution water from one side of the plume. As a result, the rate of temperature decline is reduced, thus extending the zone of potential impact across a larger area. In contrast, by locating the outfall offshore and increasing the outfall velocity, this effect can be reduced by limiting the incidence of longshore currents as thermal discharge is directed farther away from shore. The selection of an onshore versus offshore outfall structure should take into account regulatory constraints, structural damage potential, thermal plume configuration, environmental considerations and cost (Fitchko, 2001).

Parameters influencing thermal plumes are summarized below (Silberman and Stefan, 1970):

- effluent characteristics, e.g., flow rate, density (temperature) difference relative to the receiving water, velocity at outlet;
- outlet characteristics, e.g., location, orientation, submergence, shape, size (depth, width);
- flow dynamics, e.g., ambient velocity field (magnitude and direction), wind-induced and other currents, surface waves, free turbulence;
- stratification, and the frequency and duration of wind-induced upwellings and downwellings;
- receiving water geometrical characteristics, e.g., shape, size (width and depth), bottom configuration and roughness near outlet;
- wind, e.g., velocity (magnitude and direction), shear stresses at water surface;

- atmospheric temperature and relative humidity; and
- solar radiation.

Many of these parameters are taken into account during thermal plume modelling. Site-specific baseline limnological/hydrological studies can usually be limited to bathymetry, seasonal current velocity and direction, as well as seasonal water temperature, including delineation of summer stratification conditions in lentic systems. This information, together with power plant design and operational information (e.g., ΔT, flow rate, outlet velocity, outfall design), is requisite to undertake thermal plume modelling.

More than one model should be used to delineate the behaviour and configuration of thermal plumes. Modelling approaches include: a mixing model for the near field (jet and momentum driven), an equilibrium thermal balance model for the far field where the mechanisms are dilution and heat loss to the atmosphere, and three-dimensional (3-D) hydrodynamic models for the far field. Most models can produce both numerical output, such as the numbers in the tables included with each of the following examples, and visualizations. The visualizations are often maps with temperatures in a colour scale (surface maps, depth profile maps, etc.). The visualizations can be given for both static conditions and as videos for fluctuating conditions such as under changing flows. They are especially helpful to both professionals and novices in grasping the spatial extent of plumes.

The Cornell Mixing Zone Expert System (CORMIX, Version 3.2) is a software system developed by Cornell University to predict the mixing characteristics of a discharge into a natural waterbody (Jirka *et al.*, 1996). CORMIX is commonly used in support of regulatory permits and approvals for the assessment of near field mixing of effluent discharges into aquatic environments. This model was used to determine thermal plume configuration for the BBPS and the TCP (under flow conditions in the Welland Canal).

CORMIX was also used during initial screening evaluations for the PEC (Fitchko, 2002). It was not used for subsequent investigations of the PEC thermal plume, since the model was considered inappropriate for use within an embayment, such as the Outer Harbour. CORMIX is commonly used for the delineation of thermal plumes within rivers and open coastal waters. In such environments, the ambient currents are well developed and able to advect the cooling water away from the point of discharge. The algorithms that comprise CORMIX represent this among other processes.

In the case of the Outer Harbour, particularly near the Discharge Channel, the currents are very small and oriented towards the shoreline (as identified through drogue studies). These poorly developed ambient currents are not able to advect the cooling water away from the Discharge Channel. Instead, the discharge itself will induce a current that will create a circulation within the uppermost basin of the Outer Harbour. The cooling water within this circulation pattern will dissipate through heat exchange with the atmosphere and dispersive exchange within remaining areas of the Outer Harbour. The algorithms comprising the equilibrium thermal balance model represent this process. The model accounts for the heating and cooling caused by the discharge of cooling water from the PEC, exchange with water within the Outer Harbour, incident solar radiation, long wave radiation, and evaporative and conductive losses (USACOE, 2000).

The model represents the Outer Harbour as a series of three separate basins. The model assumes that waters circulate within each basin and are exchanged between basins. As a conservative assumption, the cooling water is confined to the epilimnion (i.e., surface to 3 m) and does not mix with the hypolimnion. Moreover, it was conservatively assumed that there was no initial mixing or temperature attenuation within the Discharge Channel.

The predictions from the heat balance equilibrium model were similar to those of CORMIX (Stantec and SENES, 2003).

The MOE undertook preliminary audit modelling for the PEC worst-case scenario (see Section 4.2) using a 3-D model (Delft3D-Flow). The Delft3D-Flow is a hydrodynamic and contaminant transport model, which calculates non-steady flow and various transport phenomena using a curvilinear horizontal grid. The model has the capability to simulate the transport of heat, through a system, via use of appropriate heat flux models (in conjunction with water momentum, continuity and density equations). This enables the model to be used in simulating the generation (and decay) of a temperature-induced, stratified water column structure. In the vertical, it uses the sigma-coordinate system, which divides the water column into a constant number of layers, each representing a certain fraction of the total depth (i.e., between the water surface and the bottom of the waterbody). In the horizontal, a grid of surface cells is overlaid over the study area. For the Outer Harbour, there were 15 layers and over 850 surface cells. Compared to the equilibrium thermal balance approach used by Stantec and SENES (2003), the 3-D modelling results indicated a slightly greater degree of thermal impact within the Outer Harbour (T. Belayneh, MOE, 2003, pers. comm.).

As indicated above, CORMIX was used to model TCP thermal plume discharge into the Welland Canal under most flow conditions (Stantec, 2006). However, during winter, canal flows may decrease to as low as 0 to 1 m^3/s. This low-flow scenario could not be modeled using CORMIX. Instead, a 3-D hydrodynamic model (ECOMSED) was used to simulate the thermal plume (McCorquodale and Georgiou, 2006). ECOMSED is a hydrodynamic and sediment transport model developed by HydroQual, Inc., with the hydrodynamic component the same as the ECOM3D/POM model. The hydrodynamic module is similar to that of the Delft3D-Flow model with a curvilinear-orthogonal grid on the horizontal plane and sigma-coordinates in the vertical direction. The model used had 10 vertical layers and a longitudinal grid of 5 m and lateral grid of 10 m.

For the BANRP, the 3-D RMA10 model was used, which was developed by Professor Ian P. King and Resource Management Associates (RMA) for the U.S. Army Corps of Engineers (Golder, 2005b). The model utilizes an unstructured grid and a Galerkin-based finite element numerical scheme. The model accounts for wind shear stress, gravitational force, turbulent shear stresses, Coriolis force, lake bottom shear stresses, density variations due to temperature changes, and boundary inflow and outflow momentum. The partial differential hydrodynamic equations are solved numerically using a finite element method that can handle flows in complex geometries or boundary conditions. It also maintains a heat budget for every element to account for heat inputs and losses. This heat budget incorporates net short-wave input, long-wave radiation, long-wave back radiation, evaporation and conduction with the atmosphere.

The results of thermal plume modelling for the BBPS, PEC, TCP and BANRP projects are provided in the following sections to provide for project-specific thermal plume effects assessment presented in Chapter 5.0.

4.1 BBPS Thermal Plume Modelling Results

Rodgers (2001) estimated the configuration of the thermal plume that would be discharged from the BBPS under winter, spring/fall and summer conditions. The Detroit River and cooling water discharge conditions considered in the thermal plume modelling are presented in Table 4. The conditions used in the assessment are considered “worst-case,” since they are based on the lower range of flows and conservative values for river water and cooling water discharge temperatures.

Table 4: Detroit River and Discharge Conditions for Modelling of BBPS Thermal Plume

Condition	Season	Load Condition	Ambient Temp. (°C)	Cooling Water Temp. (°C)	Temp. Diff. (°C)	Cooling Water Flow (m^3/s)	River Flow (m^3/s)
1	Winter	Full	1	12	11	13.0	3200
2	Winter	Partial	1	12	11	6.5	3200
3	Spring/Fall	Full	10	21	11	13.0	3600
4	Spring/Fall	Partial	10	21	11	6.5	3600
5	Summer	Full	25	32	7	19.8[3]	4500
6	Summer	Partial	25	32	7	13.0	4500

The results of the thermal plume modelling are presented in Table 5. Based on these results, the cooling water will be in compliance with the PWQOs for temperature under full-load conditions. Based on the modelling, the temperature at the property line (i.e., waterlot) will not increase 10 degrees above ambient and will not exceed a maximum of 30°C.

[3] Tempering water used (see Chapter 6.0).

Table 5: Estimated BBPS Thermal Plume Configuration

Condition	Season	Load Condition	Temp. Diff. in Pipe (C°)	Temp. Diff. at Property Line (C°)	Temp. at Property Line (C°)	Plume Length (m) @ 2°C	Plume Width (m) @ 2°C
1	Winter	Full	11	5.4	6.4	930	95
2	Winter	Partial	11	3.8	4.8	500	60
3	Spring/Fall	Full	11	5.1	15.1	1030	140
4	Spring/Fall	Partial	11	3.9	13.9	590	85
5	Summer	Full	7	3.5	28.5	710	100
6	Summer	Partial	7	3.0	28.0	510	75

The thermal plume will extend along the Canadian shoreline, with the temperature difference within the plume gradually decreasing with increasing distance from the outfall. The temperature difference will be reduced to within two degrees of ambient within approximately 1000 m under all conditions. The width of the thermal plume at two degrees is estimated to be less than 140 m under all conditions.

Based on the thermal plume delineation results, BEAK (2001) concluded that thermal loadings from the BBPS facilities would have a minimal, localized impact on the thermal regime of the Detroit River. Elevated water temperatures in the winter would likely preclude the formation of frazil ice along the shoreline within about 1 km of the discharge. As indicated in the footnote to Table 4, quenching (tempering) water may be used as a mitigative measure in the summer when ambient river water temperatures are 25°C or higher (see Chapter 6.0).

4.2 PEC Thermal Plume Modelling Results

As indicated in Section 2.9, the design of the PEC was intended to limit the ΔT at the head of the Discharge Channel to less than 10C°, specifically with a design ΔT of 5.5C°. As indicated above, the extent of the thermal plume was approximated by Stantec and SENES (2003) using an equilibrium thermal balance model (USACOE, 2000). During the fall, winter and spring when there is no natural thermal stratification, a buoyant plume would spread into the Outer Harbour, with a temperature increase of about 4 and 5 degrees in the uppermost basin of the Outer Harbour under typical winter and spring/fall conditions, respectively (Stantec and SENES, 2003). Based on the unusual operating condition when the change in temperature is 10 degrees, temperatures as high as 19.5°C are predicted during spring/fall in the uppermost basin with temperatures of 18.1 and 14.9°C in the subsequent two basins (see Table 6). For winter conditions, the corresponding temperatures would be 12.4, 10.8 and 6.9°C in the uppermost and subsequent two basins, respectively.

Table 6: Predicted Typical and Worst-case PEC Thermal Plume Scenarios[4]

Temperature	Temperature (ºC)						
	Summer			Spring/Fall		Winter	
	Typical[5]	Worst-case[6]	30ºC Limit	Typical	Worst-case	Typical	Worst-case
Ambient	20.6	26.0	-	10.0	10.0	3.0	3.0
Discharge	20.5	25.0	30.0	15.5	20.0	8.5	13.0
Uppermost Basin 1	20.5	25.1	29.5	15.2	19.5	7.1	12.4
Basin 2	20.6	25.4	28.0	14.4	18.1	6.0	10.8
Basin 3	20.6	25.7	25.0	12.5	14.9	3.2	6.9

During periods of thermal stratification under typical summer conditions and normal plant operations (i.e., ΔT of 5.5 degrees), the water temperature within the Outer Harbour was not expected to change appreciably. Under worst-case summer conditions (i.e., surface water temperatures of 26°C) and normal plant operations, the discharge of once-through cooling waters may reduce the surface water temperature within the uppermost basin of the Outer Harbour to approximately 21°C. This decrease was attributed to the location of the intake and cooler temperatures within the hypolimnion. Based on the unusual operating condition involving a ΔT of 10 degrees, only a minor change (i.e., less than 1°C) would occur in the water temperature within the Outer Harbour as long as cooler waters within the hypolimnion of the Ship Channel are drawn by the PEC intake (see Table 6).

Based on an unlikely scenario when only surface waters are drawn during an extremely warm summer and a maximum cooling water temperature limit of 30°C stipulated in the C of A, the "worst-case" surface water temperatures within the uppermost basin would be 29.5°C with temperatures of 28 and 25°C in the subsequent two basins. As indicated above, this extreme condition is unlikely since the water intake was designed to draw cooler waters from the hypolimnion during the summer thermal stratification period, thereby ensuring a cooler discharge temperature.

Based on the thermal plume modelling results, thermal loadings from the PEC would have a minimal impact on the thermal regime of the Outer Harbour. Elevated water temperatures in the winter would likely preclude the formation of ice in the Outer Harbour.

As indicated in Section 2.1, a CWQG has been established for thermal stratification, i.e., "thermal additions to receiving waters should be such that thermal stratification and subsequent turnover dates are not altered from those existing prior to the addition of heat

4 Source: Stantec and SENES (2003).

5 Typical = 5.5º ΔT.

6 Worst-case = 10º ΔT.

from artificial origins”. In general, during the operation of the former R.L. Hearn GS, the thermal profiles in the Outer Harbour were similar to those found immediately outside the harbour. The main exceptions occurred near the outfall, where under some conditions, the thermal plume remained in contact with the bottom for a distance of approximately 500 m. Based on the temperature profile surveys undertaken in 2002, the Discharge Channel was thermally stratified in June and isothermal in August (Stantec and SENES, 2003). The Outer Harbour was vertically mixed during the fall, winter and spring periods. During the summer, the thermocline varied in depth from 3 to 8 m depending on time and storm events. Since the discharge flow and ΔT under normal operating conditions of the proposed PEC were expected to be smaller than those of the former R.L. Hearn GS, the cooling water discharge was not expected to affect thermal stratification and subsequent turnover rates of the Outer Harbour. For example, discharge flows for the former R.L. Hearn GS and PEC are 35 and 17 m^3/s, respectively, whereas ΔTs under normal operating conditions are 12.4 and 5.5 degrees, respectively.

4.3 TCP Thermal Plume Modelling Results

As indicated in Section 2.9, the temperature rise in the cooling water through the proposed TCP is expected to be less than 10 degrees, with a maximum temperature at the outfall maintained at or below 32.0°C (with the use of quenching water, as necessary). The heated water will be discharged directly into the Welland Canal.

Flow in the Welland Canal is regulated by the SLSMC (Fitchko, 2006a). Operational curves were applied to adjust the amount of flow through the canal locks, which is dependent upon ship traffic. Currently, commercial ship traffic of slightly more than 3000 ships per year is being experienced, which equates to ambient canal flows of approximately 30 m^3/s during the canal’s navigational period (L. Malone, SLSMC, 2006, pers. comm.). During the non-navigational season, which generally runs from late December to late March, an average flow of 12 m^3/s is maintained through Lock 7, located approximately 1.5 km downstream. However, during winter, flow is variable, dependent upon power demand on the SLSMC hydroelectric facilities. During winter, flows may decrease to as low as 0 to 1 m^3/s (L. Malone, SLSMC, 2006, pers. comm.).

Temperatures in the Welland Canal at the existing Abitibi Consolidated Company of Canada (ACCC) intake located approximately 50 m upstream of the TCP outfall were monitored for several days in mid-July. A 75th percentile water temperature of 23.5°C was calculated from the observed data and applied to the thermal plume modelling of the TCP cooling water discharge. A winter temperature of 4°C, which represents the maximum density of water, was applied to the winter thermal plume modelling.

To gain an understanding of the potential extent of the thermal discharge, three main scenarios were modeled (Stantec, 2006):

1) Maximum design effluent flows of 4.2 m^3/s, summer canal flows of 30 m^3/s, with an ambient temperature of 23.5°C, and a discharge temperature of 33.5°C.

2) Low effluent flows of 1 to 3 m^3/s, summer canal flows of 30 m^3/s, with an ambient temperature of 23.5°C, and a discharge temperature of 33.5°C.

3) Maximum design effluent flows of 4.2 m^3/s, average winter canal flows of 12 m^3/s, with an ambient temperature of 4°C, and a discharge temperature of 14°C.

Scenarios 1 and 2 represented worst-case summer scenarios, i.e., with a discharge temperature of 33.5°C, since the TCP would be using quenching water during the summer period of higher ambient water temperatures to ensure that cooling water discharge temperature does not exceed 32.0°C.

For the maximum design discharge of 4.2 m^3/s and ambient summer flows of 30 m^3/s, no upstream migration was predicted, and the plume temperature was predicted to fall below 5 degrees above ambient within 100 m downstream of the discharge. The buoyant plume was predicted to occupy a layer of 3-m depth. With discharge velocities of approximately 0.5 m/s, the momentum of this plume discouraged upstream spreading.

However, when lower discharges were modeled (i.e., 1 to 3 m^3/s), upstream spreading was produced due to the lower mixing potential of these lower magnitude discharges. Upstream spreading may reach a distance of 100 to 150 m upstream for a discharge of 1 m^3/s, although the plume would be limited to less than 1 m in thickness upstream of the discharge location. For a discharge of 1 m^3/s, the plume temperature was predicted to fall to two degrees above ambient within 100 to 150 m of the proposed outfall.

For the maximum design discharge of 4.2 m^3/s and ambient winter flows of 12 m^3/s, no upstream migration was predicted, and the plume temperature was predicted to fall below 4 degrees above ambient within 100 m downstream of the discharge, and below 3 degrees above ambient within 300 to 350 m downstream of the discharge. The plume was predicted to occupy a surface layer depth of 1 m or less at these downstream distances.

The thermal plume modelling results demonstrated that temperature increases within the Welland Canal as a result of TCP discharge would generally influence the surface layer of the canal, within a 1 to 3 m depth. Upstream spreading of discharge water may be produced at lower exit velocities, within a depth of 0 to 1 m of the surface, but not reach beyond a distance of 150 m upstream, potentially affecting the ACCC intake. At the TCP intake located 150 m upstream with an intake depth of 3 m below the surface, potential temperature increases of 0 to 1 degree were predicted.

As indicated above, canal water flow may decrease to as low as 0 to 1 m^3/s in the winter. This low-flow scenario cannot be modeled using CORMIX. Instead, a 3-D hydrodynamic model was used to simulate the thermal plume (McCorquodale and Georgiou, 2006).

The modelling results indicated that the thermal discharge went through a phase of rapid mixing due to momentum and buoyancy. The temperature increase above ambient temperature decreases from 10 degrees to approximately 3 degrees in the near-field zone, providing a dilution of about 3:1. Outside of the near-field zone, the plume becomes strongly stratified, with the depth of stratification varying from 8 to 2.5 m. There is an upstream progression of the leading edge of the thermal plume for an ambient flow of 0 to 1 m^3/s. At 6 h, there is a 3 degree increase in the surface temperature at 300 m upstream of the discharge. At 10 h, this front reaches 500 m upstream of the discharge with substantial stratification at this location at 12 h. For a 12-h operational period, the design discharge of 4.2 m^3/s was predicted to migrate 500 m upstream, creating a 2-m thick plume at the intake with temperatures 3 degrees above ambient. The re-entrainment of heated water would increase ambient temperatures at the intake by approximately 1 degree. The plume would reach a downstream distance of approximately 400 m, and a surface thickness of about 1 m, with plume temperatures approximately 3 degrees above ambient.

As indicated in Section 2.9.3, the proposed TCP will operate in cogen mode for approximately 3500 h/yr (40% of the time) and will be capable of starting up and shutting down up to two times daily. Based on the 3-D modelling results, McCorquodale and Georgiou (2006) concluded that an intake 500 m upstream of the discharge could operate 6 to 10 h without significant re-circulation of heated water due to upstream migration of the thermal plume. Moreover, a low-level intake with a skimmer could delay the entrainment of heated water into the intake.

Additional one-dimensional heat balance and 3-D (ECOMSED) modelling was undertaken to depict a number of TCP operating duration scenarios under low winter flow conditions and the requisite dissipation periods after plant shutdown to permit restartup (J. McCorquodale, University of New Orleans, 2007, pers. comm.). The C of A Special Operating Requirements (see Section 2.9.3) reflect the modelling findings.

A degree of uncertainty is associated with the thermal plume modelling results, due to several factors such as the effects of wind and ship traffic on the configuration of the plume, which cannot be simulated accurately. For example, during the navigation season, the buoyant thermal plume can be expected to be mixed into deeper water during vessel passage. Between June and September 2005, monthly vessel transits per day ranged from 12.7 to 13.1.

4.4 BANRP Thermal Plume Modelling Results

Previous thermal plume surveys and modelling had indicated that the Bruce A and Bruce B plumes have little to no interaction during warm weather conditions (WWC) and some interaction, i.e., areal overlap (estimated to be less than 8% of the time), during cold weather conditions (CWC) (Golder, 2005a,b). WWC occur when ambient lake water temperature is greater than 4°C, i.e., generally in the spring, summer and fall. Under WWC, the thermal plume will always be warmer than the ambient lake water and therefore buoyant. Under WWC, plumes associated with Bruce A were predominantly directed alongshore to the northeast. Currents moving to the southwest appeared to direct the plumes offshore to the west, which turned to the southwest some distance offshore. This type of plume does not return to shore. Plumes travelling alongshore to the northeast were essentially isothermal with depth beyond about 2 km from the discharge channel, extending alongshore beyond 23 km from Bruce A and offshore up to 3 km. The alongshore plumes extended from surface to lake bottom throughout their length. Offshore floating plumes had a thickness of 1 to 4 m.

CWC occur when ambient lake water temperature is less than 4°C, i.e., during the winter. Under CWC, the thermal plume would be buoyant in the vicinity of the outlet channel. When the buoyant plume temperature decreases to 4°C, the plume could sink during calm winter days and under ice cover conditions. Under CWC, plumes associated with Bruce A also appeared to be predominantly alongshore to the northeast. The sinking plumes to the northeast extended along the lake bottom more than 10 km from the discharge channel and up to 3 km offshore. An offshore plume might extend along the lake bottom up to 8 km offshore from the discharge channel. Sinking plumes can extend more than five times farther offshore from the station as compared to the surface plumes.

Using the 3-D RMA10 model, Golder (2005b) estimated areal extents of thermal plumes (i.e., the area of the thermal plume at the surface) for the future conditions (Bruce A with four units in operation, Bruce B with four units in operation) for both WWC and CWC. The shape of a thermal plume at any given time was found to be highly variable and affected by a number of factors, including the prevailing currents and ambient water temperature.

As indicated in Table 7, the maximum areal extent of the combined thermal plumes from Bruce A (four units) and Bruce B (four units) was estimated to be approximately 6700 ha, based on the criterion of 2 degrees above the ambient and in consideration of both WWC and CWC. This is an increase of approximately 50% from baseline conditions in 2004 (Bruce A with two units operational and Bruce B with four units operational).

Table 7: Estimated Areal Extents of Bruce A and B Thermal Plumes

Temperature Excess Above the Ambient (°C)	Estimates for Two Bruce A Units in Operation		Estimates for Four Bruce A Units in Operation	
	Maximum Limits (ha)	Quartile Limits[7] (ha)	Maximum Limits (ha)	Quartile Limits[7] (ha)
2	4400+	870	6700+	1500
4	2500	390	3800+	610
6	1200	167	1800+	250
8	500	50	740	70

The area affected by the combined Bruce A and Bruce B plume at any one given time is estimated to be in the range of 70 to 3600 ha with an average plume size of 1250 ha (see Table 8). This is an increase of approximately 50%. With the BANRP operational, the thermal plume is estimated to be 7 to 14 times larger than the Bruce B thermal plume at any given time.

[7] 25% probability of occurrence.

Table 8: Existing and Predicted Bruce A and B Thermal Plume Sizes

Site	Minimum (ha)	Average (ha)	Maximum (ha)
Baseline (Bruce A with two units operational)			
Bruce A	40	850	2400
Bruce B	10	50	250
Combined	50	875	2500
Future (Bruce A with four units operational)			
Bruce A	60	1200	3400
Bruce B	10	50	250
Combined	70	1250	3600

5.0 ASSESSMENT OF THERMAL EFFECTS ON BIOTA

5.1 Background

All aspects of metabolism of organisms, e.g., photosynthesis, respiration, growth, etc., are more or less dependent upon temperature. Increased temperature results in increased rates of metabolism and respiration, as well as increased activity and food consumption. Temperature also affects reproduction and the longevity of organisms. The nature of the dependence varies among different species and between organisms with different past histories and genetic composition. For many processes in the normal range of temperature, rates increase by two or three times per 5 to 6 degree temperature rise.

Individual organisms and species show the ability to adapt or acclimate to seasonal changes in temperature or to ranges of temperature shifts that they may encounter when the acclimation period extends for an adequate length of time. As a result of acclimation, the impact of temperature changes is mollified over some range to which the organism becomes “adapted.”

The upper lethal temperature of organisms has been found to increase as acclimation temperature rises until an ultimate upper lethal temperature is reached (Hart, 1947). This ultimate upper lethal temperature represents the breaking point between the highest temperatures to which a fish can be acclimated and the lowest of the upper extreme temperatures that will kill a warm-acclimated fish (Coutant, 1977b). Above the upper lethal temperature, survival depends not only on the temperature but also on the duration of the exposure, with mortality occurring more rapidly with increasing temperatures. Thus, organisms respond to extreme high (and low) temperatures in a manner similar to the dose-response for any other toxicant. Lethal temperatures are usually defined as those for 96-hr or one week survival of 50% of the population. Below the lethal temperatures, fish can survive wide temperature changes.

Increased temperature can also have an adverse effect when combined synergistically with pollutants. With increasing temperature, toxicities of some contaminants are increased and the resistance to disease lowered.

Fish species are generally grouped into guilds with each guild having different temperature requirements to function properly. The warm-water guild generally survives well in water temperatures up to 30°C. The cold-water guild usually survives well in water temperatures below 20°C. The intermediate cool-water guild survives well in water temperatures up to the mid-20s. In general, a large number of species survives well in water temperatures below 30ºC, with some species approaching the upper limits of their temperature tolerance. Between 30 and 40ºC, the number of species that can survive declines rapidly, and very few aquatic organisms survive water temperatures warmer than 40ºC.

Numerous laboratory or theoretical studies have suggested that thermal discharges have a potential to negatively influence the aquatic environment. Most field studies, however, have demonstrated that any negative effect on freshwater communities or water use is localized and dependent on the specific physical characteristics of the site and the groups of aquatic communities that normally inhabit it.

The siting and design of a new power plant can be the most important environmental consideration in minimizing thermal plume effects (Dames and Moore, 1979), taking into account technical (engineering) and cost considerations. The location of the thermal discharge outfall should take into account biological productivity, unique ecological communities, habitat utilization, and the presence of valued ecosystem components (VECs) and species at risk.

A summary of thermal plume effects on phytoplankton, attached algae, aquatic vegetation, zooplankton, benthic invertebrates, fish, aquatic avifauna and water uses is provided below. This review is not comprehensive. More detailed studies should be conducted for site-specific studies.

5.1.1 Phytoplankton

Research on phytoplankton responses to thermal discharges has been conducted in the past. These studies indicated that at low water temperatures (i.e., 0 to 10°C), a rise in temperature led to an increase in productivity (Coutant, 1970a). At water temperatures typical of summer conditions (i.e., 15 to 20°C, slight increases in water temperature increased productivity as well; however, larger increases (i.e., above 5.6 degrees) led to a depression of primary productivity. The larger the temperature increase above 5.6 degrees, the greater the depression of photosynthesis that resulted.

McKee *et al.* (2003) reported that phytoplankton exposure to a 3 degree increase above ambient temperature, continuously over a 2-y period (first treatment) or during the summer only (second treatment), did not significantly affect chlorophyll *a* concentrations or total algal biovolume.

Noton (1972) detected significantly greater phytoplankton production in the surface thermal plume discharges of the WPP and SPP in Lake Wabamun, Alberta, than at the mid-lake control. There were no differences at water depths of 0.5 and 1 m. In contrast, Nursall and Gallup (1971) reported that during the summer, total phytoplankton standing crop (cells/mL) was almost 33% greater at a control station on Lake Wabamun than in the WPP thermal plume discharge station. This lower production was attributed to the presence of dense growths of aquatic macrophytes, which competitively inhibited the production of phytoplankton. Hasler and Jones (1949) have demonstrated the antagonistic action of aquatic macrophytes on algae and rotifers. However, during the winter, phytoplankton standing crop was higher at the ice-free WPP thermal plume discharge station than at the ice- and snow-covered control station.

When temperatures rise above 35°C, blue-green algae often become dominant if this high temperature is maintained for fairly long periods of time (Patrick, 1969). Green algae increase if the temperature is between 32.5 and 35°C for several weeks, whereas below these temperatures diatoms are usually dominant.

In general, thermal effects on phytoplankton assemblages in receiving waters have been small or indistinguishable from normal seasonal or spatial changes. This is because most well-designed thermal discharge plumes expose planktonic organisms to only transient, short-duration temperature changes that are neither lethal nor cause lasting changes in production. Reproductive rates of phytoplankton are so rapid that any cells lost are rapidly replaced. Ecosystem-scale changes in plankton communities have been rare and usually occur on smaller bodies of water. Although organic production of the phytoplankton assemblage may increase somewhat, the additional biomass does not generally increase above nuisance levels. For these reasons, most assessments of thermal discharges do not consider phytoplankton to be a biotic category that commands a great deal of attention.

Thermal discharges may contain chlorine used to suppress biofouling organisms that grow on condenser tubes, including bacteria, fungi and invertebrates such as the zebra mussel. Environmental analyses need to consider both the thermal and biocide toxicities.

5.1.2 Attached Algae

Hickman (1974) determined that epipelic algal standing crops were increased by WPP thermal discharge to Lake Wabamun particularly in and just outside the discharge canal. This increase was due to *Oscillatoria amoena* and *O. borneti* in the discharge canal that provided the algal inoculum to the heated water of the lake. The species composition of the diatoms was similar at all stations except in the discharge canal, where the number of diatom species was reduced. *Navicula cuspidata* was the dominant species in the discharge canal in the summer, where water temperatures of 31°C were recorded. The heated effluent had no effect upon the standing crop or species composition of the epipsammon.

Kirby and Dunford (1981) reported that the operation of the Nanticoke GS has resulted in an accelerated early-season growth of attached algae and a change in species composition in the immediate vicinity of the station, likely due to the discharge of heated waters and the creation of a sheltered bay (the discharge channel), respectively. McKinley (1982) reported that the low standing crop of attached algae, primarily *Cladophora*, at the Nanticoke GS discharge was due to higher current velocities resulting in ongoing removal (tearing) of the filamentous alga by the increased frictional shear force. Periodic high water temperature approaching 25°C, which is considered to be limiting to growth for *Cladophora* (Storr and Sweeney, 1971), may have periodically inhibited algal growth adding to the low algal standing crop.

Drown *et al.* (1974) reported that Lake Superior periphytic diatoms were generally not greatly altered by temperature increases (ΔT of 10 to 12 degrees) based on percent composition of the entire assemblage. In general, green algae were favoured by increased temperatures, particularly the green filamentous algae, *Mougeotia* and *Zygnema*. However, the green alga, *Utothrix zonata*, was inhibited by the warmer water conditions.

Unlike phytoplankton, attached algae of discharge canals and near-field zones of discharges are exposed to thermal plumes for extended periods of time. Attached algal assemblages of receiving waters have not been observed to change substantially unless the water temperatures exceed 32°C. There is a trend at these temperatures for blue-green algae to dominate and occasionally develop growths at nuisance levels. The nuisance blue-greens are not eaten by most macroinvertebrates, so excess algal biomass in the near field can be washed out and cause organic enrichment of the far field zones. Attached algae in discharge canals and near-field zones are also vulnerable to chlorine or other biocides, which complicate evaluation of purely thermal effects.

5.1.3 Aquatic Vegetation

Rooney and Kalff (1991) reported that inter-annual fluctuations in temperature have an effect on the distribution and biomass of aquatic macrophyte communities. Early season warm temperatures (1998) resulted in much deeper macrophyte colonization (25 to 170%), an average of 300 g/m^2 greater wet biomass and a 45 to 1160% increase in whole lake biomass in four lakes in the Eastern Townships of Quebec when compared to a cooler 1997.

McKee *et al.* (2002, 2003) reported that an increase of 3 degrees above control temperature in microcosms over a 2-y period did not affect total macrophyte abundance. However, the proportion of each community made up by curly waterweed (*Lagarosiphon major*) and its growth rate increased under 2-y continuous warming (McKee *et al.*, 2002). Warming did not significantly influence the abundance or growth rate of western waterweed (*Elodea nuttallii*). Flowering of floating-leaf pondweed (*Potamogeton natans*) occurred earlier in the season under continuous warming and floating leaf surface area increased under continuous warming and summer-only warming.

Anderson (1969) determined that clasping-leaf pondweed (*Potamogeton* (*perfolialis*) *Richardsonii*) has a high thermal tolerance. This species is capable of physiological adjustment to higher temperatures as the leaf matures, since only older leaves tended to respire less at the elevated temperatures. Death of plant material occurred at 45°C.

Stanley and Naylor (1972) reported that Eurasian water milfoil (*Myriophyllum spicatum*) had a high temperature optimum for photosynthesis. Net carbon dioxide uptake increased with temperature to and including 35°C. It was concluded that this species would have an advantage in adapting to environments with higher temperatures.

Barko and Smart (1981) reported that higher temperatures stimulated growth in large-flowered (Brazilian) waterweed (*Egeria densa*), hydrilla (*Hydrilla verticullata*) and Eurasian water milfoil. However, there was a compression of the growth cycle with a more rapid seasonal progression of senescence. Although shoot biomass in large-flowered waterweed was not appreciably influenced by temperatures between 16 and 28°C, it was significantly reduced at 32°C, the maximum test exposure temperature. Shoot biomass in hydrilla increased in a stepwise fashion between 16 and 32°C. Van *et al.* (1976) reported an optimum temperature of 36.5°C for photosynthesis in hydrilla. There was an absence of a discernible trend in shoot biomass of Eurasian water milfoil, as increase in biomass production at higher temperatures was coupled with maximum sloughing of shoot fragments.

In a similar experiment, Barko *et al.* (1982) demonstrated that shoot biomass of common waterweed (*Elodea* (*Anacharis*) *canadensis*) and wild celery (*Vallisneria americana*) increased with increasing temperature to 28°C. For knotted pondweed (*Potamogeton* (*americanus*) *nodosus*), shoot biomass continued to increase at the maximum exposure temperature of 32°C.

A few studies have been undertaken to assess the effect of power plant thermal discharge on aquatic macrophyte communities. For example, Parker *et al.* (1973) found that several species of aquatic macrophytes were eliminated in a portion of a Savannah River Plant cooling pond where water temperatures exceeded 45ºC most of the year. Moreover, the relative abundance of the remaining species was considerably lower compared to that in a nearby reference pond area. Grace and Tilly (1977) found that dry weights of common water-nymph (*Najas guadalupensis*) increased six-fold from the cold to warm site and ten-fold from warm to hot site. They also found that at the warm station Eurasian water milfoil was more than three times abundant than at the cold station; however, this species was greatly reduced in the hot station.

A number of studies have been undertaken on the effects of thermal discharges on the aquatic macrophyte communities in Lake Wabamun. For example, Allen and Gorham (1972) reported that step-wise increases in maximum generating capacity of the WPP from 70 megawatts (MW) in 1952 to 600 MW in 1968 and associated thermal discharge volume resulted in the invasion and subsequent dominance by common waterweed in that portion of Lake Wabamun affected by thermal effluent, displacing the northern water milfoil (*Myriophyllum* (*exalbescens*) *sibiricum*)–stonewort (*Chara globularis*) community. This community was still present in other parts of the lake including the discharge area of the SPP prior to its commissioning of the SPP in October 1970. By July 1971, northern water milfoil growth was retarded in the SPP discharge canal, and stonewort was eliminated, replaced by common waterweed. Peak ΔT for the SPP was 11 and 22 degrees in the summer and winter, respectively. In addition to increased temperature, it was postulated that increased turbidity and siltation during facility construction also contributed to the demise of stonewort. Within the SPP discharge canal, there was subsequently a gradual elimination of common waterweed and northern water milfoil, with a pronounced increase in sago pondweed (*Potamogeton pectinatus*), which is adapted to swift flowing currents. A similar rapid colonization by sago pondweed occurred in the WPP discharge canal when it was extended in 1961.

Haag and Gorham (1977) and Rasmussen (1982) reported higher submerged aquatic macrophyte production in the area of Lake Wabamun affected by WPP thermal discharge than in the unheated area. Common waterweed dominated early spring production in the heated area because of the large standing crop in this ice-free area overwinter (Haag and Gorham, 1977). The heated area had greater amounts of plant detritus on the sediment than the unheated area (Rasmussen, 1982). Haag (1983) reported that seedling emergence in sediments collected from the WPP discharge canal was four times higher than any other sediment collected from Lake Wabamun. Stem fragments of common waterweed were produced due to wave action and spread throughout the lake, where they established during the growing season but later declined (Haag and Gorham, 1977). It was concluded that localized changes in species composition in the heated areas of Lake Wabamun have not affected the remainder of the lake.

Aquatic macrophytes need to be considered as habitat formers for macroinvertebrates and fish in addition to consideration in their own right. They also need to be considered as potential nuisance growths where water uses require open water or where waters choked with their growths can be detrimental to other aquatic life. Like attached algae, they may experience temperature changes over long durations, especially in near-field areas. Their positive or nuisance value needs to be weighed in light of the agreed-upon water use designations for the water body.

5.1.4 Zooplankton

Reproduction and growth of zooplankton are strongly regulated by temperature. Finesinger (1926) reported that egg production of the rotifer (*Lecane* (*Distyla*) *inermis*) was optimal between 22.3 and 27°C, with lower egg production occurring at 35°C. The organisms exposed to temperatures between 27 and 35°C were much more active and matured more quickly. Brown (1929) reported lethal temperatures ranging from 40 to 50°C for a number of cladoceran species based on one-minute exposure. When exposed to slowly increasing temperatures, the lethal temperature ranged from 35 to 42°C.

Garton *et al.* (1990) reported 12-h upper LT_{50}s of 28 and 26.2°C for the predatory cladocerans *Leptodora kindti* and *Bythotrephes cederstoemi*, respectively, based on an ambient temperature of 25.0°C. For lower ambient temperatures of 10.5 and 7.1°C, the upper LT_{50}s for *L. kindti* and *B. cederstroemi* were 24.7 and 25.9°C, respectively.

Carlson (1974) demonstrated that the cladoceran, *Scapholeberis kingi* had a very high heat tolerance, e.g. 12-, 24-, 48- and 72-h TLms of 37, 36, 34 and 33°C, respectively. This species also has a high tolerance to sudden temperature increases or “heat shock”, e.g., 50% survived for more than 24 h after an immediate temperature change of 21°C, but not exceeding a temperature of 38°C. In addition, when temperature was increased by 5 and 7 degrees, young produced per day increased by 18% and 102%, respectively, compared to ambient temperatures (21 to 22°C). However, reproduction rate decreased by 32% when temperature was increased by 14 degrees.

Laboratory studies on the movement of the copepod, *Diaptomus sanguineus*, and cladoceran, *Daphnia parvula*, have shown that with slight temperature increases (2°C), all moved to deeper, cooler water (Gehrs, 1974).

Of the major zooplankton groups, copepods are more sensitive to thermal increases than rotifers and cladocerans (Kititsina, 1973). For example, Fenlon *et al.* (1971) and McNaught and Fenlon (1972) reported that additions of heat to the Lake Ontario ecosystem at Nine

Mile Point increased the standing crop of *Bosmina* spp. (25 times) and *Daphnia retrocurva* (1.2 times) in the overall study area. Adjacent to the outfall, these same populations increased 123.8 and 2.4 times, respectively. However, the abundance of the copepod, *Cyclops bicuspidatus*, had decreased in the vicinity of Nine Mile Point. At the same time, primary production was not significantly affected due to the increased zooplankton populations.

Dunstall *et al.* (1985) reported that rotifer composition and abundance was somewhat altered in the immediate discharge area of the Lennox GS on the Bay of Quinte, Lake Ontario. Again, no change in abundance of the dominant species of phytoplankton was evident. The composition of crustacean zooplankton in the immediate discharge area differed from that in the cooling water, although both assemblages were dominated by *Diacyclops bicuspidatus thomasi*. Dunstall *et al.* (1985) concluded that the transient nature of plankton communities in lake water adjacent to Lennox GS appears to be influenced primarily by the mixing of disparate water masses, while the short-term effects of station operation were influential primarily in the near-discharge area.

Evans (1981) reported that zooplankton were most abundant in the warmest waters of the thermal plume of the Donald C. Cook Nuclear Power Plant on Lake Michigan, with the region of high densities extending over an approximate area of 0.2 to 0.3 km^2.

Based on nine years of observation of protozoan communities in the Potomac and Savannah rivers, there was no evidence that indicated the protozoan communities had been degraded by small gradual temperature increases resulting from the discharge of heated wastewaters (Cairns, 1969).

Like phytoplankton, zooplankton organisms generally experience brief and transient thermal changes as they are entrained in the plume. These transient exposures are generally not sufficiently long to be lethal, even when discharge temperatures are high. However, prolonged warming or repeated exposures when the receiving water body is small could cause significant changes in zooplankton communities.

5.1.5 Benthic Macroinvertebrates

Ward and Stanford (1982) and Sweeney (1984) provide reviews on the role of temperature in the life history of aquatic insects including such processes as incubation period, hatching success of eggs, dormancy, growth and maturation, and emergence.

McKee *et al.* (2003) reported that a 3 degree increase above control temperature in microcosms over a 2-y period resulted in increased numbers of macrophyte-associated invertebrates (gastropods and ostracods).

Lush (1981) reported that thermal inputs from the Pickering Nuclear Generating Station (NGS) to the littoral zone of Lake Ontario resulted in small, temporally and spatially localized changes in macroinvertebrate populations.

The abundance of some macroinvertebrate species can increase substantially in areas of thermal discharge. For example, the heated area of Lake Wabamun had a higher standing crop of benthic macroinvertebrates than the unheated area (Rasmussen, 1982). Species composition of the benthic macroinvertebrate community was also much different in the heated area. The warmest areas supported a community dominated by tubificid oligochaetes (*Limnodrilus hoffmeisteri* and *Tubifex tubifex*), whereas the moderately heated areas supported a community dominated by large *Chironomus* species (*C. plumosus* and *C. atroviridis*). Benthos of the unheated parts of the lake consisted mainly of smaller chironomid species such as *C. maturus*, *C.* (cf.) *staegeri, Polypedilum nubeculosum, Cladotanytarsus* spp. and *Tanytarsus* spp.

Moreover, Sankurathri and Holmes (1976) reported that in the WPP thermal plume area, the rate of egg development and growth of the snail *Physa gyrina* increased with continuous reproductive activity throughout the year. As a result, together with the increased period of growth of aquatic macrophytes, there were increased densities of *P. gyrina* in the heated area during the summer.

Storr and Schlenker (1974) found that within one year of operation, the number of amphipods (*Gammarus* spp.) increased by 200% in the vicinity of the Nine Mile Point Nuclear Generating Station (NGS) on Lake Ontario. Dahlberg and Conyers (1974) reported similar results for aquatic insects and most other macroinvertebrate groups for waters 4.5 to 12 degrees warmer than ambient temperatures.

Alston *et al.* (1978) compared benthic macroinvertebrate communities in three artificial channels: one receiving unheated river water (control) with the highest average weekly temperature of 30.8°C; a thermally stratified channel receiving both unheated water and heated effluent; and one receiving only heated water with the highest average weekly temperature of 35.5°C. During most periods, oligochaetes and chironomid larvae were the dominant taxa. The channel receiving only heated water had the lowest species diversity, with the control having the highest diversity. Moreover, fewer mayflies were present in the heated water channel compared to the other two channels.

Based on a study of the effects of thermal effluents from a power plant on the Wabush River, Gammon (1973) reported that the numbers of Trichopteran larvae increased in zones with increased temperature in June and August and decreased in July. Chironomids decreased slightly in thermally enriched zones. The response of Ephemeroptera was more varied. *Stenonema* consistently increased in density in the warmer zones, whereas *Tricorythodes* decreased in thermally elevated stations. *Baetis* and *Isonychia* also generally increased in numbers in the warmer zones.

Reductions in benthic macroinvertebrate species diversity, densities and biomass have also been documented. For example, Benda and Proffitt (1974) reported that heated water from a power plant on the White River at Petersburg, Indiana, depressed the populations of caddisfly larvae, mayfly nymphs and other invertebrates in the discharge area when temperatures ranged from 31 to 39.4°C; however, these taxa recovered less than 550 m below the discharge. Chironomids, especially *Glyptotendipes lobiferus*, were the most frequently collected invertebrates in the heated section and were not numerous in the unheated section. The most productive macroinvertebrate sampling location occurred about 90 m downstream of the discharge canal mouth. The maximum temperature reached at this station was 33°C. The least productive stations were the two within the discharge canal, where the temperature averaged 6 degrees above ambient and attained maximum temperatures of 39.4°C.

Howell and Gentry (1974) also reported decreased diversity of benthic macroinvertebrate communities in a stream affected by heated effluents from the Savannah River Plant near Aiken, South Carolina. Almost 96% of the individuals collected were from two species, i.e., a corixid backswimmer and a large chironomid, *Chironomus* sp.

Nichols (1981) reported that only the oligochaete populations in the immediate vicinity (within 100 m) of the Oconee Nuclear Station discharge outlet in the Keowee Reservoir in South Carolina were affected by the thermal discharge. Average standing crop and total number of species were highest at this location, and seasonal fluctuations were significant.

Temperatures of 35°C or more in a power plant discharge channel to Baldwin Lake, a closed-cycle cooling reservoir, resulted in the virtual elimination of chironomids with an annual mean population density of less than 110/m^2 (Parkin and Stahl, 1981). With temperatures up to 32°C in the main basin, the annual mean population density of chironomids was 1037/m^2. Moreover, the elevated temperatures in the main basin resulted in increased numbers of generations of *Tanypus stellatus* from the usual two to three or four.

Wellborn and Robinson (1996) investigated the effect of a power plant thermal discharge on the benthic macroinvertebrate communities in a central Texas reservoir by comparing those in a 60-ha cooling pond directly receiving the effluent to an area of the reservoir relatively unaffected. Temperature of the pond averaged 7.2 degrees higher than the reservoir site. The abundance of Trichoptera was always lower in the cooling pond than in the main reservoir. Chironomids and amphipods were generally lower in abundance in the cooling pond, except for one sampling day in the winter, when their abundance was substantially higher than the main reservoir. In addition, the cooling pond achieved such high temperatures (40 to 42°C) in the summer that all benthic macroinvertebrates were eliminated. Although taxa recolonized the pond after the summer defaunation, with some taxa briefly obtaining high population levels, most taxa maintained lower population levels in the pond than the main reservoir throughout the winter.

Cole and Kelly (1978) reported that benthic macroinvertebrate abundance, diversity and mean individual size declined in the discharge canal of the Monroe Power Plant and an adjacent heated area. However, these decreases were related to intense fish use of the thermal discharge as a feeding habitat due to increased benthic production.

Based on laboratory test findings, Nebeker (1971) reported premature emergence of insect adults when exposed to elevated water temperatures during the winter months when air temperatures could be lethal. Mattice and Dye (1978) reported increased growth and earlier emergence of the mayfly, *Hexagenia bilineata*, in the discharge cove of the Kingston Steam Plant on the Clinch River in Tennessee. However, as the area affected was localized, i.e., <0.01% of the reservoir area, accelerated emergence was probably not significant for mayfly populations of the whole reservoir. In contrast, Langford and Daffern (1975) and Langford (1975) reported that, based on field observations, there was no evidence of a pattern of early emergence of ephemeropteran, trichopteran and megalopteran species downstream of power station cooling-water outfalls on the River Severn in Great Britain.

Wojtalik and Waters (1970) reported that water temperature affects the amplitude of drift within the diel periodicity for some benthic macroinvertebrate species but not for others.

Aquatic macroinvertebrates are generally considered valuable bioindicators of local ecosystem health as a result of their low mobility. The species composition of benthic invertebrate assemblages tends to change markedly near 32°C (Coutant 1959). Below this temperature, the annual cycles of individual species can be altered by persistent thermal change. Also, their important role as fish food assigns them particular importance in thermal effects assessments. Currently, extensive research on the thermal tolerance of various species has been conducted to support their use as evaluation tools for thermal plume effects.

Table 9 presents available thermal tolerance values for a number of benthic macroinvertebrate taxa. Most of the taxa are tolerant of elevated temperatures, i.e., up to and including 32ºC.

Table 9: Benthic Macroinvertebrate Thermal Tolerance Data

Taxon	Temp. (°C)	Thermal Metric[8]	Reference
P. Coelenterata			
Hydra pseudoligactis	34	ULT	Schroeder and Callaghan (1981)
H. oligactis	31	ULT	
P. Nematoda			
Cura foremani	33	96-h LC50	Chandler (1966)
Dugesia tigrina	35.5-39.4	ULT	Abbott (1960)
	33	No mortality	Chandler (1966)
Phagocata gracilis	30.5-34.9	ULT	Abbott (1960)
P. Annelida			
Cl. Oligochaeta			
F. Tubificidae			
Branchiura sowerbyi	35	96-h LC50 (No sed)[9]	Chapman *et al.* (1982)
	35	96-h LC50 (Sed)[10]	
Limnodrilus hoffmeisteri	34	96-h LC50 (No sed)	
	35	96-h LC50 (Sed)	
	37	96-h LC50	Birtwell and Arthur (1980)

[8] ILL = incipient lethal level; LHT = lethal high temperature; ULT = upper lethal temperature; LC50 = lethal concentration resulting in mortality of 50% of the test animals; TLm = median tolerance limit at which 50% of the test organisms are dead.

[9] No sed = bioassay testing with no sediment.

[10] Sed = bioassay testing with sediment.

Taxon	Temp. (°C)	Thermal Metric[8]	Reference
Quistadrilus multisetosus	32	96-h LC50 (No sed)	Chapman *et al.* (1982)
	35	96-h LC50 (Sed)	
Rhyacodrilus Montana	25	96-h LC50 (No sed)	
Spirosperma ferox	35	96-h LC50 (No sed)	
S. nikolskyi	25	96-h LC50 (No sed)	
	25	96-h LC50 (Sed)	
Stylodrilus heringianus	25	96-h LC50 (No sed)	
	25	96-h LC50 (Sed)	
Tubifex tubifex	35	96-h LC50 (No sed)	
	35	96-h LC50 (Sed)	
	34	96-h LC50	Birtwell and Arthur (1980)
Variechaeta pacifica	25	96-h LC50 (No sed)	Chapman *et al.* (1982)
Cl. Hirudinae			
Helobdella stagnalis	~30	No mortality	Schneider (1962)
Macrobdella	37	No mortality	Bushnell (1966)
Cl. Polychaeta			
Manayunkia speciosa	31	No mortality	Rolan (1974)
P. Arthropoda			
Cl. Arachnoidea			
O. Acariformes			
F. Hydrocarina			
Limnesia undulata	31.3	No mortality	Markowski (1959)
Cl. Malacostraca			
O. Amphipoda			
F. Gammaridae			
Gammarus fasciatus	34.6	24-h LHT	Sprague (1963)
	29	ILL	
	33.8	24-h LC50	Thibault and Couture (1980)
G. lacustris	26	96-h TL50	Smith (1973)
	25	30-d TL50	
G. limnaeus	30-32	ULT	Krog (1954)
G. pseudolimnaeus	29.6	24-h LHT	Sprague (1963)
	28	ILL	
	26	96-h TL50	Smith (1973)
	22-24	30-d TL50	
Hyalella azteca	33.2	24-h LHT	Sprague (1963)
	27	ILL	
	33	11-h LC50	Bovee (1949)
F. Pontoporeiidae			
Diporeai (Pontoporeia) affinis	12	24-h TLm	Smith (1972)
	10.8	96-h TLm	
	10.4	30-d TLm	
O. Isopoda			
F. Asellidae			
Caecidotea (Asellus) intermedius	34.6	24-h LHT	Sprague (1963)
	31	ILL	
Lirceus brachyurus	23	ILL	Cheper (1980)
L. fontinalis	40	Recorded as present	Styron (1968)
O. Decapoda			
F. Astacidae			
Orconectes obscurus	33	Acclimation temp	Hall *et al.* (1978)
O. rusticus	36.6	24-h LC50	Spoor (1955)
Pacifastacus leniusculus	32-33	ULT	Becker and Genoway (1974); Becker *et al.* (1975)
Cl. Insecta			
O. Ephemeroptera			
F. Baetidae			
Baetis rhodani	21.0-21.3	24-h LC50	Whitney (1939)
B. tenax	21.3	24-h LC50	
Cloeon dipterum	28.5	24-h LC50	
Pseudocloeon	41.1	Recorded as present	Trembley (1961) (in Jensen *et al.*, 1969)
Rhithrogena semi-colorata	22.4-24.7	24-h LC50	Whitney (1939)
F. Caenidae			
Caenis	26.7	24-h LC50	

Taxon	Temp. (°C)	Thermal Metric[8]	Reference
F. Ephemerellidae			
Ephemerella	30	Recorded as present	Trembley (1961) (in Jensen *et al*. (1969)
E. subvaria	21.5	96-h LC50	Nebeker and Lemke (1968)
F. Heptageniidae			
Arthroplea	30	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
Heptagenia	28.3	Recorded as present	
Stenonema	32.2	Recorded as present	
S. tripunctatum	25.5	96-h LC50	Nebeker and Lemke (1968)
O. Odonata			
F. Aeshnidae			
Boyeria vinosa	32.5	96-h LC50	Nebeker and Lemke (1968)
F. Coenagrionidae			
Argia	41.1	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
Ischura verticalis	31	No mortality	Baker and Feltmate (1987)
Libellula	>40	No mortality	Howell and Gentry (1974)
F. Corduliidae			
Somatochlora	36.7	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
F. Gomphidae			
Hagenius	26.7	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
Ophiogomphus rupinsulensis	33	96-h LC50	Nebeker and Lemke (1968)
F. Libellulidae			
Libellula	>40	No mortality	Howell and Gentry (1974)
	41.3	ULT (control accl temp 11°C)	Martin and Genrty (1974)
	42.8	ULT (control accl temp 15°C)	
	43.8	ULT (thermal accl temp 15°C)	
	47.6	ULT (thermal accl temp 30-33°C)	
	475	ULT (thermal accl temp 35°C)	
O. Plecoptera			
F. Capniidae			
Allocapnia granulata	23	96-h LC50	Nebeker and Lemke (1968)
F. Chloroperlidae			
Isogenus frontalis	22.5	96-h LC50	
F. Peltoperlidae			
Peltopera	27.8	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
F. Perlidae			
Acroneuria lycorias	30	96-h LC50	Nebeker and Lemke (1968)
A. californica	29	ILL	Heiman and Knight (1975)
Paragnetina media	30.5	96-h LC50	Nebeker and Lemke (1968)
F. Pteronarcyidae			
Pteronarcys dorsata	29.5	96-h LC50	
F. Taeniopterygidae			
Taeniopteryx maura	21	96-h LC50	
O. Megaloptera			
F. Corydalidae			
Corydalis	26.7	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
O. Trichoptera			
F. Brachycentridae			
Brachycentrus	28	Recorded as present	Roback (1965)
	30	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
B. americanus	29	96-h LC50	Nebeker and Lemke (1968)
F. Dipseudopsidae			
Phylocentropus	28	Recorded as present	Roback (1965)
F. Helicopsychidae			
Helicopsyche	35	Recorded as present	
F. Hydropsychidae			
Chematopsyche	35	Recorded as present	
Hydropsyche	35	Recorded as present	
	41.1	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
Macronema	35	Recorded as present	Roback (1965)
F. Hydroptilidae			
Agraylea	41.1	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
Hydroptila	30	Recorded as present	Roback (1965)
Leucotrichia	30	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
Oxyethira	32.2	Recorded as present	

Taxon	Temp. (°C)	Thermal Metric[8]	Reference
F. Lepidostomatidae			
Lepidostoma	28	Recorded as present	Roback (1965)
F. Leptoceridae			
Athripsodes	26.7	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
	35	Recorded as present	Roback (1965)
Leptocella	35	Recorded as present	
Mystacides	28	Recorded as present	
Oecetis	35	Recorded as present	
Triaenodes	35	Recorded as present	
F. Limnephilidae			
Pycnopsyche	28	Recorded as present	
F. Molannidae			
Molanna	28	Recorded as present	
	28.3	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
F. Philopotamidae			
Chimarra	28	Recorded as present	Roback (1965)
F. Phryganeidae			
Ptilostomis	23	Recorded as present	
F. Polycentropodidae			
Neuroclipsis	35	Recorded as present	
Polycentropus	35	Recorded as present	
	35	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
F. Psychomyiidae	30	Recorded as present	Roback (1965)
Psychomyia	30	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
O. Lepidoptera			
F. Pyralidae			
Parargyractis	41.1	Recorded as present	
O. Coleoptera			
F. Dytiscidae			
Deronectes elegans	42	ULT	Jones (1948)
Hydroporus palustris	36	ULT	
Hygrotus inaequalis	37	ULT	
F. Elmidae	30	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
F. Hydrophilidae			
Berosus	41.1	Recorded as present	
F. Psephenidae			
Psephenus	26.1	Recorded as present	
O. Diptera			
F. Chironomidae	41.7	Recorded as present	
S.F. Chironominae			
Calopsectra sp.	30	Recorded as present	Curry (1965)
C. dives	26.7	Recorded as present	
C. johannseni	32.8	Recorded as present	
C. neaflavella	32	Recorded as present	
C. nigripilus	32.8	Recorded as present	
Chironomus (Tendipes) sp.	30	Recorded as present	Markowski (1959)
C. albimanus	35	22-h LC50	Walshe (1948)
C. dux	32	Recorded as present	Curry (1965)
C. fumidus	26.7	Recorded as present	
C. longistylus	35.5	22-h LC50	Walshe (1948)
C. milleri	26.7	Recorded as present	Curry (1965)
C. modestus	26.7	Recorded as present	
C. neomodestus	32.8	Recorded as present	
C. nervosus	30	Recorded as present	
C. ochreatus	30	Recorded as present	
C. paganus	27	Recorded as present	
C. plumosus	30	Recorded as present	
C. riparius	32.8	Recorded as present	
	34.5	22-h LC50	Walshe (1948)
C. staegeri	30	Recorded as present	Curry (1965)
C. tendipendiformis	30	Recorded as present	
C. tentans	35	Recorded as present	
C. tuxis	27	Recorded as present	
Cryptochironoomus blarina	26.7	Recorded as present	
C. digitatus	30	Recorded as present	
C. fulvus	32.8	Recorded as present	

Taxon	Temp. (°C)	Thermal Metric[8]	Reference
Glyptotendipes sp.	32.8	Recorded as present	
G. lobiferus	32.8	Recorded as present	
		No mortality	Benda and Proffitt (1974)
G. paripes	27	Recorded as present	Curry (1965)
Harnischia tenuicaudata	30	Recorded as present	
Lauterborniella varipennis	26.7	Recorded as present	
Microtendipes pedellus	32.8	Recorded as present	
Paratendipes sp.	38.7	Recorded as present	
P. albimanus	32.8	Recorded as present	
Polypedilum sp.	30	Recorded as present	
P. fallax	32.8	Recorded as present	
P. halterale	30	Recorded as present	
P. illinoense	32.8	Recorded as present	
P. nubeculosum	26.7	Recorded as present	
P. simulans	40	Recorded as present	
Pseudochironomus sp.	30	Recorded as present	
P. richardsoni	32.8	Recorded as present	
Tanytarsus brunnipes	29	22-h LC50	Walshe (1948)
T. devinctus	26.7	Recorded as present	Curry (1965)
T. jucundus	26.7	Recorded as present	
T. nigricans	32.8	Recorded as present	
T. varius	32.8	Recorded as present	
Xenochironomus scopula	32.8	Recorded as present	
S.F. Diamesinae			
Diamesa nivoriunda	32.8	Recorded as present	
Prodiamesa olivacea	30	22-h LC50	Walshe (1948)
S.F. Orthocladiinae	30	Recorded as present	Markowski (1959)
Corynoneura sp.	32.8	Recorded as present	Curry (1965)
C. scutellata	32.8	Recorded as present	
Cricotopus sp.	38.8	Recorded as present	
C. absurdus	32.8	Recorded as present	
C. bicinctus	34	Recorded as present	
C. exilis	32.8	Recorded as present	
C. politus	32.8	Recorded as present	
C. tricinctus	32.8	Recorded as present	
C. trifasciatus	34	Recorded as present	
Hydrobaenus	34	Recorded as present	
S.F. Tanypodinae		Recorded as present	Curry (1965)
Anatopynia dyari	32.8	Recorded as present	
A. nebulosa	30.5	22-h LC50	Walshe (1948)
A. varia	38.8	22-h LC50	
Pelopia (Tanypus) sp.	39.5	Recorded as present	Curry (1965)
P. punctipennis	32.8	Recorded as present	
P. stellata	32.8	Recorded as present	
P. vilipennis	30	Recorded as present	
Pentaneura sp.	34	Recorded as present	
P. carnea	32.8	Recorded as present	
P. illinoesis	32.8	Recorded as present	
P. melanops	32.8	Recorded as present	
P. monilis	35	Recorded as present	
P. vitellina	32.6	Recorded as present	
Procladius sp.	30	Recorded as present	
P. bellus	26.7	Recorded as present	
P. choreus	26.7	Recorded as present	
P. culciformis	32.8	Recorded as present	
F. Empididae	30	Recorded as present	Trembley (1961) (in Jensen *et al*., 1969)
F. Simuliidae	41.7	Recorded as present	
F. Tabanidae	30	Recorded as present	
F. Tipulidae			
Atherix variegate	32	96-h LC50	Nebeker and Lemke (1968)
P. Mollusca			
Cl. Gastropoda			
F. Bithyniidae			
Bithynia tentaculata	33.7	48-h LC50	Thibault and Couture (1982)

Taxon	Temp. (°C)	Thermal Metric[8]	Reference
F. Physidae			
Physa gyrina	33	No mortality	Agersborg (1932)
	40	50% mortality from 7 to 16 h	Clampitt (1970)
	35	50% mortality in 11 to 13 d	
P. integra	40	50% mortality from 5 to 10 h	
	35	50% mortality in 5.7 to 8.7 d	
P. virgata	39.5	No mortality	McMahon (1975)
Cl. Pelecypoda			
F. Dreissenidae			
Dreissena polymorpha	30	ULT	Iwanyzki and McCauley (1993)
	34	100% mortality in 114 min	Rajagopal *et al.* (1997)
	38	100% mortality in 3 min	
	32	Mean survival time: 88±67 h/145±56 h	Hernandez and McMahon (1996)
	33	Mean survival time: 47±16 h/49±18 h	
F. Sphaeriidae			
Corbicula fluminea	32	ULT	Foe and Knight (1987)
C. manilensis	34	ILL	Mattice and Dye (1976)

5.1.6 Fsh

As indicated in Section 5.1, increased temperature results in increased rates of metabolism and respiration, as well as increased activity and food consumption in fish. Temperature also affects reproduction, growth and longevity. Moreover, thermal discharges may adversely affect fish by altering species diversity and/or trophic relations within communities (Spotila *et al.*, 1979).

A few field studies have demonstrated reduced condition or growth in ictalurids (bullhead catfishes) (Massengill, 1973; Stauffer *et al.*, 1976) and centrarchids (sunfishes) (Bennett, 1972; Graham, 1974) residing in thermal plumes. However, Lavis and Cole (1976) reported that neither fish growth nor condition appeared to be appreciably influenced by residence in the thermal discharge of the Monroe Power Plant. Similarly, Benda and Proffitt (1974) reported that heated water from a power plant on the White River at Petersburg, Indiana, did not affect growth (in length) of longear sunfish (*Lepomis megalotis*), spotted bass (*Micropterus punctulatus*), bluegill (*Lepomis macrochirus*) and white crappie (*Pomoxis annularis*). These centrarchids in the heated sections were fewer in number per hectare and had a higher average condition factor than those in the unheated section.

Bennett (1979) demonstrated that internal body temperatures of smallmouth bass were significantly higher in heated versus non-heated areas, which resulted in increased metabolic processes, different feeding regimes and increased food consumption.

Increased temperature may result in earlier initiation of the reproductive period and the development of embryos, young-of-the-year (YOY) and juveniles may be accelerated. Luksiene *et al.* (2000) reported that high temperature in thermal effluent areas in Europe influenced gametogenesis of female perch (*Perca fluviatilis*), roach (*Rutilus rutilus*) and pike (*Esox lucius*) negatively, indicating reduced reproductive capacity. Oocyte atresia started during vitellogenesis in autumn and was often followed by asynchronous egg cell development. However, no significant effect was observed in silver bream (*Blicca bjoerkna*). Ruffe (*Gymnocephalus cernuus*) had a tendency to produce an additional mature oocyte generation during the spawning period (i.e., three successful spawns in heated areas compared with two in the unheated areas). Sandstrom *et al. (1995*) reported a reduction in life-time fecundity in perch exposed to thermal effluent. Moreover, reproductive performance shifted to younger age-classes.

In contrast, smallmouth bass had high reproductive success (i.e., the percentage of nesting males that successfully produced free-swimming fry) in a thermal effluent canal on Lake Erie (McKinley *et al.*, 2000; Cooke *et al.*, 2003). In general, parental care activity was higher than previously reported for non-thermally altered environments and did not follow conventional stage-specific patterns. Males guarding nests resided in the heated discharge during the spawning period despite daily temperature fluctuations of up to 16°C. On several occasions, temperatures were near the published upper avoidance and tolerances, but the nesting males did not leave the canal until after the broods dispersed. The authors concluded that the high level of reproductive success implies that fish are capable of adjusting energetic expenditures in response to fluctuating thermal regimes.

Some fish species are attracted to a thermal plume (Coutant, 1975). Selection or avoidance of an outfall area by different fish species depends primarily upon their responses to temperature and food supply. Neill and Magnuson (1974) reported that some fish species avoided the outfall area of a power plant on Lake Monona, including yellow perch (*Perca flavescens*), mottled sculpin (*Cottus bairdi*), subadult yellow bass (*Morone mississippiensis*) and subadult black bullhead (*Ameiurus melas*). Other species were occasionally or usually concentrated in the outfall area relative to the reference areas, including longnose gar (*Lepisosteus asseus*), adult carp (*Cyprinus carpio*), adult yellow bass, bluegill, largemouth bass (*Micropterus salmoides*) and young pumpkinseed (*Lepomis gibbosus*).

Fish may occupy a thermal discharge for several advantageous reasons (Haynes *et al.*, 1989):

- discharge temperatures more closely approximate seasonally preferred temperatures than do ambient temperatures (Wyman, 1981; MacLean *et al.*, 1982; Spigarelli *et al.*, 1983);
- discharges attract and concentrate prey species (Spigarelli *et al.*, 1982; Janssen and Giesy, 1984);
- discharge currents resemble natural, although warmed, tributaries and produce a positive rheotactic response in some fish species (MacLean *et al.*, 1982); and
- fish maintain energetically optimal temperatures in or near discharges (Spigarelli *et al.*, 1983).

Everest (1973) reported that a greater number of fish were captured in the warm water of the former R.L. Hearn GS thermal plume than in waters unaffected by the plume, particularly during the fall, winter and spring.

Benda and Proffitt (1974) reported that eight species of fish were captured in the discharge canal at times when the water temperature ranged from 34 to 36°C, i.e., longear sunfish, channel catfish (*Ictalurus punctatus*), gizzard shad (*Dorosoma cepedianum*), river carpsucker (*Carpoides carpio*), carp, spotted bass, green sunfish (*Lepomis cyanellus*) and shortnose gar (*Lepisosteus platostomus*). No heat-induced fish kills were observed. Due to an increase in the numbers of thermally tolerant species of fish, such as carp, carpsucker spp., longnose gar and shortnose gar, the mixing zone section had a higher standing crop than the upstream control section. Most species seemed to be attracted by the discharge through the cooler months of the year (October to May), but all species were repelled by water temperatures that exceeded their thermal tolerance during the summer months.

Gammon (1973) reported that the fish assemblage in the Wabush River was stable for a range of temperatures from about 26°C to about 31°C based on numeric abundance, biomass and diversity. In the stretch of the river receiving thermal additions and gradually increasing in temperature over a period of days, subtle yet predictable shifts occurred in the composition of the assemblage. Species with relatively low optimum temperature preferences moved out of the heated segment and into cooler water, while species with relatively high optimum temperature preferences would move into the heated segment from cooler water. The net result is a shift in species composition, but no significant alteration in biotic indices. When the temperature of the segment exceeds about 31°C, there was a sharp reduction in numeric abundance, biomass and diversity as the assemblage was reduced to perhaps five or six thermally tolerant species. Further elevations to temperatures beyond 34°C or 35°C resulted in all species leaving the segment. Flathead catfish (*Pylodictis olivaris*) responded immediately with improved reproductive success and an increase in abundance within the warmed areas. In contrast, redhorse populations, mainly golden redhorse (*Moxostoma erythrurum*) and shorthead redhorse (*M. macrolepidotum*), were predicted to become permanently reduced in the warmed areas.

Coutant (1975) reported that gizzard shad are attracted to heated effluent discharged from the Bull Run Steam Plant on the Clinch River, Tennessee, in early spring, but dispersed when the heated discharge was terminated.

Spigarelli *et al.* (1982) reported that large numbers of alewife (*Pomolobus pseudoharengus*) congregate near and within thermal plumes of three power plants on Lake Michigan during the spawning period (May and June), but that adult rainbow smelt (*Osmerus mordax*) and YOY of both rainbow smelt (*Alosa pseudoharcagus*) and alewife were not strongly attracted to heated discharges between April and October. The densities of fish in plume areas can exceed densities in reference areas by two orders of magnitude during late spring and early summer, while reference area densities may exceed plume area densities by a factor of 10 in late summer and early fall. These responses indicate that adult alewife are attracted to the thermal and/or flow characteristics of the discharges during spawning season, but not during other seasons.

Similarly, Patriarche (1975) reported that carp and alewife concentrated near the discharge outlet of the Palisades Nuclear Power Plant on Lake Michigan, but elsewhere it appeared that several species were repelled to some extent and a few others attracted to the warm water of the thermal plume. There was no statistical difference in abundance of the two predominant species, yellow perch and alewife, before and after plant operation. Moreover, there was no statistical difference in the catch of yellow perch, alewife, longnose sucker (*Catostomus catostomus*) or white sucker (*C. commersoni*) in or outside the plume. Madenjian *et al.* (1986) also reported that the D.C. Cook Nuclear Power Plant on Lake Michigan had no effect on alewife and yellow perch abundances during the summer.

Lavis and Cole (1976) reported that, after preliminary operation of the Monroe Power Plant on western Lake Erie, when the discharge had stabilized, goldfish (*Carassius auratus*), carp and channel catfish were attracted to the discharge canal all year. Yellow perch, alewife, emerald shiner (*Notropis atherinoides*) and spottail shiner (*N. hudsonius*) almost constantly avoided the discharge canal. White bass (*Morone chrysops*), freshwater drum (*Aplodinotus grunniens*) and gizzard shad appeared to be sometimes attracted and at other times repelled.

Rainbow trout (*Oncorhynchus mykiss*) are attracted to thermal discharges into Lake Michigan and reside in heated plumes for variable periods of time (Spigarelli and Thommes, 1979). These rainbow trout were acclimated to plume temperatures with the mean excess acclimation temperature varying between 2 and 6 degrees over ambient. Seasonal high densities of forage fish near thermal discharges into Lake Michigan (Romberg *et al.*, 1974) appear to attract salmonid fish and may provide energetic advantages to plume-residents.

Ross and Siniff (1982) reported that yellow perch inhabiting the heated discharge from a power plant on the Upper Mississippi River during the winter selected relatively cool water (5.4°C overall mean temperature) from the 0 to 15°C gradient available and demonstrated a high degree of variability in mean selected water temperatures.

Shuter *et al.* (1985) reported a tendency for smallmouth bass of all ages to concentrate in the area of the Douglas Point NGS discharge from mid-summer until mid-fall, as well as an increase in the annual growth of all age groups. Similarly, Griffiths (1979a) reported that at Pickering NGS, radiotagged smallmouth bass maintained residence in the thermal plume for up to three months in summer and early fall. They moved out of the warmest waters available only when temperatures exceeded 30°C. However, MacLean *et al.* (1982) reported no long-term residency of plume areas at Nanticoke GS, as most tagged smallmouth bass and rock bass (*Ambloplites rupestris*) left the area within 24 h after release. These centrarchids appeared to be influenced more by currents and turbulence than temperature. However, it was reported that centrarchids did swim up the discharge canal when the lake temperatures were cold. Kelso (1974, 1976) also concluded that fish encountering thermal discharges at the Nanticoke GS and Pickering NGS respond largely to current.

Minns *et al.* (1978) reported that there was no evidence of altered fish distribution in relation to temperature at the Nanticoke GS and Douglas Point NGS. Dense clusters of fish were in the vicinity of the turbulent discharge in both locations. The lack of temperature response suggested that the fish were responding to currents and perhaps topography, e.g., in the area of shoals. Haynes *et al.* (1989) also reported that habitat is an important factor influencing the extent of fish attraction to thermal discharges, e.g., complex substrate versus flat bottom.

Most juvenile and adult fish are thought to have the swimming capacity to avoid the thermal plume of a power plant. Weakly swimming ichthyoplankton have a greater potential to intercept a thermal discharge plume. Increased vulnerability to predation of a number of juvenile fish species has been demonstrated following sublethal thermal shock (Coutant, 1973; Sylvester, 1972; Yocum and Edsall, 1974).

As indicated above, juvenile and adult fish will avoid elevated temperatures. Acclimation temperature has a direct positive effect upon avoidance temperatures (Mathur *et al.*, 1983), with fish avoiding temperatures higher than their acclimation temperatures. The differences between avoidance and acclimation temperatures decreased as acclimation temperatures increased. For example, the largemouth bass acclimated at 30°C avoided 36.1°C, a differential of 6.1 degrees, but when acclimated to 5.0°C avoided 20.6°C, a differential of 15.6 degrees.

Gray (1990) reported that thermal discharges (surface water ΔTs 0 to >17 degrees) did not block upstream migration of sonic-tagged adult chinook salmon (*Oncorhynchus tshawytscha*) and rainbow trout in the Hanford Reach of the Columbia River. Juvenile chinook salmon avoided thermal discharges in the laboratory when ΔT's exceeded 9 to 11 degrees above ambient. However, juvenile salmon were more susceptible to predation at 10 to 20% of the thermal dose causing loss of equilibrium.

Kelso (1974) reported that thermal discharge at the Pickering GS can interrupt or alter normal movement patterns of brown bullhead (*Ameiurus nebulosus*); however, their association with the plume was of short duration. In contrast, MacLean *et al.* (1982) reported that there was no evidence of the Nanticoke GS interfering with the spawning migration of smallmouth bass and rock bass.

Sudden temperature changes of a few degrees appear to have little effect on fish other than an increase in activity (e.g., Peterson and Anderson, 1969). However, sudden temperature changes of 10 degrees and above can cause significant stress in some fish, especially coldwater species, potentially resulting in mortality. Mobile adult fish will rarely be killed by heated water. They will either migrate from lethal conditions or congregate in areas with sublethal or near-optimal water temperatures. Although fish kills have been reported, these are generally due to the entrapment of fish in poorly designed discharge canals or associated lagoons when discharge temperatures increase dramatically or due to the initiation of thermal discharge after plant shutdown.

A sudden water temperature change, due to power plant shutdown during the winter, can result in cold shock to fish resulting in a loss of swimming ability (Reutter and Herdendorf, 1976). Lavis and Cole (1976) reported large numbers of dead gizzard shad in the discharge canal at the Monroe Power Plant on western Lake Erie. At these times, interruption in plant operations caused rapid temperature drops of over 10 degrees. Similarly, Ash *et al.* (1974) reported a fish kill due to cold shock in Lake Wabamun caused by a mechanical failure at the WPP. After shutdown, water temperatures in the area of the discharge canal dropped 16.9 degrees (from 21.8 to 4.9°C). It was estimated that approximately 258 000 spottail shiner and 200 northern pike were killed in the WPP discharge canal.

Based on extensive cold-shock studies at the Ginna Nuclear Power Plant on Lake Ontario, Smythe and Sawyko (2000) reported high survival rates for most fish species collected from the discharge canal and outlet area with temperatures typically between 11 and 15°C and exposed to test temperatures typically between 0.4 and 0.6°C. Gizzard shad had very low survival rates, i.e., 0 and 10%. In laboratory tests with rainbow trout, survival was 100 and 40% for fish acclimated at 10 and 15°C, respectively, when shocked at 0.5ºC. Survival rate was 100% for the 15°C-acclimated fish when shocked at 1°C. For six subsequent tests with 15°C-acclimated fish, survival rates ranged from 10 to 90% at exposure temperatures at or below 1°C. Even if fish survive the shock, they may become more susceptible to predation, or to secondary physiological effects. The potential for a summer heat shock is minimal because fish will avoid the higher plume temperatures and stay within their final temperature preferenda.

Mobile fish will rarely be killed by heated water. They will either migrate from lethal conditions or congregate in areas with sublethal or near-optimal water temperatures.

Temperature tolerance of freshwater fish species has been summarized by Brown (1974), Spotila *et al.* (1979), Houston (1982), and Wismer and Christie (1987). Table 10 presents available thermal tolerance values for a number of fish species recorded in Canadian waters. The upper incipient lethal temperatures and critical thermal maximums for warmwater fish species range from 30 to 40ºC for coolwater and coldwater fish species, these upper temperature limits range from about 21 to 30ºC.

Table 10: Representative Values for Thermal Tolerance of Canadian Freshwater Fish Species[11]

Species	**Stage**	**Acclimation Temperature (°C)**	**Upper LT50 and/or CT Max (°C)[12]**	**Reference** * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Northern brook lamprey (*Ichthyomyzon fossor*)	ammocoetes	15	30.5	Potter and Beamish (1975)

[11] Source: Spotila *et al.* (1979); Houston (1982); Wismer and Christie (1987).

[12] LT50 = incipient lethal temperature tolerated by 50% of the test population for a sustained period; CT Max = critical thermal maximum at w hich point locomotory movement becomes disorganized and the animal loses its ability to escape from conditions that may ultimately lead to its death.

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
American brook lamprey (*Lampetra lamottei*)	ammocoetes	15	29.5	Potter and Beamish (1975)
Sea lamprey (*Petromyzon marinus*)	prolarva (land-locked)	15 20	28.5 28.5	McCauley (1963)
	ammocoetes	5 15 25	29.5 30 31	Potter and Beamish (1975)
Bowfin (*Amia calva*)	-	23.8	37	Reutter and Herdendorf (1976)
Alewife (*Pomolobus pseudoharengus*)	egg	12-25 13	29.5 28	Jinks *et al.* (1981)
	YOY	10-12 18-20 24-26 10 20 25	26.5/28.3 30.3/32.7 32.1/34.4 26.5 30.3 32.1	Otto *et al.* (1976)*
	YOY	5 9	15 22.6	Graham (1956)*
	juvenile	17	24.5	McCauley (1981)*
	adult	10 15 20	23.5/29.5 23.5/30.1 24.5/31.2	Otto *et al.* (1976)*
	adult	1.4 5.3 9.5 12.4 15.1 19.5	25.1 26.7 28.4 29.5 30.6 32.4	McCauley (1981)*
	adult	10 15 20	20 22.8 22.8	Graham (1956)*
Gizzard shad (*Dorosoma cepedianum*)	underyearling	25 30 35	34-34.5 36.0 36.5	Hart (1952)
	YOY	summer	28.5	Cvancara *et al.* (1977)
	adult	15.9	31.7	Reutter and Herdendorf (1976)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Central stoneroller (*Campostoma anomalum*)	-	15 21 24 30	24 27 30 33	Cherry *et al.* (1977)*
Goldfish (*Carassius auratus*)	larvae	21-23	39.3	Jinks *et al.* (1981)
	juvenile	1-2 10 17 24 32 38	28 31 34 36 39.2 41.0	Fry *et al.* (1942, 1946)
	-	5 25	32 37.6	Talmage and Coutant (1979)
Spotfin shiner (*Cyprinella spiloptera*)	adult	15 21 24 30 36	24 27 30 36 38	Cherry *et al.* (1977)*
Common carp (*Cyprinus carpio*)	egg	25	35	Jinks *et al.* (1981)
	late-stage embryo	-	40-42.5	Crippen and Fahmy (1981)
	larva	16-21 19-27	36.4 38.8	Talmage (1978)
	-	26	35.7	Black (1953)
	-	-	40.9	Horoszewicz (1973)
Common shiner (*Luxilus cornutus*)	adult	5 10 15 20 25	26.7 28.6 30.3 31.0 31.0	Hart (1947)
	adult	10 15 20 25 30	29.0 30.5 31.0 31.0 31.0	Hart (1952)
	-	15 15	30.6 31.9	Kowalski *et al.* (1978)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Peamouth (*Mylocheilus caurinus*)	-	14	27.1	Black (1953)
River chub (*Nocomis micropogon*)	-	15	30.9	Kowalski *et al.* (1978)**
Golden shiner (*Notemigonus crysoleucas*)	adult	10 15 20 25 30	29.5 30.5 32.0 33.5 34.5	Hart (1952)
Emerald shiner (*Notropis atherinoides*)	YOY	-	35.2	Talmage (1978)
	juvenile	5 10 15 20 25	23.3 26.7 28.9 30.7 30.7	Hart (1947)
Common shiner (*Notrupis cornutus*)	-	winter spring	30.6 31.9	Kowalski *et al.* (1978)
Spottail shiner (*Notropis hudsonius*)	YOY	9 17 23-24 26	30.5 32.4 34.3 35.8	Jinks *et al.* (1981)
	YOY	26	34.7	Kellogg and Gift (1983)
	juvenile	23 26	36-37.3 36.8-38.1	Jinks *et al.* (1981)
	adult	21.7	32.8	Reutter and Herdendorf (1976)
Rosyface shiner (*Notropis rubellus*)	adult	15 21 24 30 33	24 27 27 33 34	Cherry *et al.* (1979)*
	-	15	31.8	Kowalski *et al.* (1978)**
Sand shiner (*Notropis stramineus*)	-	winter spring	32.3 33.0	Kowalski *et al.* (1978)
Bluntnose minnow (*Pimephales notatus*)	adult	5 10 15 20 25	26.0 28.3 30.6 31.7 33.3	Hart (1947)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
	adult	15 21 24 30	24 27 27 33	Cherry *et al.* (1977)*
Fathead minnow (*Pimephales promelas*)	-	15	31.9	Kowalski *et al.* (1978)**
	adult	10 20 30	28.2 31.7 33.2	Hart (1947)
	adult	15 21 24 30	24 27 30 32	Cherry *et al.* (1977)*
Northern squawfish (*Ptychocheilus oregonensis*)	-	19-22	29.3	Black (1953)
Blacknose dace (*Rhinichthys atratulus*)	adult	5 10 15 20 25	26.5 28.8 29.6 29.3 29.3	Hart (1947)
	adult	5 15 20 25	27 29.3 29.3 29.3	Hart (1952)
	-	15	31.9	Kowalski *et al.* (1978)**
Longnose dace (*Rhinichthys cataractae*)	-	15	31.4	Kowalski *et al.* (1978)
Leopard dace (*Rhinichthys falcatus*)	-	14	28.3	Black (1953)
Redside shiner (*Richardsonius balteatus*)	-	14	27.6	Black (1953)
Creek chub (*Semotilus atromaculatus*)	adult	5 10 15 20 25	24.7 27.3 29.3 30.3 30.3	Hart (1947)
	adult	10 15 20 25 30	27.5 29.0 30.5 31.5 31.5	Hart (1952)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	**Reference** * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Tench (*Tinca tinca*)	-	15 20 25	30.2 32.0 33.8	Alabaster (1964)
	-	-	39.3	Horoszewicz (1973)
Longnose sucker (*Catostomus catostomus*)	-	14	26.9	Black (1953)
White sucker (*Catostomus commersoni*)	adult	5 10 15 20 25	26.3 27.7 29.3 29.3 29.3	Hart (1947)
Largescale sucker (*Catostomus macrocheilus*)	-	19	29.4	Black (1953)
Northern hognose sucker (*Hypentelium nigricans*)	underyearling	18 21 24 30 33	27 30 33 33 34	Cherry *et al.* (1977)*
	-	15	30.8	Kowalski *et al.* (1978)
Black bullhead (*Ameiurus melas*)	YOY	summer	35.7	Cvancara *et al.* (1977)
	-	23	35.0	Black (1953)
Brown bullhead (*Ameiurus nebulosus*)	-	5 10 15 20 25 30 34	27.8 29.0 31.0 32.0 33.8 34.8 34.8	Hart (1952)
Channel catfish (*Ictalurus punctatus*)	juvenile	15 20 25	30.4 32.8 33.5	Hart (1952)
	juvenile	25 26 30 34 35	35.5 36.6 37.0; 37.8 38.0 38.0	Allen and Strawn (1968)
	juvenile	20 25 30	36.4±0.25 38.7±0.36 40.3±0.29	Currie *et al.* (1998)**

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Northern pike (*Esox lucius*)	larvae	17.7	24.8	Hokanson *et al.* (1973)
	YOY	summer	30.8	Cvancara *et al.* (1977)
	juvenile	30	33	
Tiger muskellunge (*Esox lucius* x *E. masquinongy*)	juvenile	25	34.0	Bonin and Spotila (1978)
Muskellunge (*Esox masquinongy*)	new-hatch	7 15 25	28.8 31.9 34.5	Hassan and Spotila (1976)
	1-15 d post-hatch	15±0.5	30.3-32.4	Paladino and Spotila (1978)
	larvae	25	32.8	Bonin and Spotila (1978)
Rainbow smelt (*Osmerus mordax*)	adult	1.0 1.6 3.1 5.4 6.5 8.2 12.2	22.6 22.8 23.3 24.1 24.5 25.1 26.4	McCauley (1981)
	-	<10 15	19 28.5	Ellis (1984)
	-	6	24.9	Reutter and Herdendorf (1976)
Eulachon (*Thaleichthys pacificus*)	adult	5	10.5	Blahm and McConnell (1976) (in Houston, 1982)
Cisco (lake herring) (*Coregonus artedii*)	larvae	-	19.8	McCormick *et al.* (1971)
	juvenile	2 5 10 20 25	19.7 21.7 24.2 26.2 25.7	Edsall and Colby (1970)
Lake whitefish (*Coregonus clupeaformis*)	YOY	5 10 15 20 22.5	20.62 22.67 25.78 26.65 26.65	Edsall and Rottiers (1976)**
Round whitefish (*Prosopium cylindraceum*)	embryo	-	4.8 / 5	Griffiths (1980)
	spawning	-	3	Dunford (1980)
	hatching	-	2.2	Scott and Crossman (1973)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Bloater (*Coregonus hoyi*)	yearling	5 10 15	22.2 23.6 24.8	Edsall *et al.* (1970)
Pink salmon (*Oncorhynchus gorbuscha*)	juvenile	5 10 15 20 24	21.3±0.3 22.5±0.3 23.1±0.3 23.9±0.6 23.9	Brett (1952)
Chum salmon (*Oncorhynchus keta*)	juvenile	5 10 15 20 23	21.8 22.6 23.1±0.4 23.7 23.8±0.4	Brett (1952)
Coho salmon (*Oncorhynchus kisutch*)	juvenile	5 10 15 20 23	22.9±0.3 23.7 24.3±0.3 25.0±0.2 25.0±0.2	Brett (1952)
	adult	17	25	Coutant (1970b)
Sockeye salmon (*Oncorhynchus nerka*)	juvenile	5 10 15 20 23	22.2±0.3 23.4±0.3 24.4±0.3 24.8±0.3 24.8±0.3	Brett (1952)
Chinook salmon (*Oncorhynchus tshawytscha*)	juvenile	5 10 15 20 24	21.5 24.3±0.1 25.0±0.1 25.1±0.1 25.1±0.1	Brett (1952)
	adult	18-19	21-22	Templeton and Coutant (1970)
	1-2 y	17	22	Coutant (1970b)
Cutthroat trout (*Salmo clarki*)	juvenile	10 15 20	27.6 29.1 29.9	Heath (1963)
Rainbow trout (*Salmo gairdneri*)	fingerling	5 9 13 17 21 24.5	23.7 24.2 25.2 25.7 26.2 26.2	Kaya (1978)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
	6 weeks	10 15 20	28.0±0.36 29.1±0.27 29.8±0.36	Currie *et al.* (1998)**
	underyearling	12 15 21 24	18 21 27 26	Cherry *et al.* (1977)*
	juvenile	5 9 13 17 21 24.5	23.2; 25.0 24.7; 25.2 24.7; 25.2 25.2; 25.7 25.7; 26.2 26.2;26.2	Kaya (1978)
	juvenile	11	24.0	Black (1953)
	juvenile	4 8 12 16	22.6 24.0 24.5 25.1	Threader and Houston (1982) (in Houston, 1982)
	adult	18-19	21-22	Templeton and Coutant (1970)
	-	16	25.6	Hokanson *et al.* (1977)
Atlantic salmon (*Salmo salar*)	new-hatch	6	22.0	Bishai (1960)
	30-d	5	22.2	Bishai (1960)
	post-hatch	10 20	23.3 23.5	Bishai (1960)
Brown trout (*Salmo trutta*)	new-hatch	6	22	Bishai (1960)
	30-d post-hatch	5 10 20	22.2 23.4 23.5	Bishai (1960)
	underyearling	12 15 21 24	15 18 24 25	Cherry *et al.* (1973)*
	-	6 15 20	23.2 26.0 26.4	Alabaster (1964)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Brook trout (*Salvelinus fontinalis*)	larvae	-	20.1	McCormick *et al.* (1972)
	underyearling	12 15 21 24	15 18 24 26	Cherry *et al.* (1973)*
	juvenile	13 16 19	24.0 24.9 25.8	Brett (1941)
	juvenile	3 11 15 20 22 24	23.5 24.6 25.0 25.3 25.5 25.5	Fry *et al.* (1946)
Lake trout (*Salvelinus namaycush*)	1-2 y	8 15 20	22.7 23.5 23.5	Gibson and Fry (1954)
Splake (*Salvelinus fontinalis* x *S. namaycush*)	juvenile	10 20	23.5-24.0 24.0-24.5	Fry and Gibson (1958)
Banded killifish (*Fundulus diaphanus*)	adult	25	34.5	Rombough and Garside (1977)
Threespine stickleback (*Gasterosteus aculeatus*)	-	10 20	25.7 27.2	Jordan and Garside (1972)
Prickly sculpin (*Cottus asper*)	-	18-19	24.1	Black (1953)
Mottled sculpin (*Cottus bairdi*)	-	15	30.9	Kowalsi *et al.* (1978)**
Slimy sculpin (*Cottus cognatus*)	-	20	25	Symons *et al.* (1976)**
White perch (*Morone americana*)	juvenile	27	36	Jinks *et al.* (1981)
White bass (*Morone chrysops*)	new-hatch	14 18 20 26	31.7 30.8 32.0 30.6	Yellayi and Klambi (1969)
	YOY	summer	33.5	Cvancara *et al.* (1977)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Rock bass (*Ambloplites rupestris*)	underyearling	18 21 24 30 36	27 27 30 33 37	Cherry *et al.* (1977)*
Pumpkinseed (*Lepomis gibbosus*)	-	24	30.2	Black (1953)
Bluegill (*Lepomis macrochirus*)	egg	-	33.8	Banner and Van Arman (1973)**
	underyearling	15 21 24 30 36	27 30 33 36 38	Cherry *et al.* (1977)*
	YOY	summer	28.5	Cvancara *et al.* (1977)**
	juvenile	12.1 32.9	27.5 37.3	Banner and Van Arman (1973)*
	adult	15 20 20-23 25 30 30	30.5 32.0 34.0-35.5 33.0 33.8 36.0-38.0	Hart (1952)
	-	1 13 27	23.3-23.5 29.3-30.0 25.8-36.0	Peterson and Schotsky (1976)
Longear sunfish (*Lepomis megalotis*)	-	25 30 35	35.6 36.8 37.5	Neill *et al.* (1966)
Smallmouth bass (*Micropterus dolomieu*)	underyearling	18 21 24 30 33	27 30 33 33 35	Cherry *et al.* (1977)*
Largemouth bass (*Micropterus salmoides*)	underyearling	30 35	36.4 37.5	Hart (1952)
	YOY	summer	35.6	Cvancara *et al.* (1977)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
	9-11 months	20 25 30	32 33 33.7	Hart (1952)
	9-11 months	20 25 30	32.5 34.5 36.4	Hart (1952)
	juvenile	20 25 30	35.4±0.47 36.7±0.59 38.5±0.34	Currie *et al.* (1998)**
	-	20-21	28.9	Black (1953)
Black crappie (*Pomoxis nigromaculatus*)	adult	23.8	34.9	Reutter and Herdendorf (1976)
Greenside darter (*Etheostoma blennioides*)	-	15	32.2	Kowalski *et al.* (1987)
Rainbow darter (*Ethcostoma caerulum*	-	15	32.1	Kowalski *et al.* (1987)
Fantail darter (*Etheostoma flabellare*)	-	15	32.1	Kowalski *et al.* (1987)
Johnny darter (*Etheostoma nigrum*)	-	winter spring	30.7 31.4	Kowalski *et al.* (1978)
Yellow perch (*Perca flavescens*)	larvae	-	19.9	Hokanson (1977)
	underyearling	15 21 24	21 27 29	Cherry *et al.* (1977)*
	adult	5 11 15 25	21.3 25.0 27.7 29.7	Hart (1947)
	-	22-24	29.2	Black (1953)
Sauger (*Sander canadensis*)	larvae	-	20.7	Hokanson (1977)
Walleye (*Sander vitreum*)	larvae	-	19.2	Hokanson (1977)
	juvenile	25.8	31.6	Smith and Koenst (1975)
	juvenile	8-24.0	27-31.5	Ellis (1984)
	adult	7.2 26	28.9 34	Wrenn and Forsythe (1978)

Species	Stage	Acclimation Temperature (°C)	Upper LT50 and/or CT Max (°C)[12]	Reference * Corrected data from source reference. ** Additional data not included in Spotila et al. (1979), Houston (1982), and Wismer and Christie (1987).
Freshwater drum (*Aplodinotus grunniens*)	YOY	summer	32.8	Cvancara *et al.* (1977)
	adult	21.2	34.0	Reutter and Herdendorf (1976)

5.1.7 Aquatic Avifauna

Waterfowl tend to congregate in the ice-free waters created by thermal discharges from power plants in the winter months (McCullough, 1984; Prince *et al.*, 1992). For example, diving ducks such as scaup (*Aythya* spp.), common goldeneye (*Bucephala clangula*) and bufflehead (*Bucephala albeola*) winter near the Nanticoke GS on Lake Erie and feed intensively on zebra mussel (*Dreissena polymorpha*) and quagga mussel (*D. bugensis*) in the ice-free areas (Mitchell *et al.*, 2000).

Acres (1983) reported that the inner bay of the Outer Harbour was an important wintering area for aquatic avifauna due to the warmwater discharge from the former R.L. Hearn GS.

When food is abundant, overwintering of waterfowl in thermal discharge areas is likely of little ecological concern unless the waterfowl are detained by the warm, open water without adequate food or in such concentrations that they are preyed upon severely.

5.1.8 Water Uses

Thermal discharge generally has no adverse effect on recreational/commercial vessel utilization of the receiving water. In fact, recreational boat usage may occur earlier in the spring and later in the fall due to the warmer surface water temperatures.

As indicated in sections 2.4, 2.5 and 2.6, cooling water mixing zones should not interfere with other water uses such as drinking water supply, bathing areas or other sensitive designated use areas.

Sportfish angling is often enhanced in thermal discharges in winter. Elser (1965) reported that sport catch in heated water yielded substantially more fish per trip except during the summer. Gibbons *et al.* (1972) reported a greater concentration of largemouth bass and greater angling success during the winter in an area affected by heated effluent. Similarly, Shuter *et al.* (1985) found that seasonal abundance of smallmouth bass caught by angling increased near thermal discharges. Spigarelli and Thommes (1976) reported that a successful sport fishery for rainbow trout, brown trout, brook trout and lake trout, as well as coho salmon and chinook salmon, has developed at the Point Beach Nuclear Plant thermal discharge. Rainbow trout and brown trout accounted for 75% of the total number of fish caught, and all salmonid species comprised 88% of the total catch. CPUE at this thermal discharge was higher than in most reference "shoreline" fisheries in the area.

A greater number of fish were captured in the warm water of the former R.L. Hearn GS thermal plume than in waters unaffected by the plume, particularly during the fall, winter and spring (Everest, 1973). The increased temperature and water flow served as a major attraction for fish and therefore for recreational fishermen (Richardson, 1980). During former R.L. Hearn GS operation, the Discharge Channel outlet area was used extensively for sportfishing. This location was rated second and third in terms of Toronto waterfront angling pressure and angling potential, respectively (MacNab and Hester, 1976). With the cessation of former R.L. Hearn GS operation, sportfishing success in the area has declined substantially (Kalff *et al.*, 1991).

5.2 Assessment Methodology

The assessment of potential effects of a thermal plume on fish, other biological resources and water use is based on the facility design/operational information, data on the thermal and limnological/hydrological regime of the intake and receiving water areas, thermal plume modelling results, site-specific information on benthic macroinvertebrate communities (shoreline discharge only), site-specific information on fisheries resources and fish habitat, available information on other biological resources and nearby water uses, and the thermal tolerance and resistance of specific endemic species (tables 9 and 10), particularly VECs (usually fish species). The identification of VECs should be based on their ecological significance, perceived effectiveness as indicators of environmental impact, designation as a species at risk (COSEWIC, 2007), importance to Aboriginal values and traditional uses, and importance to the commercial/recreational fishery. The results of thermal plume modelling for the BBPS, PEC, TCP and BANRP projects are presented in sections 4.1, 4.2, 4.3 and 4.4, respectively. The following sections provide a summary of the various methods used in project-specific assessments of biological responses to thermal discharges.

5.3 BBPS Thermal Discharge Effects on Biota

The BBPS in Windsor, Ontario, is a 550 MW natural gas-fired combined cycle facility that discharges cooling effluent to the Detroit River. Based on the findings of thermal plume studies at other power plants, BEAK (2001) predicted that thermal discharge by the BBPS would likely result in a localized increase in primary production and zooplankton standing crop, as well as species-specific changes in benthic macroinvertebrate populations, possibly affecting species diversity (see Section 7.2).

As indicated in Table 10, the upper incipient lethal temperatures and critical thermal maximums for warmwater fish species range from about 30 to 40°C, i.e., near or above the maximum BBPS cooling water discharge temperature of 34°C. It was predicted that those warmwater species attracted to the thermal plume will select their thermal preference to become year-round or short-term residents.

For coolwater and coldwater fish species, these upper temperature limits range from about 21 to 30°C. Some coolwater species may avoid the plume during the summer. Coldwater species are likely not present in the Detroit River in the summer due to the elevated ambient water temperatures. In the spring/fall and winter, water temperatures at about 5 m downstream of the discharge would be about 14 to 15°C and 5 to 6°C, when ambient temperatures are 10 and 1°C, respectively. With this relatively small temperature differential within such a short distance from the discharge, it is unlikely that a shutdown of the BBPS in the water would have a significant cold shock effect on sensitive fish species. Coldwater fish species may become attracted to the thermal plume to take advantage of any increased densities in forage fish. The potential for a summer heat shock is minimal because fish will avoid the higher plume temperatures and stay within their final temperature preferenda.

Overall, thermal discharge by the BBPS facility was expected to have negligible effect on fish habitat and a minimal effect on local fish populations, primarily year-round or short-term attraction by some fish species and avoidance by some coolwater species in the summer.

Subsequently, Eakins and Fitchko (2001, 2003) observed no important fish spawning habitat in the nearshore area adjacent to the BBPS and up to 1 km downstream.

Because the BBPS cooling water discharge occurs at the Detroit River shoreline and the thermal plume was predicted to extend along the Canadian shoreline downstream, the effect on nearshore benthic macroinvertebrate communities was raised as being a concern by the MOE (Section 2.9.1). As part of the Terms and Conditions of the C of A, BBPS was required to undertake pre-operational and operational benthic macroinvertebrate community surveys. The study design was presented in Secton 3.1, and the survey findings are presented in Section 7.2.

Environment Canada recommended that BBPS develop contingency plans for thermal effects that are robust enough to accommodate the likelihood of increased incidences of warmwater episodes due to global warming.

5.4 PEC Thermal Discharge Effects on Biota

The 550 MW PEC on the Toronto waterfront of Lake Ontario is a natural gas-fired facility that commenced operation using simple cycle technology, with plans for conversion to combined cycle one year after start-up. As indicated in Section 4.2, under typical summer conditions and normal plant operations (i.e., ΔT of 5.5 degrees), Outer Harbour water temperatures would not change appreciably. Under worst-case summer conditions (i.e., surface water temperature of 26°C) and normal plant operations, the discharge of once-through cooling waters may reduce the surface water temperature within the uppermost basin of the Outer Harbour from 26 to approximately 21°C, a 5 degree temperature decrease. This decrease was attributed to the location of the intake and cooler temperatures within the hypolimnion. Under typical winter and spring/fall conditions, the discharge of once-through cooling waters may increase the surface water temperature within the uppermost basin of the Outer Harbour to approximately 7 and 15°C, respectively, i.e., a temperature increase of 4 and 5 degrees, respectively. An increased temperature was anticipated since the Ship Channel where the deep intake is located is generally not stratified during these seasons. Under an unusual operating condition (i.e., ΔT of 10 degrees rather than 5.5), the water temperature within the uppermost basin of the Outer Harbour was estimated to remain relatively unchanged from ambient during the summer, i.e., 0 to 1 degree increase in cooling water discharge, and to potentially increase by approximately 9.5 degrees during the winter, spring and fall.

Stantec and SENES (2003) concluded that thermal discharge by the PEC may result in a localized increase in primary production and zooplankton standing crop during the fall, winter and spring, and no effect during typical summer conditions. The anticipated cooling effect on water temperatures of the Outer Harbour during an extremely warm summer may result in a subtle localized decrease in primary and secondary production.

Most of the benthic macroinvertebrate taxa recorded in the PEC study area, e.g., nematodes, tubificid oligochaetes, Hydrocarina (water mites), *Gammarus fasciatus, Caecidotea, Chironomus, Cryptochironomus,* snails and clams, are tolerant of elevated temperatures, i.e., at and above 30°C (Table 9). Based on available thermal tolerance data, only two species, *G. lacustris* and *Tanytarsus*, may be affected by the worst-case maximum temperature of 30°C if present in the Discharge Channel or the nearshore of the uppermost embayment. As the cooling water discharge would only affect the surface waters of the uppermost basin of the Outer Harbour, benthic macroinvertebrate communities in deeper waters would not be affected.

As indicated in Table 10, the upper incipient lethal temperatures and critical thermal maximums for warmwater fish species range from about 30 to 40°C, i.e., at or above the unlikely maximum PEC cooling water discharge temperature of 30°C stipulated in the C of A.

For coolwater and coldwater fish species, these upper temperature limits vary from about 21 to 30°C within the range anticipated during the summer under typical and worst-case conditions. In the spring/fall and winter, water temperatures in the outermost basin of the Outer Harbour would be about 15 and 7°C, when ambient temperatures are 10 and 3°C, respectively. Coldwater fish species may become attracted to the thermal plume to take advantage of any increased densities in forage fish. During the operation of the former R.L. Hearn G.S., a greater number of fish were generally captured during monthly surveys from March 1970 to June 1972 in the warm water of the thermal plume than in waters unaffected by the plume, particularly during the fall, winter and spring (Everest, 1973).

Most phases of the reproductive process of fishes exhibit some dependency upon ambient temperature. Moreover, gonadal development, gamete maturation, reproductive migration, spawning, embryonic development and hatching, and larval growth have differing thermal requirements that are closely correlated with normal seasonal variations in habitat temperature. The assessment evaluated what part of the reproductive cycle most likely would be affected, and compared the vulnerable stages to available information from the literature, as illustrated below.

Unusual operating conditions (e.g., 10 degree ΔT) are unlikely and, if occurring during PEC operation, would be of a short-term nature. As indicated in Table 6, water temperatures under these unusual conditions would be about 19.5 and 12.4°C in the uppermost basin of the Outer Harbour during the spring/fall and winter compared to ambient temperatures of 10 and 3°C, respectively. Mobile fish stages with seasonal thermal preferences below these temperatures will avoid the area of elevated temperatures and move back when temperatures decrease with a return to normal operating conditions. Therefore, the primary potential deleterious impact under both normal and abnormal operations would be on those life stages that are immobile, i.e., fish eggs and larvae. Based on review of available databases, upper incipient lethal temperatures for fish eggs and larvae of those fish species presumably or likely spawning in the uppermost basin of the Outer Harbour generally range from the upper teens to the thirties degrees Celsius depending upon acclimation temperatures (see Table 11).

Table 11: Fish Egg and Larva Upper Incipient Lethal Temperatures[13]

Species	Stage	Acclimation Temperature (°C)	Incipient Upper Lethal Temperature (°C)
Alewife	Egg	12-25	24.5
	Larva	14-24	31.4-37.1
Common carp	Egg	25; 26.3	35; 31-35
	Larva	19-27; 16-21	38.8; 36.4
White sucker	Larva	9-10; 15; 15-16; 21	28.3-28.8; 31; 30-31.1; 30
Northern pike	Egg (2-4 cells)	-	19.8
	Egg (eyed)	-	28.0
	Embryo	-	18.9; 16.8-20.5
	Larva	17.7	28.5
	Yolk-sac larva	11.8; 14-15; 17.7	24.1; 31; 24.8
White perch	Larva	8; 15; 18-24; 21-22; 24; -	27; 30.3-35.6; 38.5; 31.0-38.4; 35.4-38; 34.8
Largemouth bass	Egg	Summer	35.6
Yellow perch	Embryo	-	19.9
	Hatch	-	19.9
	Larva	7.6; 10; 15 15.8; 18; 19 22-24; -; -	>24; 10; 33.7 >26.6; 26.5; 19 29.2; 23.9; 31.3
	Swim-up larva	-	18.8

13 Source: Wismer and Christie (1987).

Optimum embryo development and hatching temperatures are generally somewhat lower than optimum larvae temperatures as indicated in Table 12. The predicted temperature of 19.5°C for unusual conditions in the spring in the uppermost basin of the Outer Harbour is near the upper end or slightly exceeds the maximum temperature for embryo development and hatching for some fish species (e.g., northern pike and yellow perch). However, as indicated in Section 5.1, fish species, like all organisms, have the ability to adapt or acclimate to seasonal changes in temperature or to ranges of temperature shifts that they may encounter. The upper lethal temperature of organisms increases as acclimation temperature rises until an ultimate upper lethal temperature is reached.

Under normal operating conditions in the spring, the water temperature in the uppermost basin is predicted to be 15.2°C. As indicated in Table 10, at acclimation temperatures of 15 and 15.8°C, the incipient upper lethal temperatures for yellow perch larvae are 33.7 and >26.6°C, respectively, well above the 19.4°C predicted under upset conditions. For northern pike embryo, the incipient upper lethal temperature ranges from 16.8 to 20.5°C (no acclimation temperature data were available). However, since this life stage would be acclimated to a predicted temperature of 15.2°C in the uppermost basin, the upper lethal temperature would likely be close to the upper end (20°C) of the range. Moreover, the incipient upper lethal temperature is based on prolonged exposure, i.e., a standard seven-day week is generally used as the lethal test exposure time. Any upset conditions would be expected to be rectified well within this time period. If sustained, embryo development and hatching may be affected, possibly resulting in an earlier hatch. However, as indicated above, the unusual operating conditions are unlikely and would be short-term if they occurred and therefore have negligible effect on embryo development and hatching.

Table 12: Optimum Fish Embryo Development and Hatching Temperatures[14]

Species	Optimum Temperature (°C)		Temperature Range (°C)	
	Embryo Development	Hatching	Embryo Development	Hatching
Alewife	17.8	17.7; 20.8	10-26.7	-
Common carp	20-25; 23-24.9	23.4	13.8-26	11-32
Emerald shiner	-	23.9	-	-
Spottail shiner	-	20	-	-
Longnose dace	-	15.6	-	-
White sucker[15]	15; 15.2	-	8-21; 9-17.2	-
Rainbow smelt	-	14	6-22.5	-
Northern pike	6.5-17.7; 12	20.8; 6.4-17.7	2-23; 7-19	5.8-21
Round whitefish	1-2	2.2	-	-
Brook stickleback	-	18.3	-	15-18
Threespine stickleback	-	19	-	-
Mottled sculpin	-	-	-	7.8-17.3
White perch	17.6	14.1; 17.6	10-24	10-24
Pumpkinseed	-	28	-	-
Largemouth bass	-	20	<32.1	13-26
Black crappie	-	18.3; 16-20	-	-
Johnny darter	-	22-24	-	-
Yellow perch	13.1-18.2	10-20[15]	7-15	7-20[15]
Walleye	7.8-8.9; 10.5-15.5; 15-19	<12; 8.4 6; 9-15; 16.7-19.4; 17.8-19.4	5-19	8-19; 7-10 6-12

As indicated in Section 2.1, CWQGs have been established to protect aquatic biota based on warm-water season and winter MWAT and short-term exposure to extreme temperature. Specifically, thermal additions to receiving waters should be such that the MWAT for warm-water seasons is not exceeded. As indicated above, cooling water discharge from the PEC will not appreciably affect or may even lower surface-water temperatures in the innermost basin of the Outer Harbour during typical and worst-case summer conditions, respectively. As indicated in Table 6, during the winter under normal plant operations, surface water temperatures in the uppermost basin of the Outer Harbour would be about 4°C above the ambient temperature. For the extreme operating condition when the ΔT is 10°C, surface water temperatures would be about 9°C above the ambient temperature. With these relatively small temperature differentials, i.e., below 10°C over a period of 2 to 3 h, it is highly unlikely that a shutdown of the PEC would have a significant cold shock effect on sensitive fish species (Houston, 1982; Wismer and Christie, 1987). As a result, the MWAT for winter will not be exceeded.

[14] Source: Wismer and Christie (1987)

[15] Temperature data provided for embryo development and hatching.

[15] Temperature data provided for embryo development and hatching.

In addition, during reproductive seasons, the MWAT must meet specific site requirements for successful migration, egg incubation, fry rearing and other reproductive functions of important species. Thermal discharges from the PEC during typical spring conditions may elevate temperatures in the nearshore waters of the uppermost basin of the Outer Harbour by 5 degrees (see Table 6). This temperature increase, which is likely comparable to that in the protected embayments of the Outer Headland, would likely have no measurable effect on the reproductive functions of important species. Available MWAT data for month of peak spawning (i.e., ≤ to optimum or middle range of spawning temperatures) for fish species presumably and likely spawning in the Outer Harbour are presented in Table 13. Based on these data, the MWAT for most species is above the predicted temperature in the uppermost basin of the Outer Harbour during the spring spawning period, or no change in water temperatures are anticipated in the innermost basin during the summer spawning period due to thermal stratification in the Ship Channel.

For walleye, the MWAT is approximately equal to the predicted spring temperature in the uppermost basin. For rainbow smelt, northern pike and yellow perch, spawning activities in the uppermost basin may occur earlier than usual; however, an overlap with the normal spawning season temperatures is still maintained. For round whitefish, any spawning in the nearshore of the uppermost and middle basins would likely be precluded under typical conditions; however, spawning in the lowermost basin, e.g., Cherry Beach, would not be affected (see Table 6). Under worst-case conditions, spawning would be precluded in the nearshore of all three basins. The collection of only two larval and one adult round whitefish during the May 2004 survey suggests that the Outer Harbour provides limited spawning and nursery habitat for this species (Eakins and Fitchko, 2004). Moreover, the loss of this spawning habitat is insignificant compared to that available over gravel, sand and rock along the Lake Ontario nearshore (e.g., Rukavina, 1969; Thomas *et al.*, 1972; Barton, 1986).

Table 13: MWAT Data for Month of Peak Spawning[16]

Species	MWAT (°C)	Spawning Months	Uppermost Basin Temperature (°C)	
			Ambient	With PEC
Alewife	14.4	June-August	Stratified	N/C[17]
Common carp	21	May-August	10-Stratified	15-N/C
Emerald shiner	23	June-August	Stratified	N/C
Spottail shiner	17.5	May-June	10-Stratified	15-N/C
Bluntnose minnow	25	June-August	Stratified	N/C
Longnose dace	15	May-July	10-Stratified	15-N/C
White sucker	10	April-June	4-Stratified	9-N/C
Rainbow smelt	13.6	March-May	2-10	7-15
Northern pike	12	March-May	2-10	7-15
Round whitefish	3	November-December	4-2	9-7
Brook stickleback	18.3	May-July	10-Stratified	9-N/C
Threespine stickleback	19	May-July	10-Stratified	15-N/C
White perch	17	May-June	10-Stratified	15-N/C
Rock bass	21	May-June	10-Stratified	15-N/C
Largemouth bass	23.9	May-June	10-Stratified	15-N/C
Yellow perch	12	April-May	4-10	9-15
Walleye	8.9	April-June	4-Stratified	9-N/C

The CWQG further states that, at a specific site, the MWAT preserves normal species diversity or prevents undesirable growths of nuisance organisms. The PEC cooling water discharge would not affect normal species diversity, although during the cooler months additional fish species may be attracted to the warmer waters. Moreover, since the surface water temperatures in the uppermost basin of the Outer Harbour would not change appreciably or were predicted to decrease under typical and worst-case summer conditions, respectively, further undesirable growths of nuisance organisms, e.g., aquatic vegetation, would be prevented.

5.5 TCP Thermal Discharge Effects on Biota

The 305 MW TCP in Thorold, Ontario, is designed to burn natural and landfill gas and supply steam to an adjacent paper mill, discharging its cooling effluent into the Welland Canal. As indicated in Section 4.3, modelling results demonstrated that the thermal plume created by the TCP cooling water discharge may extend 1500 to 2000 m downstream and 100 to 150 m upstream of the outfall, for a continuous 12-h operation. However, due to the

[16] Source: Wismer and Christie (1987).
[17] N/C = little or no change in Outer Harbour w ater temperatures.

significant temperature difference between the effluent and receiving water, the discharge will be strongly buoyant and generally restricted to the upper 1 to 3 m of the Welland Canal.

Based on the findings of thermal plume studies at other power plants, Fitchko (2006a) concluded that thermal discharge by the TCP may result in a localized increase in primary production and zooplankton standing crop and possibly alteration of nearshore benthic species composition. As the thermal plume will be buoyant, and even with intermixing with deeper waters during vessel passage, the very sparse benthic macroinvertebrate communities in the middle of the channel would likely not beaffected.

Roshon (2002) concluded that cooling water discharge from the previously proposed larger cogeneration plant would have a minimal effect on the algal population within the thermal plume. This conclusion is based on a literature review that suggests:

- the thermal plume exists primarily in the upper metre of the water where nuisance algae (e.g., *Cladophora glomerata, Oscillatoria* sp.) are less dominant;
- depending on the strain of nuisance algae, there might be a slight reduction in their population levels or no change;
- warmer summer temperature may have a negative effect on the survival and growth of diatoms in the plume; and
- warmer winter temperatures will have a positive impact on the survival and reproduction of the diatoms in the plume, possibly negating the adverse summer effects.

However, the following species composition changes are possible:

- the longer growing season may stimulate growth of the filamentous algal species;
- the filamentous nuisance algae might become more prevalent in the upper waters attached to surfaces because these species appear to have a greater temperature tolerance; and
- the unicellular algal species (e.g., diatoms), which compete with the nuisance algae, may become less dominant in the summer providing a greater niche for the nuisance algae.

Temperature is not the only factor that can influence algal growth. Light quality and quantity and nutrient availability are also vital factors that will affect the algal species composition and density. Total phosphorus concentrations in the Welland Canal are below the interim PWQO of 30 µg/L established to avoid excessive plant growth.

These conclusions were based on continuous operation of the previously proposed larger cogeneration plant. As indicated in Section 2.9.1, the TCP will only operate 40% of the time during the year.

Concern was expressed by the SLSMC regarding the effect of higher temperatures on the local zebra mussel population. Zebra mussels are best adapted to cool temperate conditions, with optimal temperatures for growth in the 18 to 20°C range. Upper incipient lethal temperatures have been reported at 30°C or higher (Iwanyzki and McCauley, 1993;

Hernandez and McMahon, 1996). In general, temperatures above 25°C have a negative impact on zebra mussel growth and other physiological functions (Rajagopal *et al.*, 1997). At temperatures below 12°C, zebra mussel growth is poor and reproduction is halted (Claudi and Mackie, 1999). Based on review of these and other data, O'Reilly (2002) concluded that the impact of the TCP thermal plume on the local zebra mussel population would likely be minimal. The potential increase in survival and growth due to increased temperatures in the fall, winter and spring would likely be negated by increased stress and mortality due to the elevated temperatures in the summer.

The Roshon (2002) and O'Reilly (2002) reports are included as appendices in Stantec (2006).

The upper incipient lethal temperatures and critical thermal maximums for warmwater fish species range from about 30 to 40°C, i.e., near or above the maximum TCP cooling water discharge temperature of 32°C (Table 10).

For coolwater fish species, e.g., alewife and rainbow smelt, the upper temperature limits are below the ambient Welland Canal water temperature during the summer. It was anticipated that these species vacate the Welland Canal as ambient temperatures approach their upper temperature limits (Fitchko, 2006a).

As indicated in Section 4.3, during the winter when there is no boat traffic, the buoyant plume will be about 6 degrees above the ambient temperature within 100 m upstream and downstream of the outfall. Due to thermal plume restriction to the upper 1 m of the receiving water and the relatively small temperature differential, it is highly unlikely that a shutdown of the TCP would have a significant cold shock effect on fish. The potential for a summer heat shock is minimal because fish will avoid the higher plume temperatures and stay within their final preferenda.

Overall, thermal discharge by the TCP was expected to have negligible effect on fish habitat and a minimal effect on local fish populations, primarily short-term attraction or avoidance by some fish species.

McCorquodale (2006) evaluated the force of cooling water discharge on a passing vessel based on the momentum flux of the discharge, impact area of the plume on the vessel and ship speed. Based on initial outfall design, it was determined that the cooling water discharge for jet velocities in the range of 0.2 to 0.5 m/s would have negligible effect on the sailing line for ships travelling at 4 to 6 knots. For slow-moving (e.g., 2 knots) ships, the net displacement would be of the order of 1 m at the highest cooling discharge velocity. In addition, a passing ship would experience a small planwise rotation of the bow away from the jet followed by a counter-rotation back to a line parallel to the initial starting sailing line. At 2 kt and boat length of 122 m, this rotation would be about ±18 minutes. This effect decreases with increased ship speed.

As the SLSMC expressed concern about the 1-m net displacement of slow-moving ships, McCorquodale (2006) recommended a revised outfall design that would reduce the maximum lateral deflection to 20 cm. After peer review, the SLSMC stated that the revised design is acceptable (P. Pesant, SLSMC, 2006, pers. comm.).

The McCorquodale (2006) report is included as an appendix in Stantec (2006).

5.6 BANRP Thermal Discharge Effects on Biota

The BANRP in Tiverton, Ontario, provides the Bruce Nuclear Complex with an additional 3000 MW, resulting in vast amounts of heated effluent to be discharged into Lake Huron. As indicated in Section 4.4, the combined thermal plume from the Bruce facility is estimated to be 50% larger as a result of refurbishment and restart of four existing reactors. The thermal plume effects assessment was based on four fish VECs: smallmouth bass, lake whitefish, spottail shiner and deepwater sculpin (*Myoxocephalus thompsoni*) (Golder, 2005c).

Smallmouth bass are known to utilize the Bruce A discharge channel for spawning. Limited MWAT exceedances for both existing (i.e., two units operating) and predicted conditions (i.e., four units operating) were determined for juvenile and adult stages in the discharge channel during mid-summer. There were no other exceedances observed in any other applicable habitat area or depth zone modeled. The exceedances may last up to four weeks during mid-summer when only adults and juveniles would be present. These mobile fish may choose to relocate to nearby habitat where temperatures are at or near their optima.

Potential spawning habitat for lake whitefish has been identified on nearby shoals with Loscombe Bank potentially directly influenced by the discharge plume. Based on the existing and predicted modeled temperature data and *in-situ* continuous temperature monitoring data, exceedances of the MWAT criteria were noted during spawning, just after spawning with the initial deposition of eggs and during late winter/early spring just prior to hatching. These exceedances were also evident at a control site and shoals likely unaffected by the plume and reflect the normal cooling and warming conditions in Lake Huron during late fall and early spring. The operation of the BANRP will result in typical increases in water temperatures of 0.1 degree with a maximum of 0.3 degrees at the spawning shoals. Based on this assessment, it was concluded that thermal discharge may have a minor adverse (not significant) effect on lake whitefish and their habitat.

Spottail shiner is a common forage species in the nearshore area and discharge channel. Based on existing and predicted water temperatures, MWAT exceedances were determined for juvenile and adult stages in the discharge channel, as well as for spawning in the discharge channel and Baie du Doré to the north. There is no evidence of spottail shiner spawning or YOY in the discharge channel, with suitable spawning habitat present south of the Bruce Nuclear Complex. While spottail shiner adults and juveniles have been documented in the discharge channel, as in the case of smallmouth bass, they may choose to relocate to nearby habitat where temperatures are at or near their optima.

Deepwater sculpin reside in the deeper offshore waters of Lake Huron beyond the thermal plume changes associated with the BANRP. Although there is potential inshore movement by larvae, no change in temperature above ambient is predicted in the presumed nearshore habitat areas where deepwater sculpin larvae may be found.

In addition, the Port Elgin and Southampton Water Supply Plants (WSPs) located north of Bruce A will likely be affected by the combined operations of Bruce A and Bruce B, with elevated water temperatures of 2°C or more occurring 10 to 12% of the time under warmwater conditions. For Kincardine WSP located south of Bruce A, an elevated water temperature of 2 degrees or more above ambient would occur less than 1% of the time under warmwater conditions.

6.0 MITIGATION STRATEGIES

As indicated in Section 5.1, numerous studies have suggested that thermal discharges have the potential to negatively influence the aquatic environment; however, most studies have demonstrated that any negative effect is localized. Moreover, a number of mitigation strategies should be considered to lessen negative effects. These strategies are outlined below.

Power Plant Siting

As indicated in Section 5.1, the siting of a new power plant can be the most important environmental consideration in minimizing detrimental thermal effects within the plume and the entire receiving aquatic environment. The location of the thermal outfall and mixing plume should avoid environmentally significant features, e.g., areas of high biological productivity, presence of unique ecological communities, significant spawning and nursery habitat utilization, and the presence of VECs and species at risk.

Intake Design/Location

Intake location should take into consideration opportunities to take advantage of cooler waters of the hypolimnion during summer thermal stratification conditions in order for the thermal discharge to be minimally warmer than the epilimnion. As indicated in Section 4.2, the PEC water intake located within the Ship Channel seawall was designed to take advantage of the cooler, deeper waters during summer thermal stratification, thereby ensuring a cooler discharge temperature. In fact, under typical summer conditions and normal plant operations (i.e., ΔT of 5.5 degrees), the water temperature within the Outer Harbour receiving the cooling water discharge was not expected to change appreciably, and may be cooler than the ambient epilimnion.

However, the siting of a new offshore intake to take advantage of thermally stratified summer conditions in lentic systems should consider the following potential constraints:

- the siting may trigger the *Canadian Environmental Assessment Act* under the *Fisheries Act* and/or *Navigable Waters Protection Act*, likely resulting in a more formal EA process of significantly longer duration than the provincial EA process;
- potential for structural damage, e.g., due to vessel anchor dragging; and
- sediment quality constraints resulting in sediment resuspension and turbidity generation during intake construction, as well as potential release of associated contaminants.

Power plant siting should avoid significant fish habitat and other environmental features in the vicinity of the intake, thereby minimizing the potential for fish impingement and entrainment.

Outfall Design/Location

As indicated in Section 4.0, increasing discharge velocity, e.g., with the use of an offshore diffuser outfall design, results in an increase in cooling water entrainment and promotion of dilution, as well as increase in vertical mixing. However, in comparing alternative outfall structures for the BBPS, BEAK (2001) demonstrated, based on CORMIX modelling results, that although an offshore diffuser outfall provided somewhat greater thermal plume attenuation compared to shoreline alternatives, the difference in thermal regimes would likely not be measurable in the field.

The criteria listed above for the siting of new offshore intakes should also have been considered in the outfall siting, i.e., avoiding biologically important areas such as spawning beds, regulatory considerations, structural drainage potential and sediment quality constraints.

Use of Quenching Water

Quenching involves the mixing of water at ambient temperature with the plant cooling water prior to discharge to maintain discharge temperatures at or below the stipulated effluent temperature objective. This tempering process is usually required during those times in the summer when ambient temperatures are high. For example, the BBPS uses quenching water in July and August when temperatures in the Detroit River reach more than 22ºC to meet its effluent temperature objective of 32ºC (see Section 7.1). Similarly, the TCP will require the use of quenching water to meet its maximum effluent temperature of 32ºC during the summer period. The use of quenching water should be evaluated in the context of possible increases in effects of fish entrainment and impingement at the intakes. Under some circumstances, added damage from entrainment and impingement may exceed the benefit to fish from lowering discharge temperatures.

Power Production Reduction

A decrease in power production will result in lower cooling water discharge temperatures (assuming all intake pumps continue to operate) or in reduced volumes of water being pumped for cooling. As indicated in Section 2.9.3, Special Operating Requirements were included in the C of A for the TCP requiring plant shutdown during the Canal navigation season when the cooling water temperature exceeds 33ºC for any continuous period of 6 h, which could occur under low-flow conditions. For the non-navigation (winter) season, plant shutdown is based on a decrease in Welland Canal Power House flow to less than 6 m^3/s or a temperature difference of greater than 2 degrees between the intake and upstream canal water temperatures, indicating entrainment of recirculated heated water.

As part of the EA and license review of the Selkirk GS, which discharges cooling water to Cooks Creek, Manitoba Conservation has stipulated that power output at the plant should be reduced if the prevailing weekly average temperature in the Red River (the cooling water source) approaches the designated prevailing MWAT value (UMA *et al.*, 2005). The power output must be reduced to a level at which it can be demonstrated that heat loading from Cooks Creek into the Red River is not causing the downstream prevailing weekly average temperature of the Red River in the fully mixed zone to exceed the MWAT value. Power plant generation must be discontinued if the prevailing weekly average temperature of the Red River intake cooling water is equal to or exceeds the designated MWAT value for a prevailing month. Power generation reduction should be gradual to minimize the temperature decline rate of water in Cooks Creek with designated temperature decline rates during ice-cover and open water conditions at a monitoring location approximately 1 km downstream of the outfall.

For the BANRP, Golder (2005c) undertook a literature review to identify and evaluate potential technologies and/or methods for lowering discharge temperature or reducing the size of the thermal plume. Eight alternative technologies were initially identified as follows:

- redirecting the discharge away from sensitive fish habitat (in this case, Loscombe Bank);
- constructing a submerged discharge;
- extending the cooling water intake further offshore to draw cooler water;
- increasing the cooling water flow rate;
- diverting the cooling water to a cooling pond;
- diverting the cooling water to a mechanical draft cooling tower;
- diverting the cooling water to a natural draft cooling tower; and
- diverting the cooling water to a dry cooling tower.

Subsequent screening determined that there are two potential beneficial technologies that could be considered at Bruce Nuclear Complex, i.e., redirecting the discharge or increasing cooling water flow.

In other cases, two or more technologies may be combined as a hybrid system, in order to mitigate potential adverse effects.

Additional feasible mitigation technologies for existing facilities include:

- extending the cooling water discharge further offshore where water temperature tends to be lower;
- increasing flow (volume and velocity) over condensers;
- increasing discharge velocity by reducing the size of the discharge channel opening;
- decreasing discharge velocity by widening the mouth of the discharge channel;
- enhancing cooling of discharge water by tempering water or installing fountains or sprays in the discharge channel;
- increasing the number of planned outages in a year;
- increasing the number of planned outages in a year while maintaining the cooling water pumps in operation;

- de-rating power production during sensitive periods of fish embryo development;
- alterations to existing diffuser technology (i.e., additional mitigative adaptations may be necessary);
- extending the intake deeper into the waterbody where temperatures are lower.

The selected mitigation measures should evaluate the potential increase in impingement and entrainment of fish that could result from the implementation of the preferred thermal mitigation technology or technologies. Some mitigation options that could effectively reduce the impacts of thermal discharges may result in increased impingement and entrainment effects due to increased water uptake from the lake. As a result, the evaluation of thermal mitigation options should also evaluate the potential for the given technology to increase impingement and entrainment effects and whether there is a need to implement mitigation options for the reduction of impingement and entrainment effects as well.

7.0 MONITORING PROGRAM

Although efforts are made to be conservative in assessing the environmental effects of thermal discharges prior to operation based on facility design/operational information, thermal plume modelling results, site-specific/local/regional environmental baseline data and the thermal tolerance and resistance of specific endemic species, there is always a degree of uncertainty. Operational monitoring provides confirmation that the predictions are accurate. Sections 7.1 and 7.2 provide the findings of operational monitoring of the BBPS thermal plume and its effects on benthic macroinvertebrate communities, respectively. Section 7.3 provides the findings of an operational field study to assess potential effects of thermal discharge from the Iroquois Falls GS on lake whitefish. These examples present a variety of useful monitoring tools/protocols for the assessment of thermal effects. Additional information on monitoring protocols can be found in Environment Canada's Environmental Effects Monitoring (EEM) Technical Guidance Documents for Pulp and Paper Mills and Metal Mines.

7.1 BBPS Temperature and Thermal Plume Monitoring

As part of its due diligence, BBPS recognized the need for thermal plume delineation and a monitoring program to provide operational data regarding the temperatures and compliance of BBPS for its thermal discharge to the nearshore Detroit River.

Continuous water temperature monitoring was undertaken during the fall of 2004 within the mixing zone, and during the summer of 2005 from upstream of the BBPS intake to approximately 1.3 km downstream of the outfall (Fitchko, 2005). Three thermal plume delineation surveys were also undertaken during the 2005 temperature monitoring program.

Based on the comprehensive continuous water temperature monitoring study undertaken during the summer of 2005, maximum monthly temperature differences between the upstream intake location and two stations nearest to the outfall averaged 5.46, 5.34, 5.33, 5.58 and 5.82 degrees in June, July, August, September and October, respectively, well below the maximum allowable ΔT of 10 degrees. The slightly lower temperature differences in July and August were due to the use of quenching water. The maximum effluent temperature of 34.0°C was never exceeded, whereas the effluent temperature objective of 32°C was slightly exceeded for generally very short periods of time on seven and two days in July and August, respectively. However, for the intervals of temperatures above 32.0°C, the IDH value was well below 1.0 for any 24-h period.

The thermal plume could be visibly demarcated during the three summer surveys due to density differences. The lateral extent of the thermal plume in August 2005 was about two times greater than in June and October, reflecting lower heat dissipation capacity due to higher ambient temperatures.

Based on conservative modelling using the CORMIX model, BEAK (2001) predicted that the temperature difference within the thermal plume will be reduced to within 2°C of ambient within approximately 1000 m. The width of the thermal plume at 2°C was estimated to be less than 140 m. The findings of the summer 2005 thermal plume delineation studies confirmed these predictions; however, the macroinvertebrate survey described below suggests that the plume extended farther into the river than anticipated.

7.2 BBPS Operational Benthic Macroinvertebrate Study Findings

Possible changes in the macroinvertebrate assemblage of the Detroit River were the main concern at the BBPS site. As indicated in Section 3.1, the pre-operational benthic survey plan for the BBPS was based on the predicted thermal plume configuration. For the operational surveys, transect locations were consistent with those of pre-operational surveys (Fitchko, 2006a,b). However, during the summer operational surveys, the sampling stations were located at slightly greater water depths, i.e., 1 to 1.1 m, 1.4 to 1.6 m and 1.7 to 2.1 m, to facilitate sediment sample collection and to take into account the wider expanse of the thermal plume offshore, which temperature monitoring showed was still in contact with the riverbed (Fitchko, 2005).

Ten benthic macroinvertebrate community metrics were analyzed for differences within years as well as for differences between years. These metrics included total abundance, number of taxa, Shannon-Wiener diversity, evenness, richness, HBI, WQI, % Oligochaeta, % Ephemeroptera and % Chironomidae. The depth of sampling was ignored in order to provide within-site replication.

Within-year differences between sites included a one-way ANOVA with a post-hoc test involving comparisons to the reference site (Dunnett Test). The benthic data were also analyzed according to a before-after-control-impact (BACI) analysis (Smith *et al.*, 1993). The BACI test is incorporated into the *Phase*Site* interaction term of the ANOVA. The *Phase* term refers to the different time periods of the sampling, with pre-operational and operational phases investigated. This approach tests the null hypothesis that:

$$H_0\text{: (Discharge-Control)}_{before} = \text{(Discharge-Control)}_{after}$$

All analyses were conducted using SPSS V11 for Windows.

For the winter pre-operational (2002) and operational (2004) survey data, there was a significant difference only for total abundance at the 1000-m downstream transect, as total density values increased markedly from 2002 to 2004, likely reflecting a relocation of one of the sampling locations and the finer substrates sampled at the other two locations (Fitchko, 2006a). As a result, no further monitoring was recommended.

For the summer pre-operational (2003) and operational (2005) survey data, there were significant differences for only the relative abundance of Ephemeroptera and HBI (after one outlier was removed) (Fitchko, 2006a). Mayfly nymphs were present in 2005 at the upstream reference after being absent in 2003, whereas their relative abundance decreased at the near-field transect from 2003 to 2005. The decrease in mayfly nymph abundance may be attributed to the increased water temperatures due to BBPS cooling water discharge.

The HBI results indicated that whereas the reference benthic community changed only slightly, the benthic communities at the near-field and especially the 100-m downstream transects shifted to more tolerant assemblages. Cole and Kelly (1978) postulated that the production per unit of biomass of benthic macroinvertebrates would be stimulated by heated discharge as long as the maximum temperature remains below 34°C. Based on comparison of the pre-operational and operational data, there appears to be a trend towards increasing tubificid oligochaete numbers, particularly in the summer of 2005, which would result in lower HBI values. As a result, another benthic macroinvertebrate survey was recommended for the summer of 2006 to further assess this apparent trend towards increasing tubificid oligochaete numbers.

Based on the 2006 survey findings, there were no significant differences in benthic macroinvertebrate community metrics with the exception of HBI (Fitchko, 2006b). The trend towards increasing tubificid oligochaete numbers continued in 2006, resulting in lower HBI values. It was concluded that the intermittent, slightly higher water temperatures in the near-field due to cooling water discharge may be stimulating increased abundance of the more tolerant tubificid oligochaetes. However, the numbers of tubificid oligochaetes at the upstream reference location also increased in 2005 and 2006 compared to 2003. Based on these findings, it was recommended that no further benthic macroinvertebrate community surveys be undertaken except when an IDH parameter value equals or exceeds 1.0 in any 24-h period, as stipulated in the C of A (see Section 2.9.3).

7.3 Iroquois Falls GS Lake Whitefish Monitoring

Some monitoring studies have focused nearly exclusively on specific aspects of one biotic component of special concern. Such is the case for the lake whitefish in the Abitibi River that might be affected by the Iroquois Falls GS. The Iroquois Falls GS, which commenced operations in September 1996, is located on the west bank of the Abitibi River approximately 1.7 km downstream of the Iroquois Falls hydroelectric dam. The nominal capacity of the station is 75 MW utilizing a once-through cooling system and discharging 218 000 m^3/d at a maximum of 10ºC above intake temperature. Discharge is via a multiport diffuser on the river bed. Previous fisheries surveys determined that an apparently stable population of lake whitefish utilizes spawning habitat present in the reaches between the Iroquois Falls GS and the hydroelectric dam (North Shore, 1990; BEAK, 1996a).

To assess the potential effect of the Iroquois Falls thermal plume on spawning whitefish and overwintering embryos, BEAK (1996b) undertook the following:

- gillnet sampling with sufficient effort and appropriate strategy to determine the relative importance of spawning habitat in the study reach;

- a thermal plume delineation study to determine the spatial extent of the plume and its spatial effect on whitefish spawning habitat; and
- a long-term, *in situ* temperature monitoring system to determine potential thermal effects on whitefish spawning and overwintering embryos.

Gill nets were set for approximately 4 h during the day to minimize fish mortality, with a total of 35 4-h sets during the spawning period (01–07 November). Thermal plume delineation was based on *in situ* flow-through fluorometer measurements of a conservative dye (Rhodamine WT) pumped into the cooling water prior to discharge (16 October) and *in situ* continuous temperature monitoring using TidbiT loggers established at four locations and up to three depths per location. Due to the extremely high turbidity of the Abitibi River, substrate observations based on hardness and texture were made using a copper pipe probe.

The dye study indicated a buoyant plume approximately 500 m downstream and vertical mixing within 1 km. Ambient water temperature was 9ºC at all sites, except in the diffuser upwelling where a slightly higher temperature of 9.5ºC was detected.

The continuous temperature monitoring data indicated that the differences between the mean temperature at the upstream reference and a surface monitoring location approximately 100 m downstream of the diffuser was 0.4ºC.

Based on the fish catch data and substrate mapping, the spillway of the hydroelectric dam upstream of the Iroquois Falls GS appeared to be the focal area for whitefish spawning in this section of the Abitibi River. Spawning was also evident, to a lesser degree, in the immediate vicinity of the Iroquois GS diffuser, i.e., along the west bank of the river from approximately 40 m upstream to 75 m downstream of the diffuser. The fish utilizing this spawning habitat would be exposed to fluctuating temperature increases on the order of 0.4 to 0.8ºC. This utilization indicated that there was no avoidance of the plume by spawning whitefish. Based on available literature, exposure to temperature elevations of 0.1 to 0.8ºC would have no effect on incubating whitefish embryos. For example, Griffiths (1979b) determined that a sustained increase in temperature of 10ºC above ambient will kill 100% of incubating whitefish embryos, but that continuous elevation of 1 to 2ºC and periodic elevations of 2 to 4ºC has no significant effect. BEAK (1996b) concluded that the Iroquois Falls GS thermal plume had no effect on whitefish movements, spawning habitat utilization and incubating whitefish embryos.

8.0 SUMMARY

Numerous studies have suggested that thermal discharges have the potential to negatively influence the aquatic environment; however, most studies have demonstrated that any negative effect is localized. This localized effect may be small in relation to a large aquatic system (river or lake) but may be large relative to a small aquatic system. Thermal discharge may result in localized alteration of organism physiological and behavioural processes, primary and secondary production, and time-varying species composition.

Site-specific factors of the facility, and of the receiving water environment and the biota it contains, largely determine the temperatures, thermal plumes and biological impacts of temperature changes. Nonetheless, there are standardized, and often quantitative, methods and approaches for evaluating both temperatures and biological impacts. Once the patterns of temperatures in the plume and receiving water body have been estimated through predictive analyses (models), the biotic categories most at risk can be identified and the impacts forecast. Pre-operational baselines can be measured through field studies that are compared to subsequent field studies after operation commences. This document has provided several Canadian examples illustrating site specificity and analysis methods that should be the basis for any evaluation.

As presented in this guidance document, the assessment of potential effects of a thermal discharge is based on the following:

- facility design/operational information;
- data on the thermal and limnological/hydrological regime of the intake and receiving water areas;
- thermal plume modelling results;
- site-specific information on benthic macroinvertebrate communities (shoreline discharge only);
- site-specific information on fisheries resources, fish habitat and important locations (such as spawning sites and migration routes);
- available information on other biological resources and nearby water uses; and
- the thermal tolerance and resistance of specific endemic species, particularly VECs, including lethal levels for high and low temperatures, behaviourally preferred temperatures, optimal temperatures for growth and reproduction, some of which define regulatory limits such as warm-season and cold-season MWAT.

Although efforts are made to be conservative in assessing the environmental effects of thermal plumes, there is always a degree of uncertainty. Operational monitoring of the thermal plume and its effects on benthic macroinvertebrate communities (usually shoreline discharge only) and fisheries resources provides an opportunity to confirm (or not) that the predictions are accurate.

A number of mitigation strategies can be considered to lessen negative effects of thermal discharges, including prudent power plant siting, intake and outfall design/location, use of quenching water and reduction of power production.

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APPENDIX: 2016 Addendum

UPDATE TO ENVIRONMENT AND CLIMATE CHANGE CANADA'S GUIDANCE DOCUMENT: ENVIRONMENTAL EFFECT ASSESSMENT OF FRESHWATER THERMAL DISCHARGE

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UPDATE TO ENVIRONMENT AND CLIMATE CHANGE CANADA'S GUIDANCE DOCUMENT: ENVIRONMENTAL EFFECT ASSESSMENT OF FRESHWATER THERMAL DISCHARGE

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Note: the number system is meant to fit with the existing guidance document

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5.1.6 Fish

Temperature has been documented as one of the most important factors in the life of fish. Most species of fish are "classic" ectotherms and their body temperature is within the range of 0.1°C to 1°C warmer than the water temperature (Beitinger *et al.,* 2000). Metabolism, respiration, activity, food consumption, growth, reproduction, and longevity are all affected by temperature.

For each fish species, there is a temperature range that is optimal for life functions. Temperatures outside this range affect fish function and, depending on the amount of time spent at these temperatures, survival. The highest and lowest temperatures of a fish's tolerance are the acute lethal temperatures (Golovanov, 2012).

Changes in water temperatures can be due to various factors including solar heating, storms, dams, thermal effluents, changes in riparian vegetation, changes to adjacent land use, and climate change. Changing temperatures have been found to alter fish communities and trophic relations within the receiving waters (Luksiene *et al.,* 2000; Golovanov, 2013)

5.1.6.1 Field Studies

Potential changes caused by thermal effluents in an aquatic system are complex in nature. They may involve aspects that may not be predicted or cannot be fully duplicated in a laboratory environment. In order to gain a more conclusive understanding of thermal discharge effects, some in *situ* studies have been completed. Fish responses documented in these studies have been highly variable between locations, seasons, species and life stages. The results of a few recent studies are presented below. Investigations into specific responses are discussed in more detail within following subsections of this report.

McKinley *et al.* (2000) studied Smallmouth Bass (*Micropterus dolomieu)* in a thermal plume that discharges to Lake Erie. They found that males guarding nests remained in the heated discharge despite temperature fluctuations of up to 16°C, whereas non-nesting males and females moved in and out of the discharge. Nesting males did not leave the canal even when temperatures approached the upper tolerance temperatures. At the same discharge, Cooke *et al.* (2003) studied parental care and reproductive success in Smallmouth Bass compared to non-temperature influenced populations. Temperatures during the study were 5 to 10°C higher in the plume relative to ambient and spawning in the plume began approximately a month earlier. They found that parental care rates in the plume were higher but the nesting fish did not follow conventional behavioral patterns. Relative to published values the success rates of reproduction were high and they concluded that Smallmouth Bass were capable of adjusting energetic expenditures.

Many large scale multi-species studies have been completed in Europe where long term data have been collected. Although not all of these species occur in Canada, results are relevant as they occupy similar thermal ranges and climate conditions.

Sandstrom *et al.* (1995) examined Perch (*Perca fluviatilis*) populations from 1978–1990 in an artificial enclosure exposed to thermal effluent on the Baltic coast. Following the initiation of discharge, the Perch population showed improved recruitment but increased mortality of larger adults. Fish began maturing very early with reduced gonad sizes. Following spawning fish condition dropped to very low levels most likely causing the increased mortality seen. The surviving fish not only showed a reduced fecundity in future years but also delayed spawning by one or more years. In 1997, Sandstrom *et al.* (1997) reported on the survival in Perch exposed to the same discharge, from spawning through embryo stages. Advanced spawning of approximately one month and a prolonged spawning period was reported. They also documented high fertilization rates, although mortality rates were high due to egg strand disintegration. Mortality was attributed to high temperatures during the final maturation of the gonad influencing the matrix of the egg strand. They suggested that there is a conflict between reproduction and preferred temperatures in some temperate fish and that fish may be attracted to warmer temperatures, at the expense of reduced reproduction.

In 2000, Luksiene *et al.* (2000) reported on the effects seen in thermal effluent areas in Sweden and Lithuania. They found that in the temperature influenced areas female Perch, Roach (*Rutilus rutilus*) and Pike (*Esox Lucius)* had negatively influenced gametogenesis and asynchronous egg cell development; whereas temperature impacts to summer spawning species were much less. Silver Bream (*Blicca bjoerkna*) showed no significant impacts and an additional spawning period was documented for Ruffe (*Gymnocephalus cernuus*). They also confirmed the conflict reported by Sandstrom *et al.* (1997) between temperature preference behaviour and the temperature needed for optimal reproduction in some temperate fish.

5.1.6.2 Thermal Guilds

Fish that have similar temperature preferences have been grouped into coldwater, coolwater, and warmwater guilds. Eakins (2016) defines thermal guilds as species that are best adapted to, prefer, or usually occur at water temperatures that are less than 19^{o}C for coldwater, 19 to 25^{o}C for coolwater, and greater than 25^{o}C for warmwater species during summer months. Fisheries and Oceans Canada (DFO) uses the same temperature ranges for classification in a report that includes Canadian fish species temperature guilds (Coker *et al.,* 2001). The Electric Power Research Institute (EPRI, 2011) has not defined the temperature range for each guild but has grouped species based on tolerance and acute toxicity to high temperatures.

The species that are classified differently between these three sources are summarized below. Thermal guilds are provided for all species in **Tables 10** and **11** (see **Sections 5.1.6.3** and **5.1.6.5**).

Species	Eakins, 2016	Coker *et al.*, 2001	EPRI, 2011
Lake Sturgeon	Coolwater	Cold/Coolwater	NA
Central Stoneroller	Coolwater	Cool/Warmwater	NA
Emerald Shiner	Coolwater	Coolwater	Warmwater
White Sucker	Coolwater	Coolwater	Warmwater
Northern Pike	Coolwater	Coolwater	Warmwater
Muskellunge	Warmwater	Warmwater	Coolwater
Brown Trout	Coldwater	Cold/Coolwater	Coldwater
Burbot	Coldwater	Cold/Coolwater	NA
Threespine Stickleback	Coolwater	Coldwater	NA
Mottled Sculpin	Coolwater	Coldwater	Coldwater
Smallmouth Bass	Coolwater	Warmwater	Warmwater
Black Crappie	Coolwater	Coolwater	Warmwater
Greenside Darter	Warmwater	Cool/Warmwater	Warmwater
Rainbow Darter	Coolwater	Coolwater	Warmwater
Johnny Darter	Coolwater	Coolwater	Warmwater
Yellow Perch	Coolwater	Coolwater	Warmwater
Sauger	Coolwater	Coolwater	Warmwater
Walleye	Coolwater	Coolwater	Warmwater

NA – species not included in report.

5.1.6.3 Thermal Assessment Methods

There are a variety of methods to determine endpoints that describe the temperature tolerances of fish (**Figure 1**).

This includes acute and chronic laboratory-based toxicity tests. Acute toxicity is evaluated by methods that either abruptly transfers animals or a change in temperature by a degree every hour or less. Chronic toxicity is evaluated with a gradual change in temperature by 2 degrees or less every day (Golovanov, 2012).

Although most *in situ* mortality is due to a rapid decrease in temperature, most toxicity studies examine tolerance to high temperatures (Beitinger *et al.,* 2000). Previously, this guidance presented the upper incipient lethal temperature (UILT or LT50) and the critical thermal maximum (CT Max) as acute toxicity endpoints representing the upper temperature threshold.

The updated guidance presents these endpoints, as well as the chronic lethal maximum (CLMax) and the maximum weekly average temperature (MWAT) as potential values for the upper temperature threshold for a variety of species in Canada. **Section 5.1.6.5** discusses cold shock and provides data associated with lower temperature thresholds (**Table 11**). The information in **Tables 10** and **11** excludes references older than 1970 and also only includes data from peer-reviewed published journals as well as a post-secondary thesis.

The UILT and CTMax are the most common measures of acute toxicity and are used to calculate benchmarks in many jurisdictions. The CLMax has been included as a chronic toxicity endpoint, and although likely the most realistic endpoint in terms of its experimental methods simulating *in situ* thermal situations, it is not as commonly measured. The MWAT, although not an endpoint directly determined from a toxicity test, provides a chronic estimate of thermal tolerance. MWAT is commonly used in the US, but its use has been criticized for not protecting cold water species (McCullough, 2010). Measurements of temperature change, Delta T, are used to monitor thermal discharges. The thermal preferences and toxicity values of fish are used to calculate the acceptable change of temperature. The terms referenced above are described in more detail below.

Acclimation temperature influences the test results. As much as a 10^{o}C difference has been seen in the same species for UILT test with different acclimation temperatures (EPRI, 2011). Therefore, acclimation temperatures were presented where available in **Tables 10** and **11** and benchmarks should be based on results of tests that used acclimation temperatures that are representative of the receiving waters.

Incipient Lethal Temperature (ILT) – The ILT is the temperature lethal to 50% of the test population for a sustained period (EPRI, 2011; Golovanov, 2012). It is considered to be an endpoint for an acute toxicity test. The upper (UILT) measures the highest temperature, whereas the lower (LILT) measures the lowest temperature, where mortality occurs. For testing purposes, fish are placed in aquaria with a range of temperatures; either a series of high temperatures to evaluate the UILT or a series of low for the LILT. The time to mortality (the endpoint) is recorded for each fish. The regression of percent mortality per treatment and temperature for a given time interval is used to determine the temperature in which 50% of the population survived. The ultimate UILT (UUILT) is determined by increasing the acclimation temperature in progressive tests until there is no change in the result (Golovanov, 2012). This was the preferred method to determine maximum fish tolerance through to the 1960s.

Test results associated with measuring the ILT can be influenced by a number of factors. For example, fish can be stressed by this method from the translocation to aquariums with different temperatures. Also, this test does not represent in-field conditions very well, as temperature changes in the environment are rarely instantaneous *in situ*. Moreover, the time interval to evaluate ILT is not standardized which complicates comparison of results with different test lengths. In addition, the measurement of ILT is labor and resource intensive, as a number of fish and aquaria required, as well as constant observation for mortality.

In some studies, the ILT method has been modified such that all aquaria are at the acclimation temperature to start. Once fish have acclimated, the temperatures change in each tank until the desired temperature series has been established. The fish are then held in the aquaria at a constant temperature until test end (mortality or maximum time). This eliminates the stress of translocation and shock (Beitinger *et al.,* 2000; Todd *et al.,* 2008).

Critical Thermal Methodology (CTM) – The CTM is the point at which locomotary movement becomes disorganized and the animal loses its ability to escape from conditions that may ultimately lead to its death (EPRI, 2011). It is considered to be an acute toxicity test. In this test, after acclimation, the temperature is changed at a constant rate (i.e. 1^{o}C/min or 1^{o}C/hr) and changes in fish behavior are noted. Endpoints include loss of equilibrium (LOE; turning to side or belly-up), onset of muscle spasms (OS), termination of bronchial-operculum movement (OV – opercular ventilation), or prodding with a glass rod without response (PGR). Fish in these experiments recover once returned to their acclimated temperature, but some tests are run until death (DP – death point). The temperature reported is usually a mean response of the test fish. Effects associated with a temperature increase are reported as the CTMax, whereas the CTMin is the endpoint associated with effects from a temperature decrease.

Fewer fish are required with this method and this method was preferred over ILT since the 1990s. This is also the preferred method when test fish are in limited quantity, typically with species at risk (Beitinger *et al.,* 2000) due to recovery after testing and fewer numbers needed.

The rate of temperature change in a given experiment can influence results. If the change is too fast, there will be a lag time before response and a higher water temperature will be recorded. The size of the fish (surface area to volume) will also affect how quickly internal temperatures adjust to external ones. It has been recommended that 0.3^{o}C/min be the standard for determining CTM (Becker and Genoway, 1979). A study by Kilgour and McCauley (1986) showed that heating rates of 0.1 to 0.3^{o}C/min are gradual enough to avoid any significant lag in internal body temperature, but are fast enough to prevent acclimation.

Although the CTM occurs at different temperatures in different species, the locomotary response (i.e. onset of spasms) is the same across a diversity of taxa. For these reasons, CTM is an excellent index and standard for evaluating the thermal requirements and physiology of an organism (Lutterschmidt and Hutchison, 1997; Wilkes, 2011). Although LOE and OS are the most commonly used endpoints, in some species, (e.g. carp), it is difficult to assess these endpoints. Cessation of respiratory movement or prodding is more appropriate for these species. In addition, when evaluating a CTMin the OS endpoint often cannot be observed. Therefore, experimenters need to use the most appropriate endpoint for their species and test (Beitinger *et al.,* 2000). This can make test to test comparisons difficult when endpoints differ.

Chronic Lethal Methodology (CLM) – The CLM is similar to the CTM, except that the rate of temperature change is slower (1-2^{o}C/day). The rate is slow enough that the fish constantly acclimatize to the changing temperature (Golovanov, 2012). The CLM method likely provides a more realistic estimate of upper lethal temperatures since the fish are subjected to temperature changes that better approximate what fish experience on a daily basis. Sublethal effects, such as reduced feeding rates, weight loss, or development can also be addressed. CLMax is a measure of lethality as a result of increasing temperature whereas CLMin is a measure of lethality from decreasing temperatures.

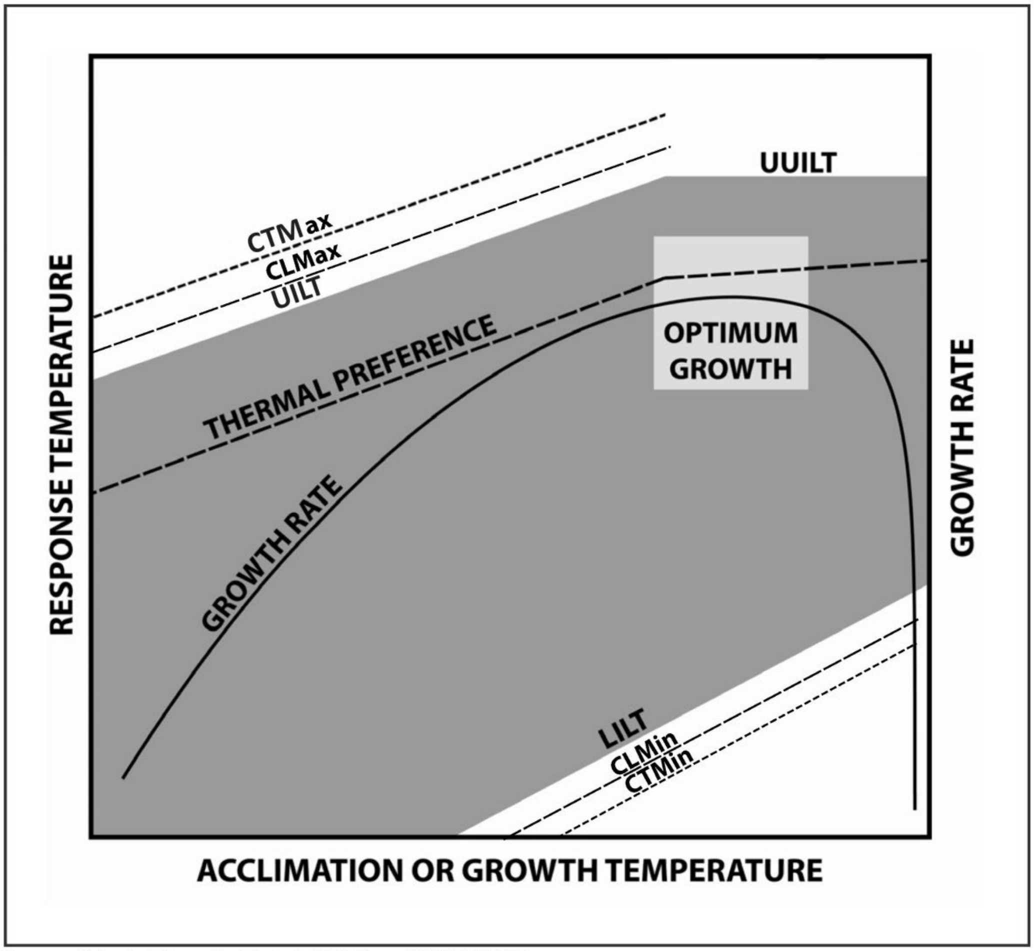
Although more realistic, this method takes more time to complete then the ILT and CTM methods. Instead of hours, as in a CTM, the CLM experiment can take several weeks.

Differences between ILT, CTM, and CLM

The three tests yield different endpoint estimates. ILT and CLM provide a lethal endpoint whereas CTM represents (typically) a non-lethal locomotary change. ILT involves abrupt immersion of multiple fish into test aquaria, whereas CTM and CLM are constant temperature changes with few fish. ILT takes between a couple days to a week, CTM a couple hours to a day, and CLM a couple days to multiple weeks. ILT results are calculated from a dose-response curve for multiple treatments, where as CTM and CLM results are descriptive statistics (i.e. mean).

CTMax results are usually greater than UILTs because CTM often overshoots the temperature that the change would be observed due to lag in response and the constant rate of temperature change. (Beitinger *et al.,* 2000). The slower temperature change of the CLM allows the temperature to exert its effects on the fish more thoroughly at each temperature than the CTM, which often means the results of the CLMax are lower than the CTMax (Beitinger *et al.,* 2000; Golovanov, 2013). Results of these test for the same species are usually UILT < CLMax < CTMax.

The CLM test likely approximates daily changes in water temperature in temperate regions, and is the most realistic estimate of temperature tolerance (EPRI, 2011). CLM also weakly depends on season, whereas CTM has been found to largely depend on season (Golovanov, 2013).



Source: Adapted from Todd et al. (2008) and USGS (2009)

Figure 1: Temperature Relations of Fish

Maximum weekly average temperatures (MWAT) – MWAT can be measured in the waterbody and determined for the fish species. This is a measure of upper temperature limit for long-term exposure (chronic toxicity). The fish MWAT is calculated as one third of the range between the optimum growth temperature and the ultimate upper incipient lethal temperature (UUILT). The water MWAT is the average of the maximum water temperature for 7 days (McCullough, 2010). A water MWAT that is equal to the fish MWAT is assumed to be protective of that fish species. The US EPA uses the MWAT statistic as the standard method for chronic assessments (US EPA, 2012). Recently the river MWAT has been modified to the 7 day mean of the daily mean water temperature for development of temperature criteria for fish protection in Colorado (Todd *et al.,* 2008).

The MWAT has been criticized that it does not adequately protect cold water fish (McCullough, 2010). Growth rate curves vary in shape between species and can also vary between life stages. As an example, **Figure 2** shows hypothetical peaked and broad curves for Bull Trout growth rates. The optimum temperature for growth is 13.2°C (point a) and the UUILT for Bull Trout is 20.9°C. This results in a fish MWAT of 15.8°C, which are points b for the broad and c for the peaked curves. If the fish's growth rate follows the peaked curve, at the MWAT its growth rate will be 10% less than the broad curve growth rate.

In reality temperatures fluctuate and the average water temperature does not present the extreme temperatures that may occur. If the water MWAT was 15.8°C, temperatures could fluctuate, for example, between 13.2°C and 18.5°C, resulting in growth rates between points b and d on the broad curve or between points c and e on the peaked curve. Most species in the literature follow the peaked curve, including both the warm- and coldwater guilds. Dial fluctuations can vary even more than the example presented and reach the UUILT if the waterbody is managed to the MWAT, resulting in massive mortality and extirpation (McCullough, 2010).

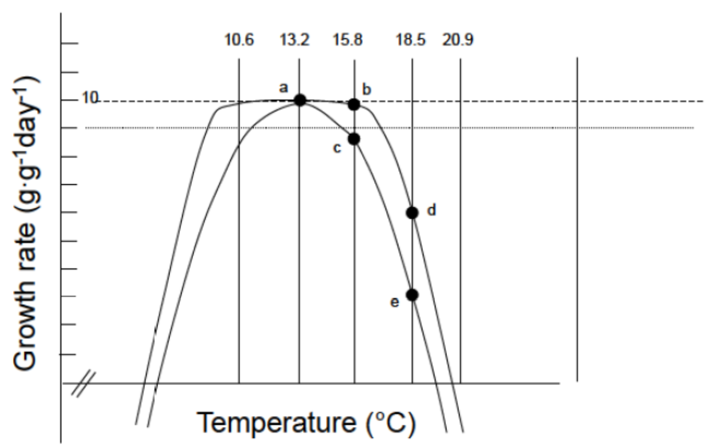


Figure 2: Two hypothetical growth rate patterns for Bull Trout to demonstrate problems associated with MWAT (source: McCullough, 2010)

Delta T – Delta T is simply defined as a change in temperature. In the context of a thermal discharge Delta T would be the difference in water temperature between the thermal effluent and ambient water temperature. Although simple in concept, it is difficult to estimate the three-dimensional extent of the thermal plume in practice. Detailed site-related information, such as knowledge of seasonal current and wave height patterns and other coastal features and a clear understanding of "true" background conditions that are not influenced by other factors such as thermal inputs from nearby streams or rivers or sheltered harbors, is required. Technical guidance on plume dispersion and how to conduct a plume delineation can be found in Environment Canada's "Revised Technical Guidance on How to Conduct Effluent Plume Delineation Studies" (2003).

Currently many jurisdictions use a temperature water quality guideline based on a Delta T (**Section 2.0** of this guidance).

As fish are ectotherms, they are significantly influenced by temperature and can be greatly affected by temperature changes. In general, higher temperatures increase fish metabolism. Therefore, fish require more food and oxygen at higher temperatures, which can lead to use of energy stores when food cannot meet energy demands (Todd *et al.,* 2008). Fish grow quicker in warmer temperatures that are below sublethal thresholds.

Eme *et al.* (2015) and Mueller *et al.* (2015) examined temperature changes during different stages of egg and larval development and concluded that there are specific points in development where temperature change has a much larger effect. Other effects of temperature change are described in the sections below.

Since Delta T effects are unknown for a variety of species in the published literature, they are not included in **Tables 10** or **11**, but effects of temperature change are discussed in more detail in the following sections.

Table 10: Representative Values for Upper Thermal Tolerance of Canadian Freshwater Fish Species with Acclimation Temperatures Shown in Parentheses

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Northern Brook Lamprey (*Ichthyomyzon fossor*)	Coolwater	ammocoetes	30.5 (15)				Potter and Beamish (1975)	
American Brook Lamprey (*Lethenteron appendix*)	Coldwater	ammocoetes	29.5 (15)				Potter and Beamish (1975)	
Sea Lamprey (*Petromyzon marinus*)	Coolwater	eggs spawning	12 -20 (18)			15.5	Wismer and Christie (1987)	
		ammocoetes	29.5 (5) 30 (15) 31 (25)				Potter and Beamish (1975)	
		juvenile				20.5	Wismer and Christie (1987)	
Shortnose Sturgeon (*Acipenser brevirostrum*)	Cold/ Coolwater (Coker *et al.*, 2001)	YOY	34.8 (19.5) 36.1 (24.1)	33.7 (19.5) 35.1 (24.1)			Ziegeweid and Jennings (2008)	CTM LOE
Lake Sturgeon (*Acipenser fulvescens*)	Coolwater (Cold/ Coolwater in Coker *et al.*,	YOY	35.0 (18.1) 36.8 (24.9)	33.2 (18.1) 35.7 (24.9)			Wilkes (2011)*	CTM LOE
		juvenile	34.8 (18.1) 36.7 (24.9)	33.1 (18.1) 35.1 (24.9)				
Bowfin (*Amia calva*)	Warmwater	spawning				17.5	Wismer and Christie (1987)	
		-		37 (23.8)			Reutter and Herdendorf (1976)	CTM LOE
Alewife (*Alosa pseudoharengus*)	Coldwater	egg	24.5 (12-25) 28 (13)				Wismer and Christie (1987)	
		larva	31.4-37.1 (14-24)					
		YOY		28.3 (10-12) 32.7 (18-20) 34.4 (24-26)			Otto *et al.* (1976)	CTM LOE
			26.5 (10) 30.3 (20) 32.1 (25) 32.2-32.9 (23-25)					
			26.5 (10-12) 30.3 (18-20) 32.1 (24-26)				Wismer and Christie (1987)	
		juvenile	<23 (9) 24.5 (20)					
			23.5 (10) 23.5 (15) 24.5 (20)	29.5 (10) 30.1 (15) 31.2 (20)			Otto *et al.* (1976)	CTM LOE
		adult	28.2 (27) >20 (10) <22.8 (20) 22.8 (15) 31.4 (summer)				Wismer and Christie (1987)	
			23.2 (20)	30.2 (18.2)				CTM LOE

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Gizzard Shad (*Dorosoma cepedianum*)	Coolwater	egg				22	Wismer and Christie (1987)	
		YOY	31.0 (summer)					
			28.5 (summer)				Cvancara *et al.* (1977)	
		adult		31.7 (15.9)			Reutter and Herdendorf (1976)	CTM LOE
						23.2		
		-	34-34.5 (25) 36.0 (30) 36.5 (35)				Wismer and Christie (1987)	
Central Stoneroller (*Campostoma anomalum*)	Coolwater (Cool/ Warmwater in Coker *et al.*, 2001)	hatching				22.5	Wismer and Christie (1987)	
		juvenile				27		
		-		28.8 (7.5) 35.8 (23) 37.7 (24) 37.2 (26)			Beitinger *et al.* (2000)	CTM LOE
				31.8 (10)				CTM OS
Goldfish (*Carassius auratus*)	Warmwater	spawning				24	Wismer and Christie (1987)	
		juvenile				30.4		
		-		32 (5)			Talmage and Coutant (1979)	CTM LOE
				37.6 (25)				
				35.8 (10)			Beitinger *et al.* (2000)	CTM OS
			29-38.6 (5-30)	36.6 (25)			Wismer and Christie (1987)	CTM DP
			39.9-41 (5-40)	>35 (23.9)				CTM LOE
Spotfin Shiner (*Cyprinella spiloptera*)	Warmwater	adult		31.8 (11)			Beitinger *et al.* (2000)	CTM LOE
Common Carp (*Cyprinus carpio*)	Warmwater	eggs	35 (25)			21	Wismer and Christie (1987)	
		fry				34		
		larva	36.4 (16-21) 38.8 (19-27)				Talmage (1978)	
		-		39.0 (23.3)			Wismer and Christie (1987)	CTM LOE
			40.9				Horoszewicz (1973)	
Common Shiner (*Luxilus cornutus*)	Coolwater	-		30.6 (15) 31.9 (15)			Kowalski *et al.* (1978)	CTM OS
				30.6 (winter) 31.9 (spring)			Kowalski *et al.* (1978)	CTM OS
				35.7 (26)			Beitinger *et al.* (2000)	CTM LOE
River Chub (*Nocomis micropogon*)	Coolwater	nest building				23.5	Wismer and Christie (1987)	
		-		30.9 (15)			Kowalski *et al.* (1978)	CTM OS

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Golden Shiner (*Notemigonus crysoleucas*)	Coolwater	spawning				20	Wismer and Christie (1987)	
		juvenile	31.6 - 33.4 (summer) 30.4 (fall)					
		adult	29.3 (10) 30.5 (15) 31.8 (20) 33.2 (25) 34.7 (30) 30.3 (spring) 31.8-33.5 (summer)	33 (10) 35 (15) 36 (20) 38 (25) 39 (30)				
Emerald Shiner (*Notropis atherinoides*)	Coolwater (Warmwater in EPRI, 2011)	hatch				23	Wismer and Christie (1987)	
		YOY	35.2				Talmage (1978)	
			23.3 (5)	28.6 (7.8)				CTM LOE
			26.7 (10) 28.9 (15) 30.7 (20) 30.7 (25)				Wismer and Christie (1987)	
		juvenile				30		
					36.7 (20)		EPRI (2011)	
		-		37.6 (25) 34.1 (10)			Beitinger *et al.* (2000)	CTM OS
Spottail Shiner (*Notropis hudsonius*)	Coolwater	spawning				17.5	Wismer and Christie (1987)	
		YOY	34.7 (26)				Kellogg and Gift (1983)	
			30.5 (9) 32.4 (17) 34.3 (23) 35.8 (26)			30	Wismer and Christie (1987)	
		juvenile	36-37.3 (23) 36.8-38.1 (26)					
		adult		32.8 (21.7)			Reutter and Herdendorf (1976)	CTM LOE
Rosyface Shiner (*Notropis rubellus*)	Warmwater	spawning				24.5	Wismer and Christie (1987)	
		-		31.8 (15)			Kowalski *et al.* (1978)	CTM OS
				35.3 (26)			Beitinger *et al.* (2000)	CTM LOE
Sand Shiner (*Notropis stramineus*)	Warmwater			32.3 (winter) 33 (spring)			Kowalski *et al.* (1978)	CTM OS
				32.3-33.0 (15)			Beitinger *et al.* (2000)	CTM OS - all seasons
				37.0 (26)				CTM LOE
Bluntnose Minnow (*Pimephales notatus*)	Warmwater	spawning				25	Wismer and Christie (1987)	
		juvenile				27.9		
		adult		27.8 (6.0) 31.9 (15)			Kowalski *et al.* (1978)	CTM OS
		-		31.3 (11) 37.9 (24) 36.6 (26)			Beitinger *et al.* (2000)	CTM LOE
				33.7 (10)				CTM OS

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Fathead Minnow (*Pimephales promelas*)	Warmwater	spawning				23	Wismer and Christie (1987)	
		larvae		34.4 (22.5)			Beitinger *et al.* (2000)	CTM LOE
		juvenile				28	Wismer and Christie (1987)	
		adult		36.9/36.8/32.4/34.0 (14.5)			Beitinger *et al.* (2000)	female nonspawn/postspawn/male diel min 2200/max 1400 and 0200 CTM LOE
				33.2 (15)				CTM OS
				35.1 (21)				CTM LOE
				34.8/34.9/34.8 (20)				day 1/5/10 CTM LOE
				33.1 (20)				CTM LOE
		-	28 (10) 31.7 (20) 32.3 (25) 33 (30)				Wismer and Christie (1987)	
				28.6 (5) 30.7 (12) 36.4 (22) 40.4 (32)			Beitinger *et al.* (2000)	CTM LOE
Blacknose Dace (*Rhinichthys atratulus* for eastern and *Rhinichthys obtusus* for Western)	Coolwater	spawning				21	Wismer and Christie (1987)	
		adult	28.8-30 (20)					
		-	25 (5) 27 (10) 29.3 (20)					
				31.9 (15)			Kowalski *et al.* (1978)	CTM OS
				32.4 (20)			Beitinger *et al.* (2000)	CTM LOE
Longnose Dace (*Rhinichthys cataractae*)	Coolwater	spawning				15	Wismer and Christie (1987)	
		-		31.4 (15)			Kowalski *et al.* (1978)	CTM OS
Creek Chub (*Semotilus atromaculatus*)	Coolwater	spawning				19.8	Wismer and Christie (1987)	
		-	24.7 (5) 27 (10) 30.1-30.5 (17.1-17.5) 30 (20) 31.8 (21) 32.1 (22.8) 32.6 (25)					
				35.7 (26)			Beitinger *et al.* (2000)	CTM LOE
Tench (*Tinca tinca*)	Coolwater (Coker *et al.*, 2001)	-	39.3				Horoszewicz (1973)	
White Sucker (*Catostomus commersoni*)	Coolwater (Warmwater in EPRI, 2011)	spawning				10	Wismer and Christie (1987)	
		larvae	28.1-30.5 (8.9-21.2)	32.7		28		CTM DP
		adult		31.6 (19)			Beitinger *et al.* (2000)	CTM LOE
		-		34.9 (26)				CTM LOE
Northern Hog Sucker (*Hypentelium nigricans*)	Warmwater	hatching				17.5	Wismer and Christie (1987)	
		juvenile				28.1		
		-		30.8 (15)			Kowalski *et al.* (1978)	CTM OS

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Muskellunge (*Esox masquinongy*)	Warmwater (Coolwater in EPRI, 2011)	1-15 d post-hatch		30.5-34.0 (15)			Paladino and Spotila (1978)	CTM LOE
		1-31 d post-hatch		27.2-34.0 (7)			Beitinger *et al.* (2000)	CTM body bends
		1-31 d post-hatch		29.8-34.4 (15)				CTM bends and LOE
		1-19 d post-hatch		33.2-36.1 (25)				CTM LOE
		fry		29.9-36.0 (16-22)			Bonin and Spotila (1978)	CTM OS and PGR
		juvenile	32.25 (25) 23.75 (27.5) 33.25 (30) 32.5 (25)			28.4	Wismer and Christie (1987)	
Rainbow Smelt (*Osmerus mordax*)	Coldwater	hatching				23	Wismer and Christie (1987)	
		adult	21.5 - 28.5 (10.2-15) 18.9, 19 (17)					
		-		24.9 (6)			Reutter and Herdendorf (1976)	CTM LOE
Cisco (Lake Herring) (*Coregonus artedii*)	Coldwater	larvae	19.8				McCormick *et al.* (1971)	
						3	Wismer and Christie (1987)	
		juvenile	19.7 (2) 21.7 (5) 24.2 (10) 26.2 (20) 25.7 (25)				Edsall and Colby (1970)	
			26 (25)				EPRI (2011)	
		-				17	Wismer and Christie (1987)	
Lake Whitefish (*Coregonus clupeaformis*)	Coldwater	egg	10 (0.5)			7	Wismer and Christie (1987)	
		YOY	20.62 (5) 22.67 (10) 25.78 (15) 26.65* (20) 26.65* (22.5)				Edsall and Rottiers (1976)	
Round Whitefish (*Prosopium cylindraceum*)	Coldwater	egg				3	Wismer and Christie (1987)	
Mountain Whitefish (*Prosopium williamsoni*)	Coldwater (Coker *et al.*, 2001)	fry	23.6/22.6 (10)	36.7 (13.4)			Brinkman and Crockett (2013)	UUILT 7/33 days CTM LOE
Bloater (*Coregonus hoyi*)	Coldwater	yearling	22.2 (5) 23.6 (10) 24.8 (15)				Edsall *et al.* (1970)	
Pink Salmon (*Oncorhynchus gorbuscha*)	Coldwater	spawning				10	Wismer and Christie (1987)	
		juvenile				21.7		

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Coho Salmon (*Oncorhynchus kisutch*)	Coldwater	fry	22.9 (5) 23.7 (10) 24.3 (15) 25 (20 and 23)			10	Wismer and Christie (1987)	
		fingerling	21 (12)	26.5 (2.6) 25/26/27/28/29 (5) 27/28/29/30/31 (15)			Becker and Genoway (1979)	1/6/18/30/60°Chr^{-1}, CTM LOE 1/6/18/30/60°Chr^{-1}, CTM LOE
		juvenile		28.7-29.7 (15) 29.1-29.2 (15-17)			EPRI (2011)	CTM LOE
		adult	25 (17)				Wismer and Christie (1987)	
		-		28.2 (11.5) 29.2 (14.6) 29.1 (16.6) 27.6 (11.1) 27.9 (11.1)			Beitinger *et al.* (2000)	CTM LOE
						18	Wismer and Christie (1987)	
Sockeye Salmon (*Oncorhynchus nerka*)	Coldwater	egg				8.5	Wismer and Christie (1987)	
		juvenile				18.3		
Chinook Salmon (*Oncorhynchus tshawytscha*)	Coldwater	egg	14.9				Wismer and Christie (1987)	
		fry	21.5 (5) 24.3 (10) 25 (15) 21.1 (20 and 24)					
		1-2 y	25.1 (17) 22 (17)				Coutant (1970)	
		adult	21-22 (18-19)				Templeton and Coutant (1970)	
		-				18.7	Wismer and Christie (1987)	
Cutthroat Trout (*Salmo clarki*)	Coldwater (Coker *et al.*, 2001)	-	23.5-24.3 (18) 24.2 (18)	29.4-30.0 (18)			EPRI (2011)	CTM LOE and OS
				27.6 (10)			Beitinger *et al.* (2000)	CTM LOE

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Rainbow Trout (*Oncorhynchus mykiss*)	Coldwater	Egg	>29 (15)			9	Wismer and Christie (1987)	
		fingerling	23.7 (5) 24.2 (9) 25.2 (13) 25.7 (17) 26.2 (21) 26.2 (24.5)				Kaya (1978)	
		6 weeks		28.0±0.36 (10) 29.1±0.27 (15) 29.8±0.36 (20)			Currie *et al.* (1998)	CTM LOE
		YOY		27.6±0.47 (9.8) 28.1±0.29 (9.8)			Carline and Machung (2001)	Domestic Strain, CTM LOE Wild Strain, CTM LOE
		juvenile	23.2/25.0 (5) 24.7/25.2 (9) 24.7/25.2 (13) 25.2/25.7 (17) 25.7/26.2 (21) 26.2/26.2 (24.5)				Kaya (1978)	Firehole/Winthrop hatchery
				28.4-28.8 (15)			Galbreath *et al.* (2004)	CTM LOE
				27.3-29.3 (10-20 cycle)			Currie *et al.* (2004)	CTM LOE
			26.2				Brinkman and Crockett (2013)	UUILT 7 days
		adult				19 17	Wismer and Christie (1987)	
			21-22 (18-19)				Templeton and Coutant (1970)	
				29.3 (20)			EPRI (2011)	
				29.0-29.7 (14)			Rodnick *et al.* (2004)	CTM LOE
			25.6 (16)				Hokanson *et al.* (1977)	
		-		26.9 (8) 28.4 (10) 29.4 (20) 29.4 (15)	26.3 (17)		Bietinger *et al.* (2000)	CTM LOE
Atlantic Salmon (*Salmo salar*)	Coldwater	YOY		32.9 (15) 32.8 (20)	28.7/29.2 (17)		Bietinger *et al.* (2000)	CTM LOE
		juvenile		32.6 (15) 32.7 (20)				

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Artic Grayling (*Thymallus arcticus*)	Coldwater (Coker *et al.*, 2001 and EPRI, 2011)	-		24.8 -24.9 (5) 27.5-27.9 (15)			Bietinger *et al.* (2000)	CTM DP
				26.4 (8.4) 28.5 (16) 29.3 (20)				CTM LOE
			25.0 (20)				EPRI (2011)	
Burbot (Ling) (*Lota lota*)	Coldwater (Cold/ Coolwater in Coker *et al.*, 2001)	spawning				1.2	Wismer and Christie (1987)	
		juvenile		27.1 (5.2) 27.8 (9.9) 30.2 (14.9) 31.5 (19.6) 26.8 (5.9) 27.5 (9.8) 29.6 (14.9) 31.7 (19.6			Hofmann and Fischer (2002)	CTM LOE
Banded Killifish (*Fundulus diaphanus*)	Coolwater	spawning				23	Wismer and Christie (1987)	
		adult	34.5 (25)				Rombough and Garside (1977)	
Threespine Stickleback (*Gasterosteus aculeatus*)	Coolwater (Coldwater Coker *et al.*, 2001)	spawning				19	Wismer and Christie (1987)	
		larvae		33.5 (18.6)			Beitinger *et al.* (2000)	CTM DP
		juvenile		33.6 (18.6)				CTM DP
		-		30.5 (8)				CTM DP
				32.5 (13)				CTM DP
				33.5 (15.6)				CTM DP
				34.6 (22.7)				CTM DP
			25.7 (10) 27.2 (20)				Jordan and Garside (1972)	
Mottled Sculpin (*Cottus bairdi*)	Coolwater (Coldwater in Coker *et al.*, 2001 and EPRI, 2011)	-		30.9 (15)			Kowalski *et al.* (1978)	CTM OS
Slimy Sculpin (*Cottus cognatus*)	Coldwater	-	25 (20)				Symons *et al.* (1976)	
			26.5 (20)	29.4 (20)			EPRI (2011)	CTM LOE
				22.7, 24.0 (5) 24.8, 25.1 (10) 26.3, 27.3 (15)			Beitinger *et al.* (2000)	CTM LOE
White Bass (*Morone chrysops*)	Warmwater	spawning				24	Wismer and Christie (1987)	
		eggs				26		
		larvae	31.7-30.6 (14-26)					
		YOY	33.5 (summer)	35.3 (21.7)			Cvancara *et al.* (1977)	CTM LOE
		Juvenile				26.7	Wismer and Christie (1987)	

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Pumpkinseed (*Lepomis gibbosus*)	Warmwater	YOY				29.3	Wismer and Christie (1987)	
		juvenile	33/30/32 /33/35 (10) 36/36/37 /37/38 (20)	31/29/30 /30/31 (10) 35/35/35 /35/36 (20)			Becker and Genoway (1979)	1/6/18/30/60°Chr^{-1}, CTM LOE
		adult	34.5 (25)	37.5 (21.1)			Wismer and Christie (1987)	CTM LOE
		-		30.1 (10) 35.1 (20)			Beitinger *et al.* (2000)	CTM LOE
Bluegill (*Lepomis macrochirus*)	Warmwater	spawning				25	Wismer and Christie (1987)	
		egg	33.8				Banner and Van Arman (1973)	
		YOY	28.5 (summer)				Cvancara *et al.* (1977)	
		juvenile	27.5 (12.1) 37.3 (32.9)				Banner and Van Arman (1973)	
		juvenile		33.9±0.3 (10) 37.2±0.2 (20) 41.2±0.3 (30)			Dent and Lutterschmidt (2003)	CTM LOE
		adult	31-34 (15-30)	38.3 (22.8)		29	Wismer and Christie (1987)	CTM LOE
		-			35.5 (21.5)		Beitinger *et al.* (2000)	
Smallmouth Bass (*Micropterus dolomieu*)	Coolwater (Warmwater in Coker *et al.*, 2001 and EPRI, 2011)	spawning				17	Wismer and Christie (1987)	
		Larvae	38				EPRI (2011)	
		YOY/fry/juvenile	35 (33)	36.9 (26)			EPRI (2011)	CTM LOE
		juvenile				29 32/33	Wismer and Christie (1987)	
		adult	35-38				McKinley *et al.* (2000)	
		adult		34.8 (10)			Beitinger *et al.* (2000)	CTM OS

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Largemouth Bass (*Micropterus salmoides*)	Warmwater	spawning				29-32	Wismer and Christie (1987)	
		eggs	32.5	36.7 (20) 40.1 (28)				
		YOY	35.6 (summer)				Cvancara *et al.* (1977)	
		YOY			39.8 (30) 39.3 (36) 38.9 (30)		Beitinger *et al.* (2000)	
		Juvenile		35.4±0.47 (20) 36.7±0.59 (25) 38.5±0.34 (30)			Currie *et al.* (1998)	CTM LOE
		Juvenile		35.4±0.47 (20) 36.7±0.59 (25) 38.5±0.34 (30) 35.6-37.3 (20-30 cycle)			Currie *et al.* (2004)	CTM LOE
		Juvenile		36.3 (25)			EPRI (2011)	CTM LOE
		subadult				32	Wismer and Christie (1987)	
		adult			39.1 (30) 37.3 (32)		Beitinger *et al.* (2000)	
		-		29.2 (8) 33.6 (16) 36.5 (24) 40.9 (32)				CTM DP
Black Crappie (*Pomoxis nigromaculatus*)	Coolwater (Warmwater in EPRI, 2011)	adult		34.9 (23.8)			Reutter and Herdendorf (1976)	CTM LOE
Greenside Darter (*Etheostoma blennioides*)	Warmwater (Cool/ Warmwater in Coker *et al*., 2001)	-		32.2 (15)			Kowalski *et al.* (1978)	CTM OS
				28.8-33.5 (10) 32.2-34.5 (20)			Beitinger *et al.* (2000)	CTM LOE
Rainbow Darter (*Ethcostoma caerulum*)	Coolwater (Warmwater in EPRI, 2011)	-		32.1 (15)			Kowalski *et al.* (1978)	CTM OS
				29.8-32.0 (10) 32.8-34.0 (20) 35.6 (26)			Beitinger *et al.* (2000)	CTM LOE
Fantail Darter (*Etheostoma flabellare*)	Coolwater	-		32.1 (15)			Kowalski *et al.* (1978)	CTM OS
				31.1 (15) 31.3 (15)			Beitinger *et al.* (2000)	CTM OS - winter CTM OS - summer
				31.1-34.0 (10) 32.9-35.0 (20) 33 (20) 37.7 (24) 36.0 (26)				CTM LOE

Table 10 continued

Species	Thermal Guild	Stage	Upper Incipient Lethal Temperature UILT (°C)	Critical Thermal Maximum CTMax (°C)	Chronic Lethal Maximum CLMax (°C)	Maximum Weekly Average Temperature MWAT (°C)	Reference	Comment
Johnny Darter (*Etheostoma nigrum*)	Coolwater (Warmwater in EPRI, 2011)	-		30.7 (winter) 31.4 (spring)			Kowalski *et al.* (1978)	CTM OS
				30.9 (15) 30.5 (15) 33 (20) 34.0-37.4 (20-30) 36.4 (26)			Beitinger *et al.* (2000)	CTM OS - winter CTM OS - summer CTM LOE
Yellow Perch (*Perca flavescens*)	Coolwater (Warmwater in EPRI, 2011)	spawning				11.9	Wismer and Christie (1987)	
		egg	19.9 (spring)				Hokanson (1977)	
		larvae	>24 (7.6) >26.6 (15.8)				Wismer and Christie (1987)	
		juvenile	29.7 (25) 32-34 (21)	33.4 (25)				
		adult	33.4 (25)				EPRI (2011)	
			21-32.3 (5-25)	35 (22)		22	Wismer and Christie (1987)	CTM LOE
Sauger (*Sander canadensis*)	Coolwater (Warmwater in EPRI, 2011)	egg	20.9 (spring)				Hokanson (1977)	
		juvenile	30.4 (26)		30		EPRI (2011)	CLM two months
Walleye (*Sander vitreum*)	Coolwater (Warmwater in EPRI, 2011)	spawning				8.9	Wismer and Christie (1987)	
		eggs	19.2 (spring)				Hokanson (1977)	
					21		Smith and Koenst (1975)	all eggs die
		larvae	19.2				Hokanson (1977)	
		juvenile		34.8-35 (23) 35.8-35.9 (23)			EPRI (2011)	CTM LOE CTM OS
			31.6 (25.8)				Smith and Koenst (1975)	
		adult	28.9 (7.2)			25	Wismer and Christie (1987)	
		-	34 (26) 33 (22) 34.1 (26) 34.1 (28)				EPRI (2011)	
Freshwater drum (*Aplodinotus grunniens*)	Warmwater	YOY	32.8 (summer)				Cvancara *et al.* (1977)	
		adult		34 (21.2)			Reutter and Herdendorf (1976)	CTM LOE
Round Goby (*Neogobius melanostomus*)	Coolwater	adult		33.4±0.30/ 31.5±0.46 (15)			Cross and Rawding (2009)	CTM OV/LOE

* Source: Thesis
Thermal Guild was taken from Eakins, 2016 for Ontario species. Contrary results or Canadian species that are not in Ontario are referenced.
CTM LOE - Critical Thermal Method endpoint Loss of Equilibrium
CTM OS- Critical Thermal Method endpoint Onset of Muscle Spasms
CTM DP- Critical Thermal Method endpoint Death Point
CTM PGR- Critical Thermal Method endpoint Prodding with Glass Rod
CTM OV- Critical Thermal Method endpoint sessation of Opercular Ventilation
A dash for stage indicates the life stage is unknown

5.1.6.4 Advanced Hatch and Early Development - Delta T Effects

Early life stages of fish are sensitive. Environmental fluctuations in temperature, or Delta T, during these stages can alter embryonic growth, time to hatch, metabolism, gene expression, thermal acclimation, and growth, all of which can effect survival (Eme *et al.*, 2015; Mueller, 2015; Patrick *et al.*, 2015)

Embryos of many fish species have shown changes in development due to temperature increases. The early hatch response is one of these and demonstrates that embryonic development has been altered. Patrick *et al.* (2013) examined the effects of increasing temperature on hatching Lake and Round Whitefish (*Coregonus clupeaformis* and *Prosopium cylindraceum*, respectively) and found that advanced hatch is more pronounced with greater increases in temperature. A fixed 5°C increase over ambient incubation temperature could produce an early hatch response of 41 days in Round Whitefish. Under a variable temperature incubation of up to 5°C above ambient the early hatch response was 33 and 35 days in Round and Lake Whitefish, respectively.

Mueller (2015) examined temperature increases in Lake Whitefish at different stages of embryonic development. An increase in constant incubation temperature resulted not only in advanced hatch, but fewer and smaller hatchlings. Increasing incubation temperature also decreased yolk conversion efficiency and increased the overall cost of development.

Temperature changes in early development have also been shown to alter metabolism through gene expression. Eme *et al.* (2015) inferred rates of metabolism by measuring oxygen consumption and heart rate of embryonic and hatchling Whitefish. It was shown that the fish metabolism was influenced by incubation temperatures and that embryos incubated at higher temperatures had higher metabolic rates.

Advanced hatch may be a disadvantage to recently hatched larvae. Escape response of Walleye Pollock (*Theragra chalcogramma*) larvae was shown by Porter and Bailey (2007) to be affected by time to hatch. In the laboratory the earlier hatching larvae were not only smaller but less sensitive to tactile stimulation, had a slower escape response and higher rates of mortality due to predation.

Schneider *et al.,* (2002) changed water temperatures of incubating Walleye (*Sander vitreus*) eggs on either day 3 or 7. This had no effect on eye-up rates, but embryos in warmer temperature treatments hatched earlier. Swim-up rates were significantly lower for treatments exposed to temperatures 20°C above ambient. These extreme temperature changes also produced smaller fry with severe deformities.

Advanced hatch may also lead to larval mortality due to an unavailable food supply. Several studies have reported a strong relationship between food availability, growth rates and survival during the first few months of life for Lake Whitefish (Freeberg *et al.,* 1990; Fernandez, 2009; Hoyle *et al.*, 2011).

5.1.6.5 Cold Shock - Delta T Effect

A decrease in ambient water temperature can lead to a rapid decrease in fish body temperature that can cascade to physiological and behavioral responses. Water temperatures can decrease for natural and anthropogenic reasons.

Fish may experience natural decreases in temperature in a number of situations including: travelling between zones in a water column that is thermally stratified; when there is a rapid change in solar heating of the water surface; when there are abnormal water movements; during rapid precipitation events; and, during rapid seasonal temperature changes.

Cold fronts, convection of warm moist air above a mass of cool low-pressure air, can rapidly decrease water temperatures (VanDeHay *et al.,* 2013). Human induced decreases in temperature occur when thermal effluents decrease in discharge or flow, around dams and other water control projects, and during fish handling practices (Donaldson *et al*., 2008).

Early research on cold shock examined mortality, but recently sublethal effects have been investigated. In a review of cold water temperature effects on fish, Donaldson *et al.* (2008) grouped responses into primary (neuroendocrine, catecholamine, and corticosteroid), secondary (metabolic, hematological, osmoregulatory, and immunological), and tertiary (developmental changes, change in disease resistance, habitat use, changes in foraging, and changes in migration).

A temperature decrease also seems to result in death more frequently than an increase in temperature of the same magnitude. Beitinger *et al.* (2000) summarized the reasons for this as: 1) fish are able to increase their tolerance to higher temperatures quicker than to colder temperatures; 2) fish lose their tolerance to high temperatures slowly; 3) high temperatures increase activity allowing potential escape whereas cold temperatures induce lethargy; and 4) upper temperature tolerances are well above ambient temperatures in natural habitats. This suggests fish are better adapted to increases in temperature rather than decreases.

Survival in winter of juveniles of various fish species has been observed to be better in stable, warmer years. Years with severe cold fronts have been associated with low recruitment. In addition to the lack of food sources and increased predation from compromised functions, it is thought that the juvenile life stage of fish is more sensitive to cold water temperatures (Jansen *et al.,* 2009; Fetzer *et al.,* 2011; VanDeHey *et al.,* 2013). Below are examples of experimental observations of cold shock of Yellow Perch (*Perca flavescens*) and Gizzard Shad (*Dorosoma cepedianum*), as well as an example of cold shock seen in various fish species during a power plant shutdown.

Yellow Perch

Yellow Perch eggs exposed to a simulated cold front had no change in hatching success when temperatures dropped 6°C in 24 hours and 8°C in 45 minutes, although it took the exposed eggs 4 days longer to hatch than control (Jansen *et al.,* 2009). Fry exposed to simulated cold fronts of a water temperature decrease of 4°C did not differ from the control in swimming activity. Fry in tanks that decreased 8°C ceased swimming and settled to the tank bottom once temperatures reached 3.9°C and resumed normal swimming activity when temperatures increased 6°C after the 48 hour simulated cold front (VanDeHey *et al.,* 2013). Although the delay in hatch and reduced activity are not direct mechanisms to cause mortality, food availability and sublethal stress as well as increased predation may affect survival.

Gizzard Shad

At the northern range of Gizzard Shad overwinter mortality is common. Fetzer *et al.* (2011) examined this further with field, cage, and lab experiments. Mortality was low until temperatures dropped below 8°C in the field, at which time 75% mortality was noted. In the lab mortality increased below 4°C, but was higher at 2 and 1°C. The proposed reason for the increased mortality based on the experimental results was that the Gizzard Shad cannot access lipid energy reserves in cold temperatures, leading to the use of emergency stores, such as the liver.

Fish were captured in a discharge channel of the Ginna Nuclear Power Plant and exposed to cold water (0.4-0.6°C) for cold shock experiments (Smythe and Sawyko, 2000). Most of the species tested (mostly salmon and perch) had a high survival rate. The Gizzard Shad had the lowest survival rate in the experiments. Based on their results and others, they surmised that loss of equilibrium would happen within 2 to 5 minutes followed by death within four days of exposure to cold shock water temperatures.

Unplanned Plant Shut Down

During an unplanned reactor shutdown at Gentilly Nuclear Generating Station, and subsequent pause in cooling water discharge, at least 6 species of fish were found dead within a few hours within the discharge channel (Lair, 2007). There were also lethargic and immobile large fish lined along the warmest channel's bank. Fish were examined and it was concluded that the rapid drop in temperature from the sudden termination of warm water discharge caused cold shock in these fish, resulting in reduced function and in some cases death. At Bay Shore Power Plant the number of Gizzard Shad impinged increased in October to December due to cold shock (Patrick *et al.,* 2015)

Cold shock has been found to influence genetics of fish and as a result has been useful for fisheries stocking (Donaldson *et al.,* 2008). Cold shock may either result in the egg retaining the second meiotic polar body or blocking the first mitotic division, resulting in cells with more chromosomes than normal, or polyploidy. Triploidy produces sterility and larger and fewer cells in tissue and organs with a larger growth rate. Sterility of hatchery fish allows stocking to occur without reproductive contamination with wild stock as well as sterility of products in aquaculture facilities to reduce impact of escapees.

Table 11 presents the lower acute and chronic temperature tolerances for a variety of species in Canada. The lower incipient lethal temperature, critical thermal minimum, and chronic lethal minimum are measured in a manner similar to their upper and maximum counter parts, as defined above.

Table 21: Representative Values for Lower Thermal Tolerance of Canadian Freshwater Fish Species with Acclimation Temperatures Shown in Parentheses

Species	Thermal Guild	Stage	Lower Incipient Lethal Temperature LILT (°C)	Critical Thermal Minimum CTMin (°C)	Chronic Lethal Minimum CLMin (°C)	Reference	Comment
Alewife (*Alosa pseudoharengus*)	Coldwater	juvenile	3			Wismer and Christie (1987)	
		adult	2.5 (5 and10) <6 (15) 6-8 (21)			Otto *et al.* (1976)	
		adult	3 (5.21) <6 (21) 8 (20) 4 (winter)			Wismer and Christie (1987)	
Gizzard Shad (*Dorosoma cepedianum*)	Coolwater	juvenile	9 3			Wismer and Christie (1987)	
		-	0 (10) 3.5 (15) 7.5 (20) 11 (25) 14.6 (30) 6-7 (15-20)				
Goldfish (*Carassius auratus*)	Warmwater	juvenile	1.0 (19) 5.0 (24) 15.5 (38)			Wismer and Christie (1987)	
Golden Shiner (*Notemigonus crysoleucas*)	Coolwater	juvenile	3.4 (21 and 22.8)			Wismer and Christie (1987)	
		adult	1.5 (15) 4.0 (20) 4.0 (25) 11.2 (30)				
Emerald Shiner (*Notropis atherinoides*)	Coolwater (Warmwater in EPRI,	YOY	1.6 (15) 5.2 (20) 8.0 (25)			Wismer and Christie (1987)	
Spottail Shiner (*Notropis hudsonius*)	Coolwater	spawning	4.9 (21.8)			Wismer and Christie (1987)	
Bluntnose Minnow (*Pimephales notatus*)	Warmwater			6.4 (22)		Beitinger *et al.* (2000)	CTM LOE
Fathead Minnow (Pimephales promelas)	Warmwater	larvae		5.9 (22.5)		Beitinger *et al.* (2000)	CTM PGR
		-	2 (20) 10.5 (30)			Wismer and Christie (1987)	
Blacknose Dace (*Rhinichthys atratulus* for eastern and *Rhinichthys obtusus* for Western)	Coolwater			6.7 (28)		Beitinger *et al.* (2000)	CTM LOE
Creek Chub (*Semotilus atromaculatus*)	Coolwater		1.0 (20) 1.7 (21-21.9)			Wismer and Christie (1987)	

Table 11 continued

Species	Thermal Guild	Stage	Lower Incipient Lethal Temperature LILT (°C)	Critical Thermal Minimum CTMin (°C)	Chronic Lethal Minimum CLMin (°C)	Reference	Comment
White Sucker (*Catostomus commersoni*)	Coolwater (Warmwater in EPRI, 2011)	larvae	4.8-6.1 (15.2-21.2)			Wismer and Christie (1987)	
Channel Catfish (*Ictalurus punctatus*)	Warmwater	juvenile		2.7±0.41 (20) 6.5±0.40 (25) 9.8±0.41 (30) 6.1-6.6 (30-30 cycle)		Currie *et al.*, 2004	CTM LOE
Northern Pike (*Esox lucius*)	Coolwater (Warmater in EPRI, 2011)	spawning embryo larvae	4.9 (21.8) 5 (17.7) 3 (18)			Wismer and Christie (1987)	
Rainbow Smelt (*Osmerus mordax*)	Coldwater	adult	8.5 (17)			Wismer and Christie (1987)	
Cisco (Lake Herring) (*Coregonus artedii*)	Coldwater	juvenile	<0.3 (2) 0.5 (5) 3.0 (10) 4.7 (20) 9.7 (25)			Wismer and Christie (1987)	
Coho Salmon (*Oncorhynchus kisutch*)	Coldwater	fry	0.2 (5) 1.7 (10) 3.5 (15) 4.5 (20) 6.4 (23)			Wismer and Christie (1987)	
Chinook Salmon (*Oncorhynchus tshawytscha*)	Coldwater	egg fry	5.1 0.8 (10) 2.5 (15) 4.5 (20) 7.4 (24)			Wismer and Christie (1987)	
Rainbow Trout (*Oncorhynchus mykiss*)	Coldwater	egg hatch	3			Wismer and Christie (1987)	
		juvenile		<0 (10) 0.2±0.16 (15) 2±0.22 (20) 0.13-1.45 (10-20 cycle)		Currie *et al.* (2004)	CTM LOE
		-	<0.0 (10) 0.7 (15) 2.1 (20)			Beitinger *et al.* (2000)	
Pumpkinseed (*Lepomis gibbosus*)	Warmwater	-		1.7 (15) 4.1 (20) 8.7 (25) 12.1 (30)		Beitinger *et al.* (2000)	CTM LOE

Table 11 continued

Species	Thermal Guild	Stage	Lower Incipient Lethal Temperature LILT (°C)	Critical Thermal Minimum CTMin (°C)	Chronic Lethal Minimum CLMin (°C)	Reference	Comment
Bluegill (*Lepomis macrochirus*)	Warmwater	egg YOY juvenile adult	21.9 (26) 11 (26) 6 (26) 3-11 (15-30)			Wismer and Christie (1987)	
Largemouth Bass (*Micropterus salmoides*)	Warmwater	juvenile		3.2±0.27 (20) 7.3±0.52 (25) 10.7±0.61 (30) 5.9-7.7 (20-30 cycle)		Currie *et al.* (2004)	CTM LOE
		YOY juvenile			1.3 (10) <1.0 (10) 3.0 (10) <1.0 (10)	Beitinger *et al*. (2000)	summer winter
		adult	5.5 (20) 11.8 (30)			Wismer and Christie (1987)	
Yellow Perch (*Perca flavescens*)	Coolwater (Warmwater in EPRI, 2011)	egg juvenile	6.8 (spring) 4 (25)			Wismer and Christie (1987)	
Sauger (*Sander canadensis*)	Coolwater (Warmwater in EPRI, 2011)	egg	6.0 (spring)			Hokanson (1977)	
Walleye (*Sander vitreum*)	Coolwater (Warmwater in EPRI, 2011)	egg	<6.0 (spring)			Hokanson (1977)	

Thermal Guild was taken from Eakins, 2016 for Ontario species. Contrary results or Canadian species that are not in Ontario are referenced.

CTM LOE - Critical Thermal Method endpoint Loss of Equilibrium
CTM OS- Critical Thermal Method endpoint Onset of Muscle Spasms
CTM DP- Critical Thermal Method endpoint Death Point
CTM PGR- Critical Thermal Method endpoint Prodding with Glass Rod
CTM OV- Critical Thermal Method endpoint sessation of Opercular Ventilation
A dash for stage indicates the life stage is unknown

5.1.6.6 Factors to Consider for Developing Thermal Criteria

A number of factors influence the thermal tolerance of a fish, such as acclimation temperature, rate of temperature change, time of day, season, photoperiod, age, gender and maturity of gonads, hormonal and immunological status, metabolism, starvation, infections, and some toxins (EPRI, 2011; Golovanov, 2012). These are discussed further below.

Acclimation

Acclimation temperature is one of the two main factors that influence results of temperature tolerance experiments. It has been shown that higher test endpoint results are obtained when fish are acclimated at higher temperatures. Endpoint estimates have been found to vary as much at 10°C for warmwater species (EPRI, 2011). EPRI (2011) has summarized CTM results for fish species with multiple acclimation temperatures. These data are presented below. The seasonally appropriate acclimation temperature, or warmer, should be considered before choosing benchmark values. Summer acclimation temperatures are suggested to follow the thermal guilds categorization as defined by EPRI (2011).

Species	Acclimation Temperature (°C)	CTMax (°C)	Reference
Bluegill	16	31.5	Murphy *et al.* (1976)
	24	37.5	
	32	41.4	
Largemouth Bass	8	29.2	Fields *et al.* (1987)
	16	33.6	
	24	36.5	
	32	40.9	
Fathead Minnow	5	28.6	Richards and Beitinger (1995)
	12	30.7	
	22	36.4	
	32	40.4	
Channel Catfish	12	34.5	Cheetham *et al.* (1976)
	16	34.2	
	20	35.5	
	24	37.5	
	28	39.2	
	32	41.0	
Slimy Sculpin	5	22.7	Otto and O'Hara Rice (1977)
	10	24.8	

Species	Acclimation Temperature (°C)	CTMax (°C)	Reference
Slimy Sculpin	15	26.3	Otto and O'Hara Rice (1977)
	20	29.4	
Rainbow Trout	10	28.1	Currie *et al.* (1998)
	15	29.1	
	20	29.8	

Source: EPRI, 2011

Season and Time of Day

Golovanov (2013) observed temperature preferences of fish in different seasons and times of the day. Multiple species had temperature preferences that were 4 to 7°C higher during daytime than at night. Fish also preferred colder temperatures in winter, the warmest temperatures in summer and moderate temperatures in fall and spring. When acclimation temperatures were the same in summer and winter toxicity results usually varied by ~1°C, but can be as high as 2.6°C (EPRI, 2011). Toxicity results for a variety of fish have been found to be ~2°C higher during the day than at night (Govonanov, 2012).

Size and Life Stage

Juvenile fish are generally more tolerant to higher temperatures than adults of the same species, although only by about a 1-2°C difference (EPRI, 2011). Juvenile fish prefer warmer temperatures to adult fish, except when spawning adult fish will move to higher temperatures. Upper tolerance temperatures, as CLMax, are low in larvae, high in YOYs, highest in juveniles 1-2 years old, and decreases as the fish ages to levels that are below the larvae level (Golovanov, 2013). Juveniles are more sensitive to cold shock than adults (Donaldson *et. al.,* 2008).

Specific temperatures are required at certain stages for development. For example, Yellow Perch eggs do not mature in the female if winter temperatures remain above 12°C. It has been found for 11 species that spawning temperatures are almost identical to optimal embryogeny temperatures. The YOY's optimal growth is ~10°C warmer than that required for embryogeny (Golovanov, 2013). Hatch survival is significantly reduced when water temperature drops below 4°C and colder temperatures delay hatch (Donaldson *et. al.,* 2008). Advanced hatch can also occur with increased temperature (Patrick *et al.,* 2013).

Recestar *et al.* (2012) examined fish size (YOY to adult/subadult sizes) and CTMax for six fish species and found no significant difference between size and test results for four of the species. Two of the species had significantly lower CTMax for larger fish, although only a 1°C difference between the smallest and largest fish.

Thermal Guild and Reproduction

When comparing acute thermal toxicity endpoints to thermal guilds, Hasnain *et al.* (2013) and EPRI (2011) both found that CTMax and UILTs values were lowest for coldwater, highest for warmwater, and moderate for coolwater species.

Hasnain *et al.* (2013) also compared optimal spawning temperatures to spawning season and reproductive behavior. Spawning seasons were either fall or spring and included a variety of reproductive behaviors. Fall spawning species had significantly lower optimal spawning temperatures with a narrower temperature range than spring spawning species. Most reproductive behaviors were similar in optimal temperature except the nonguarder brood hiders had significantly lower temperatures.

Condition of Test Fish

EPRI (2011) and others suppose that anything that reduces fish health likely reduces the thermal tolerance. Starvation and exposure to nitrite in separate studies have reduced CTM (Golovanov, 2012). Golovanov (2013) noted that many diseased fish demonstrate behavioral fever, selecting higher temperatures.

5.1.6.7 Climate Change

Climate change is rapidly becoming one of the most challenging issues for management of coldwater fisheries (Williams *et al*., 2015) and will result in potentially large scale changes in water availability (Vliet *et al*.,2016).

Many studies have documented increasing trends in water temperature. Increases have been seen in not only small creeks and streams but have been documented in major rivers and lakes. Kaushal *et al*. (2010) found that 50 percent of major streams and rivers, with long term data in the US, have shown statistically significant long-term warming. The US EPA has documented increasing surface water temperatures for Lakes Superior, Michigan, Huron, and Ontario since 1995 (US EPA, 2015).

These warmer temperatures could alter aquatic ecosystems via changes in trophic status, ecosystem processes such as biological productivity and stream metabolism, contaminant toxicity, and loss of aquatic biodiversity (Kaushal *et al*., 2010).

Farmer *et al*. (2015) evaluated Yellow Perch populations in Lake Erie and showed that failed annual recruitment events followed short, warm winters. Following warmer winters the females spawned at warmer temperatures and produced smaller eggs. This resulted in lower hatch rates and smaller larvae.

Mills *et al*. (2005) reported a variety of climate change effects seen over the past 60 years in Lake Ontario. These included substantial winter kills of Alewife during severe winters, stronger year classes of warm water species with increased spring temperatures and decreased year classes of coldwater fishes during warmer fall water temperatures.

Water availability is also being affected by climate change. In the Great Lakes Basin the US EPA (2015) has documented lower water levels due to the increasing water temperatures. Higher water temperatures increase the rates of evaporation and causes lake ice to form later than usual extending evaporation potential. In rivers and streams, climate change is changing the timing of peak flows, altering flow regimes, and creating more frequent and intense disturbances (Kaushal *et al*., 2010). This will affect fish and will challenge scientists that manage aquatic systems (Mills *et al*., 2005).

This changing water availability will also have a strong implication for future cooling water use. Vliet *et al*. (2016) documented that even a 20% increase in efficiencies of power generation will be insufficient to mitigate overall reductions in cooling water use potential under a changing climate with increasing ambient water temperatures.

5.1.6.8 Population Level Thresholds for Thermal Effects

Thresholds are limits beyond which cumulative change becomes a concern, such as extensive disturbance to a habitat resulting in the rapid collapse of a fish population. The Canadian Environmental Assessment Agency (CEAA, 2014) states that thresholds may be expressed in terms of goals or targets, standards and guidelines, carrying capacity, or limits of acceptable change, each term reflecting different combinations of scientific data and societal values. For example, a threshold can be a maximum concentration of a certain pollutant (i.e. thermal discharge limit) beyond which the viability of the population is threatened.

For most pollutants, there is no readily available guidance on what percent mortality would constitute an undue population level effect on either a monthly or seasonal basis. Most regulatory guidance documents avoid setting such a level. The State of Oregon provides the only US Ecological Risk Assessment guidance document (ORDEQ, 1998) with an explicit stated population-level directive related to exposure to hazardous materials. In this document, risk is acceptable when either the chance of exposure exceeding the toxicity reference value (TRV) is <10% regardless of the fraction of the population exposed or the chance of exposure exceeding the TRV is >10% for an individual organism but < 20% of the local population is exposed. Additionally, according to Menzie *et al.* (2008), when there is a population abundance of 90% (i.e., 10% mortality) relative to absolute population abundance, growth or density would be indicative of recovery of a population after a disturbance. These results suggest that using 90% cumulative survival as a threshold in some situations would still allow a population to recover after a disturbance.

Other researchers have suggested a slightly more conservative estimate of a mortality threshold based on the type of pollutant and the way it acts on the population. For example, Lemny (1979) describes a method for using teratogenic deformities in fish as the basis for evaluating impacts of selenium contamination. An index was developed for teratogenic-based assessment of impacts to fish populations. The index is composed of three ratings that signify increasing levels of terata-induced population mortality: 1) negligible impact (<5% population mortality); 2) slight to moderate impact (5–20% population mortality); and 3) major impact (>20% population mortality). Each rating is based on the anticipated population-level impact of the corresponding degree of mortality.

In summary, there is not always an objective technique to determine appropriate thresholds, and professional judgment must usually be relied upon. For thresholds relating to thermal pollution and fish, more research is needed on a species specific basis which should include both acute and chronic effects to the most vulnerable life stages of the fish (i.e. incubating eggs, or larvae). When an actual capacity level cannot be determined, CEAA suggests that an analysis of trends can assist in determining whether goals are likely to be achieved or patterns of degradation are likely to persist. In the absence of defined thresholds, CEAA recommends that the practitioner either: 1) suggest an appropriate threshold; 2) consult various stakeholders, government agencies and technical experts (best done through an interactive process such as workshops); or 3) acknowledge that there is no threshold, determine the residual effect and its significance, and let the reviewing authority decide if a threshold is being exceeded.

5.1.6.9 Monitoring

In order to determine the area of potential influence of the discharge a plume delineation should be completed. A thermal plume delineation should consist of multiple season study conducted at a variety of positions within the water column. This can easily be achieved by anchoring multiple suspended temperature loggers in the water column at areas throughout the mixing zone. Technical guidance on plume dispersion and how to conduct a plume delineation can be found in Environment Canada's "Revised Technical Guidance on How to Conduct Effluent Plume Delineation Studies" (2003).

EPRI (2009) has discussed in detail fisheries monitoring methods. This is summarized below.

Traditional methods for monitoring thermal impacts of fish in the field have included capturing fish in the plume and general area with equipment such as electrofishing, gill nets, and seines. This provides presence or avoidance information but little information on duration of avoidance, number of affected species, age class, or affected critical habitats.

Emergent methods such as video, hydroacoustics, and biotelemetry can monitor the movement and behavior of the fish in real time. Video is non-invasive but has a limited view and water clarity and light can limit the spatial coverage. Software is available that can use footage to track behavior.

Hydroacoustics uses sound to detect objects by reflected signals and is widely accepted for fish surveys. Fish can be monitored through mobile tracking (from boat) or fixed stations. The use of multiple signals (DIDSON – Dual Frequency Identification Sonar) can capture reflected signal from multiple angles to produce 3-D images. Species may not be identifiable with this method, especially if fish are close to the water surface or substrate. Habitat components can be measured simultaneously, although depth and distance measurements are limited.

Biotelemetry involves capturing fish and attaching a tag that transmits signals to a receiver (hydrophones). Movement, depth, temperature, and even muscle movement can be transmitted depending on the device used. Ultrasonic, radio, and acoustic signals are possible in tags. As a general rule tags should not be more than 2% of the weight of the fish, although studies with 8.5% of weight tags have not adversely affected salmonid fish. Fish can be monitored through mobile tracking (from boat) or fixed stations, although multiple fixed stations can produce 3-D tracking. Depth, conductivity, and habitat structure can interfere with signals and can limit effectiveness of this method.

The EPRI document (2009) has developed a monitoring program using telemetry, which may provide the best insight into behavior and responses of aquatic species to thermal discharges. The program has three tiers to determine actions to take.

In Tier I the site and species are selected. The thermal plume must be delineated and a fish community survey in the study area completed. If there are no species in the area, then further study is not required. Different seasons need to be considered, as fish may avoid the plume for only part of the year.

In Tier II fish species that are likely to interact with the plume regularly are chosen. If possible it is best to tag fish from predators, prey and benthic insectivore species as well as adults and juveniles. The study area should include the thermal plume plus the area encompassing 200 m upstream and 200 m downstream of the plume boundaries. A further contingency equal to 10% of the total above-referenced area should also be included. Hydrophones will need to be positioned in the study area so that receiver ranges slightly overlap. A CDMA (code division multiple access) tag is recommended since multiple parameters can be monitored simultaneously. This portion of the study should encompass multiple seasons when there is variability in discharge and species' habits.

Tier III is completed if Tier II shows that fish interact significantly with the plume. Electromyogram (EMG) tags are required, which cost more than the tags used previously and also have a shorter battery life. The EMG tag will transmit electrical activity of muscles to evaluate physiological response to the thermal plume.

This tiered approach will provide a description of the *in situ* fish community, real data relating to behavior and physiological responses of fisheries to thermal plumes, and a deeper understanding of effects of thermal plumes. EPRI (2009) believes that current regulations in many US jurisdictions are more conservative than necessary for the protection of aquatic resources. Further *in situ* study is required to develop consistent and effective discharge criteria across jurisdictions.

5.1.6.10 Management Recommendations

Thermal benchmarks have been summarized for a wide range of species from published literature in **Tables 10** and **11**. CTM and ILT are recommended benchmarks for acute thermal toxicity measurements for a wide variety of warmwater, coolwater and coldwater species. The CLM chronic toxicity test is recommended as it's more realistic relative to thermal plumes, although data is limited for this relatively new benchmark. There is still the need to determine Delta T benchmarks for all life stages of a wide variety of species.

Temperatures are important for fish development and thermal discharges have a variety of effects such as changes to community composition, advanced hatch, changes in development and growth, and survival. With the contribution of other thermal changes such as climate change, urbanization, and agriculture, it is more important than ever to manage thermal inputs. Understanding adverse effects from temperature change to fisheries and implementing restrictions on inputs based on science will ensure healthy fish communities now and in the future.

Thermal discharges can impact fish in multiple ways. The larger the difference in temperature between ambient waters and the plume the greater the potential for harm. Impacts can be reduced by decreasing the volume and/or temperature of the discharge. Future plants requiring cooling water could be built in areas where biological activity is lower or the discharge could be designed to reduce interaction with organisms (Donaldson *et al.,* 2008) such as diffuser type systems. There are also advances cooling technology design that reduces and in some cases eliminates the need for cooling water discharge but these can be expensive options.

When plants shutdown the cooling water discharge also stops. To prevent cold-shock, discharge could be designed to slowly stop and/or never stop. Shutdowns could also be scheduled during non-critical life stages to prevent adverse effects (Donaldson *et al.,* 2008).

The MWAT and use of lethal thermal tolerances are not recommended for benchmark development for cold water species (Donaldson et al., 2008; McCullough, 2010). The use of the biological MWAT is linked to significantly large reductions in growth and impacts the population more than the 10 to 20% usually deemed acceptable for population thresholds. Sublethal effects may cause reduced function that significantly reduces the ability for the affected fish to escape lethal temperatures or predation. Therefore, CTM for acute and CLM for chronic thermal toxicity are recommended for benchmark development.

Ideally the optimal growth curve for the most sensitive species and life stages would be used in benchmark development to provide the most effective protection of thermal regimes and populations (McCullough, 2010). Further study is recommended in this area to ensure available benchmarks are providing adequate protection.

With a variety of heat sources and climate change, cumulative effects can increase water temperatures overall even if each source is managed. Thermal inputs should be managed on a watershed basis to reduce the additive increase of temperature at the downstream end. Best management practices such as maintaining riparian vegetation and controlling runoff from urban and agricultural sources will also reduce non-point source thermal inputs and the overall creep up of temperatures over time.

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