Anthropogene Temperaturveränderungen in Flüssen und Seen – eine Literaturanalyse

Klimbo Teilprojekt 2.a

Literaturstudie zu Auswirkungen, Risikopotenzial und Toleranzbereichen von Wärmenutzungen

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Dezember 2011

Titelbilder

Isar Kernkraftwerk, Deutschland

http://www.zeit.de/wissen/umwelt/2010-09/patrick-moore-atomkraft-interview

Kahlschlag von Flussufergehölz, Deutschland

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Restwasserabfluss im Fluss Traun Deutschland

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Der vorliegende Literaturstudie wurde im Rahmen des Projektes „Klimawandel am Bodensee (KlimBo)“ der Internationalen Gewässerschutzkommission für den Bodensee (IGKB) erstellt. KlimBo wurde durch das INTERREG-IV-Programm „Alpenrhein, Bodensee, Hochrhein“ gefördert.

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Inhalt des Anhangs (Englisch)

Abstract

Water temperature is a key factor for aquatic ecosystems. Temperature mediates survival, growth, reproduction and distribution of organisms. Furthermore, it affects many factors of the physical and chemical aquatic environment. Various human activities alter temperatures in water bodies. The most important anthropogenic sources of such thermal pollutions are changes in river discharge, reservoir operations, changes in riparian shading (riparian forest / woodland) and thermal discharges (such as by thermal power plants). The anthropogenic-induced climate change is another important source of water temperature alteration, which puts a background pressure on aquatic ecosystems by increasing water temperatures (in middle and high latitudes especially by increasing air temperatures). However, this work does not include climate change aspects.

The effects on the thermal regime vary between the different sources of thermal pollution: Reduction in river discharge alters extreme temperature values on all time scales, hypolimnetic water discharge by reservoirs strongly damps the annual temperature cycle and increases the short-term temperature change rates, surface water release by reservoirs increases summer temperatures, reduction of riparian shading increases temperature maxima during spring and summer and finally warm-water discharges increase temperatures all year round.

Beside direct temperature effects, other factors are important for the ecosystem reaction, such as the affected area compared to the total area of the water body, the naturalness of the water body, contaminants in the water and the temperature tolerance of the ecosystem species. Effects of temperature alterations are more pronounced at high temperatures, when temperatures approach or reach lethal limits of many organisms. Therefore, temperature increases in summer are more critical than during the rest of the year. It is easier to assess upper acceptable thermal limits for a water body than for temperature changes. Effects are shown to vary widely between the reported cases. Hence, it is difficult to assess temperature limits. However, Swiss law seems to be rather restrictive by allowing maximum temperature increases of 3 ^{o}C (1.5 ^{o}C if salmonids are present) until an upper temperature limit of 25 ^{o}C. The findings of this literature review suggest that up to 21 ^{o}C increases of about 4 ^{o}C could be tolerable. Between 21 and 28 ^{o}C, further temperature increases of about 2-3 °C could be acceptable, if the natural fish species composition allows. Further heating has strictly to be avoided. Lakes are more sensible to temperature changes than streams. Temperature should not increase more than 1 ^{o}C, if a large area is affected.

Generally, cold water pollution is less critical than warm water pollution. Fixed temperature limits are arguable and many other parameters need to be considered. Therefore, the development of a simple model considering the most important facts could be helpful to assess thermal limits.

Zusammenfassung

Menschliche Aktivitäten beeinflussen natürliche Wasserkörper in verschiedener Weise. Dabei sind Veränderungen der Wassertemperatur von grosser Bedeutung. Temperatur ist einer der wichtigsten Faktoren in aquatischen Systemen. Sie beeinflusst die physikalischen, die chemischen sowie die ökologischen Prozesse in den Gewässern. So beeinflusst zum Beispiel die Temperatur die Stratifikation in Seen, den Sauerstoffgehalt des Wassers und chemische Prozesse laufen gemäss der Reaktionsgeschwindigkeit-Temperatur-Regel bei einer Temperaturerhöhung um 10 °C (Q_{10}) doppelt so schnell ab. Negative Effekte von Wärmebelastungen können wieder auf den Menschen zurückfallen, z.B. durch die Verschlechterung der Wasserqualität (besonders wichtig für die Trinkwasserversorgung) oder Verluste bei der Fischerei.

Diese Arbeit soll die wichtigsten Quellen thermischer Belastung und die dazugehörigen Auswirkungen auf die Wassertemperatur aufzeigen sowie die Reaktion von Ökosystemen erörtern.

Quellen thermischer Belastung von Gewässern

Die wichtigsten anthropogenen Effekte auf Wassertemperaturen entstehen durch Veränderung der Abflussmengen in Flüssen, den Betrieb von Stauseen, das Roden oder Aufforsten von Auenwäldern (Beschattung von Flüssen) und Zuflüsse thermisch veränderten Wassers (vor allem Kühlwasser). Diese Faktoren können sich auch gegenseitig beeinflussen, wodurch sich die Auswirkungen auf die Wassertemperatur sowohl verstärken als auch abdämpfen können. Auch die Auswirkungen des Klimawandels auf Wassertemperaturen können zu diesen anthropogen bedingten Temperaturänderungen gezählt werden. Diese Arbeit erwähnt zwar den Klimawandel, konzentriert sich aber auf die unmittelbaren anthropogenen Quellen lokaler thermaler Belastung, welche einzelne Wasserkörper betreffen. Alle thermischen Beeinträchtigungen betreffen Flüsse, während Seen nur von thermisch verändertem Wasser direkt beeinflusst werden. Durch das Zuströmen von Flusswasser werden Seen indirekt auch von den anderen Quellen thermischer Belastung beeinflusst.

Klimawandel

Die grossflächigste thermale Belastung wird durch den Klimawandel verursacht. Dieser beeinflusst Gewässertemperaturen hauptsächlich auf zwei Arten. Zum einen wird der Energiefluss zwischen Atmosphäre und Wasserkörper verändert (in mittleren und hohen Breiten hauptsächlich durch die erhöhte Lufttemperatur), zum anderen führen modifizierte Niederschlagsmuster zu Änderungen des Abflussregimes in Flüssen. Generell verursacht der Klimawandel eine Erhöhung der Wassertemperaturen und erhöht damit den Druck auf aquatische Ökosysteme, insbesondere in den Sommermonaten. In diversen Gewässern sind schon heute die oberen thermischen Grenzwerte aquatischer Organismen erreicht, insbesondere jene von Salmoniden. Bisherige lokale anthropogene Veränderungen der Wassertemperatur könnten unter künftigen klimatischen Bedingungen zu problematischen Maximaltemperaturen führen.

Abflussmenge in Flüssen

Anthropogene Veränderungen der Flusswasserführung werden hauptsächlich durch den Betrieb von Stauseen, Umleitungen von Wasser (Flüssen) und Bewässerung verursacht, letzteres hauptsächlich in trockenen und warmen Klimaregionen. Reduzierte Abflussmengen erhöhen aufgrund reduzierter Wärmeaufnahmefähigkeit und verkleinerter Wassertiefe die Temperatur-

schwankungen, die Extremwerte sowie die Sensibilität eines Flusses auf weitere temperaturrelevante Einflussfaktoren. Werden die Abflussmengen erhöht, treten die gegenteiligen Effekte ein.

Stauseen

Der Abfluss von hypolimnischem Wasser aus einem Stausee reduziert die saisonalen Temperaturamplituden der betroffenen unterliegenden Flüsse. Im Sommer wird die Wassertemperatur erniedrigt, während sie im Winter erhöht wird. In Alpenflüssen unterhalb von Speicherseen wurde beispielsweise die täglichen Maximaltemperaturen im Sommer und Winter um ~6 °C reduziert respektive um ~4 °C erhöht. Zudem treten die saisonal höchsten Wassertemperaturen im Jahresverlauf um mehrere Wochen verspätet auf. Je nach Klimaregion und Reservoir kann die jährliche Durchschnittstemperatur unverändert bleiben oder sich markant senken. Die Auswirkungen von Stauseen auf die Wassertemperatur können sich je nach ausströmender Wassermasse, des Temperaturunterschiedes zum Restwasser, dem lokalen Klima und der Menge an flussabwärts zuströmendem Wasser über mehrere hundert Kilometer hinziehen. Schwall-Sunk-Betrieb zur Stromproduktion kann zu starken Temperaturänderungen in kurzer Zeit führen und zum wichtigsten Einflussfaktor auf die täglichen Temperaturschwankungen werden. Bei kleineren Speichern erfolgt die Wasserentnahme teilweise an der Wasseroberfläche. Dies führt vor allem zu einer Erhöhung der Sommertemperatur flussabwärts (Erwärmungen von bis zu ~5.5 °C beobachtet). Der Betrieb von Stauseen verändert zudem durch die Umleitung von Wasserströmen und unregelmässigen Wasserauslasses die Wasserführung der unterliegenden Flüsse, was zusätzlich die Wassertemperatur beeinflusst.

Beschattung durch Auenvegetation

Meteorologische Faktoren bestimmen zu einem grossen Teil die Temperatur von Wasserkörpern. In Flüssen ist dabei die kurzwellige Einstrahlung der wichtigste Parameter des Energieaustausches zwischen Wasser und Atmosphäre. Deshalb hat die Beschattung von Flüssen, überwiegend verursacht durch Auenbewaldung, grossen Einfluss auf das Temperaturregime. Dies ist vor allem in Sommermonaten der Fall, wenn die solare Einstrahlung gross ist. Der Effekt ist besonders bei kleineren Wasserläufen mit geringer Wassertiefe ausgeprägt. Während die jährliche Durchschnittstemperatur nur leicht erhöht wird, bewirkt die Rodung von Auenwald teilweise massiv erhöhte Maximaltemperaturen. Typischerweise werden die Temperaturmaxima um ~4°C angehoben, es wurden aber auch bis ~13°C höhere Werte gemessen. Saisonale wie auch tägliche Temperaturamplituden nehmen zu. Zudem tritt eine zeitliche Verschiebung des saisonalen Temperaturverlaufes auf mit früheren jährlicher Maximaltemperaturen. Je nach Gewässer haben Veränderungen der Beschattung durch Auenwälder entlang kurzer Flussabschnitte (ab rund 100 m) einen markanten Einfluss auf das Temperaturregime flussabwärts. In vielen Flussläufen werden regelmässig kritische Maximaltemperaturen erreicht (besonders für Salmoniden), weshalb die Aufforstung von Auenwald verschiedentlich als Massnahme zum Schutz temperatursensibler aquatischen Organismen vorgeschlagen wird.

Thermisch belastete Abflüsse

Thermisch belastete Abflüsse werden mehrheitlich in Flüsse, teilweise aber auch in Seen eingeleitet. Die deutlich wichtigste Ursache dieser Wärmebelastung sind Kühlwasserauslasse von stromproduzierenden thermischen (fossilen oder nuklearen) Kraftwerken. Künftig könnte auch der Betrieb von Wärmepumpen vermehrt thermisch belastete Abflüsse erzeugen. Das Ausmass der Temperaturänderung des Wasserkörpers ist von dem Temperaturunterschied zwischen eingeleitetem- und Umgebungswasser, der Menge des eingeleiteten Wassers sowie dem Volumen des betroffenen Wasserkörpers abhängig (in Flüssen ist der momentane Abfluss entscheidend). Allerdings kann sich der Effekt auf die Wassertemperatur im Gegensatz zu den anderen Quellen anthropogener Temperaturveränderungen sehr lokal in einem Wasserköper auswirken und zu starken Temperaturgradienten führen. Dies ist vor allem in Seen der Fall, da in Flüssen die Durchmischung meist schnell stattfindet. Die Verteilung der thermischen Beeinträchtigung ist unter anderem abhängig von der Art und dem Ort des Auslasses, der Fliessgeschwindigkeit eines Flusses und bei Seen von den Windverhältnissen und den vertikalen Mischungsprozessen. Die Temperatur des ausströmenden thermisch veränderten Wassers gleicht sich mit zunehmender Distanz zur Quelle durch Vermischung mit Umgebungswasser und durch Wärmeaustausch (vor allem mit der Atmosphäre) immer mehr an die natürlich vorherrschende Temperatur an. Bei Seen kann die Temperaturdifferenz zwischen Umgebungswasser und an der Wasseroberfläche eingeleitetem Kühlwasser verringert werden, indem die Wasserentnahme im kühleren Hypolimnion erfolgt. Allerdings bringt dies unerwünschte Effekte auf aquatische Ökosysteme mit sich, da hypolimnisches Wasser generell nähr- und schadstoffreicher ist als epilimnisches Wasser.

Bei Studien zu Kühlwassereinleitungen in Flüsse kommen verschiedene Autoren zum Schluss, dass bisher trotz teilweise massiv erhöhten Wassertemperaturen nur selten bedeutende Veränderungen der aquatischen Ökosysteme verursacht wurden.

Aquatische Biota und Wassertemperatur

Temperatureffekte auf aquatische Organismen können unterteilt werden in letale Effekte, Kontrolleffekte (Beeinflussung physiologischer oder biochemischer Prozesse wie Wachstum, Metabolismus oder Fortpflanzung) und direktive Effekte (Verhalten und Migration). Jede Spezies aquatischer Organismen hat ein genetisch bestimmtes thermisch-finales Preferendum (geht einher mit dem optimalen Wachstum) sowie einen thermischen Toleranzbereich. Letzterer nimmt generell mit steigender Entwicklung des Organismus ab. Fische sind daher die temperatursensibelsten aquatischen Lebewesen. Ausserhalb des thermischen Toleranzbereiches ist ein zeitlich beschränktes Überleben eines Individuums möglich, sofern ein letaler Grenzwert nicht überschritten wird. Neben der genetischen Veranlagung ist auch die Akklimatisierungstemperatur (angewöhnte Temperatur, Anpassung bei Fischen meist schneller als 1 °C in 24 Stunden) für den thermischen Toleranzbereich eines Individuums entscheidend. Diverse weitere Faktoren wie Wasserverschmutzung oder parasitäre Erkrankungen können die thermischen Grenzwerte von Individuen verringern. Mobile aquatische Organismen suchen ihr thermisches Preferendum, wenn in einem Gewässer Regionen unterschiedlicher Temperatur zur Verfügung stehen. Je naturbelassener ein Gewässer, desto eher ist dies der Fall. Solche thermische Rückzugsorte sind für Fische während sommerlichen Maximaltemperaturen oft überlebenswichtig.

Anthropogen verursachte Temperaturgradienten (besonders durch thermisch belastete Abflüsse) machen sich mobile aquatische Organismen (hauptsächlich Fische) zu nutzen, um eine möglichst optimale Körpertemperatur aufrecht zu erhalten. Dafür suchen oder meiden sie beispielsweise Kühlwasserauslasse, abhängig von der Jahres- und Tageszeit sowie ihrem Entwicklungsstand. Erwärmungen im Sommer können bei Überschreiten von Temperaturgrenzwerten letale Folgen haben oder die betroffenen Organismen physiologisch beeinträchtigen. Erhöhte Wintertemperaturen hingegen führen zur Ausdehnung der Vegetationsperiode, der Steigerung der Primärproduktion, grösserer Aktivität aquatischer Organismen und somit zu Eutrophierungstendenzen. In kalten Gewässern führt eine Erwärmung oft zur Zunahme der Diversität und der Anzahl an aquatischen Organismen, wobei jedoch Kaltwasserspezies benachteiligt werden. Zudem vergrössern steigende

Wassertemperaturen die Wahrscheinlichkeit für Parasiten- und Pilzerkrankungen. Werden beispielsweise 15°C während einer längeren Zeitperiode (ca. zwei Wochen) überschritten, erhöht sich die Gefahr eines Ausbruches der Proliferaktiven Nierenkrankheit (PKD) und den damit verbundenen Todesfällen bei Salmoniden massiv. PKD ist einer der Hauptgründe für den Fischrückgang in Schweizer Gewässern.

Die Auswirkungen rascher Temperaturänderungen, hauptsächlich verursacht durch Thermopeaking bei Schwall und Sunk-Betrieb von Reservoiren, sind wenig erforscht. Die wenigen vorhandenen Studien zeigen, dass die rasche Temperaturänderung zu erhöhtem Driften von Invertebraten führt und sich damit die Zusammensetzung der aquatischen Ökosysteme verändern kann.

Schlussfolgerungen

Um die Auswirkungen von thermischen Belastungen auf aquatische Ökosysteme abzuschätzen, muss bekannt sein, welche Aspekte des Temperaturregimes sich wie stark verändern. Diese Veränderungen unterscheiden sich stark zwischen den unterschiedlichen Quellen thermischer Belastung (Tabelle 1). Ebenso sind die Reaktionen der aquatischen Ökosysteme je nach Ausprägung der Temperaturveränderungen völlig unterschiedlich. Veränderungen der Wassertemperatur im Winter ist dabei weniger kritisch als im Sommer, da im Sommer bereits geringe Erwärmungen letale oder stark nachteilige Folgen auf Organismen haben können. Wenn nur eine geringe Fläche des Wasserkörpers betroffen wird, ist eher mit Verhaltensänderungen mobiler Organismen zu rechnen als mit Veränderungen des ganzen Ökosystems. Organismen in verbauten oder verschmutzten Gewässern werden sensibler auf Temperaturänderungen reagieren als solche in naturbelassenen Gewässern, da z.B. schlechte Wasserqualität als Stressfaktor die Temperaturtoleranz beeinflusst und das Fehlen von thermischen Zufluchtsorten das Überdauern der sommerlichen Maximaltemperaturen verhindern kann. Thermisch belastete Gewässer weisen neben dem Temperaturunterschied zum Umgebungswasser oft auch andere chemische und physikalische Eigenschaften auf, z.B. reduzierter oder erhöhter Sauerstoffgehalt, veränderter Nährstoffgehalt (z.B. durch Wasserentnahme von anderen Wasserkörpern oder andere Schichtungsniveau) oder Verunreinigungen durch Zusatzstoffen. Solche Effekte interagieren oft mit der Temperatur und sollten berücksichtigt werden.

Einfacher als die Definition von maximal tolerierbaren Temperaturänderungen ist die Bestimmung von Maximaltemperaturen, die in einem bestimmten Gewässer nicht überschritten werden sollten. Einige wichtige Temperaturgrenzwerte werden in **Tabelle 2** gezeigt. Ab ca. 21 °C nehmen die möglichen Auswirkungen einer Temperaturerhöhung zu. 28 °C sollten nicht überschritten werden. Die thermischen Grenzwerte der temperatursensibelsten Fische eines betroffenen Wasserkörpers können auch eine gute Orientierungshilfe zur Bestimmung von Temperaturober- und Untergrenzen leisen. Beobachtete Auswirkungen bestimmter Temperaturveränderungen auf aquatische Ökosysteme fallen aufgrund der hohen Komplexität und der vielen Einflussfaktoren sehr variabel aus (**Tabelle 3**). Deshalb ist die Festlegung einer fundierten Grösse der akzeptablen Temperaturänderungen schwer vorzunehmen.

Die schweizerische Gesetzgebung, welche maximale Temperaturerhöhungen von 3 °C (1.5 °C in Forellengewässern) bis zu einem oberen Temperaturgrenzwert von 25 °C zulässt, scheint allerdings aufgrund der dokumentierten Fälle restriktiv. Für Fliessgewässer scheinen Temperaturerhöhungen von 4° bei Temperaturen ≤ 21 °C vertretbar. Im Temperaturbereich zwischen 21 bis 28 °C könnten Temperaturerhöhungen von ca. 2-3 °C als akzeptabel beurteilt werden, wenn dies die vorhandene Fischgemeinschaft zulässt. Erwärmungen ab 28 °C sollten hingegen strikte vermieden werden. In Seen sind die Folgen von thermischer Beeinträchtigung deutlich ausgeprägter. Deshalb sollte eine grossflächige Temperaturerhöhung um 1 °C nicht überschreiten. Thermische Belastung durch

Abkühlung ist generell weniger problematisch als Erwärmungen. Verringerung der Wassertemperaturen könnte eine Annäherung an frühere Temperaturregimes bedeuten, da ein Grossteil der Gewässer in den vergangenen Jahrzehnten bereits deutlich erwärmt wurde. Fixe Temperaturgrenzwerte sind aber unter den Autoren der untersuchten Literatur umstritten und die hier empfohlenen Grenzwerte müssten auf jeden Fall an die lokalen Bedingungen angepasst werden. Die Entwicklung eines einfachen Modells, welches die wichtigsten Ökosystem-relevanten Faktoren kombinieren würde, scheint die sinnvollste Möglichkeit zur Ermittlung tolerierbarer thermaler Beeinträchtigung in einem spezifischen Gewässer.

Tabelle 1. *Bedeutung spezifischer Ursachen thermischer Beeinträchtigung auf das Temperaturregime. Wenn bei der Abflussmenge und der Beschattung anstelle einer Reduktion eine Erhöhung erfolgt, treten die gegenteiligen Effekte ein. +, ++ und +++ bedeuten eine geringe, mittlere und starke Erhöhung, während -, - - und - - - für ebensolche Verringerungen stehen. Bei der zeitlichen Verschiebung bedeuten die Pluszeichen ein verspätetes, die Minuszeichen ein verfrühtes Auftreten im Jahr. ○ bedeutet, dass mit keinen signifikanten Änderungen zu rechnen ist.*

Aspekte des Temperatur-Regimes	**Quelle der thermischen Belastungen**				
	Reduktion der Abflussmenge in Flüssen	Stausee mit hypolimnischem Wasserauslass	Stausee mit Wasserauslass an Oberfläche	Reduktion der Beschattung von Flüssen (Rodung)	Thermisch belastete Zuflüsse
Durchschnittliche Jahrestemperatur	+	○ bis - -	+	+	+ bis +++
Sommertemperatur	++	- - -	++	++	+ bis +++
Wintertemperatur	- -	++	○	○	+ bis +++
Maximaltemperatur	+++	- - -	+	+++	+ bis +++
Minimaltemperatur	- - -	++	○	○	+ bis +++
Tägliche Amplitude der Temperatur	+++	○	-	+++	-
Rate der täglichen Temperaturänderung	+	+++	+	+	○
Zeitliche Verschiebung der jährlichen Max/ Min-Temperaturen	○	++	○	- -	○

Tabelle 2. *Einige obere Temperaturgrenzwerte und deren Auswirkungen auf aquatische Organismen.*

T (°C)	Auswirkungen	Kommentare	Autoren und Kapitel
15 °C	Starker Anstieg des Risikos für PKD-Erkrankung und der PKD-bedingten Todesfälle bei Salmoniden	90% der PKD-bedingten Todesfälle von Bachforellen in Schweizer Gewässern treten auf, wenn die Wassertemperaturen 15 °C für > zwei Wochen übersteigen	(Burkhardt-Holm et al. 2005), (Wahli et al. 2002) Kapitel 6.1
20-22 °C	Salmoniden meiden Gebiete mit höherer Wassertemperatur	Zone des Kühlwasser-Einlasses des Kraftwerkes Point Beach im Michigansee (USA)	(Haynes et al. 1989), (Spigarelli et al. 1983) Kapitel 6.2
21 °C	Beginn der letalen Temperaturgrenzwerte einer grösseren Anzahl von Fisch-Spezies		(Bush et al. 1974) Kapitel 6.1
25 °C	Letaler Temperaturgrenzwert von Bachforellen	Bachforellen sind in vielen Gewässern die temperatursensibelste Spezies	(Langford 1990), (Lessard and Hayes 2003) Kapitel 6.1
	Bis zu dieser Temperatur keine bedeutenden physiologischen Schäden bei Organismen	Stechlinsee, Nordostdeutschland	(Koschel et al. 1985) Kapitel 5.2.1
27 °C	Von grösseren Gewässererwärmungen ist aufgrund der ökologischen Schäden abzusehen	Stechlinsee, Nordostdeutschland und vergleichbare Gewässer	(Koschel et al. 1985) Kapitel 5.2.1
28 °C	Überdurchschnittliche Beschädigung von Organismen und Reduktion der Primärproduktion	Stechlinsee, Nordostdeutschland	(Koschel et al. 1985) Kapitel 5.2.1
	Die meisten Fische in Europäischen Flüssen tolerieren diese Temperatur während langen Zeitperioden		(Langford 1990)
28-30 °C	Verschwinden der Hälfte der Fisch-Spezies	Columbia-Fluss, USA	(Bush et al. 1974) Kapitel 6.1
30 °C	Letale Temperaturgrenzwerte einer grossen Anzahl an Fisch-Spezies		(Bush et al. 1974) Kapitel 6.1
	Irreversible Schäden an Phyto- und Zooplankton	Stechlinsee, Nordostdeutschland	(Koschel et al. 1985) Kapitel 5.2.1
32 °C	Maximal tolerierbare Temperatur unbestimmter Dauer bei Warmwasserflüssen, maximaler Temperaturgrenzwert für übliche Invertebraten-Gemeinschaften		(Bush et al. 1974) Kapitel 6.1
32-34 °C	Verschwinden der Hälfte der Fisch-Spezies in Warmwasserflüssen	Flüsse in den USA (Upper / Lower Mississippi, Tennessee River, Delaware River, Sacramento River)	(Bush et al. 1974) Kapitel 6.1

Tabelle 3. *Änderungen der Wassertemperatur und die Auswirkungen auf aquatische Organismen.*

ΔT (°C)	Auswirkungen	Kommentare	Autoren und Kapitel
-5 °C	Bedeutende Gefährdung für aquatische Organismen, Störung der Vermehrung einheimischer Fisch-Spezies	Namoi-Fluss, New South Wales (Australien), Stausee (hypolimnischer Wasserauslass)	(Preece and Jones 2002) Kapitel 3.1
-3-4 °C	Erhöhtes Driften von benthischen Invertebraten	Abrupte Abkühlung (0.24 °C min^{-1}), typisch für Thermo-peaking bei abruptem Wasserauslass aus Stauseen	(Carolli et al. 2011) Kapitel 3.1
-2 °C	Keine Auswirkungen auf die meisten aquatischen Organismen	Abrupte Temperaturverringerung in alpinen Flüssen (Thermo-peaking bei abruptem Wasserauslass aus Stauseen)	(Frutiger 2004b) Kapitel 3.1
-0.5 °C	Keine oder unbedeutende Auswirkungen auf Ökosysteme		(Langford 1990)
+0.5 °C	Keine oder unbedeutende Auswirkungen auf Ökosysteme		(Langford 1990)
+1 °C	Erhöhung der Primärproduktion, starke Zunahme von Makrozoobenthos, Veränderungen in der Struktur der Organismen-Gemeinschaften, Veränderung der Lebenszyklen, Erhöhung der Bakterienaktivität, Trends zu Eutrophierung	Stechlinsee, Nordostdeutschland, thermisches Kraftwerk, lokale Temperaturerhöhungen von bis zu 10 °C (Auswirkungen erhöhten Nährstoffeintrages involviert)	Verschiedene Autoren Kapitel 5.2.1
+2 °C	Keine Auswirkungen auf die meisten aquatischen Organismen	Abrupte Temperaturerhöhung in alpinen Flüssen (Thermo-peaking bei abruptem Wasserauslass aus Stauseen)	(Frutiger 2004b) Kapitel 3.1
+2-3 °C	Erhöhtes Driften von benthischen Invertebraten	Abrupte Erwärmung (0.24 °C min^{-1}), typisch für Thermo-peaking bei abruptem Wasserauslass aus Stauseen	(Carolli et al. 2011) Kapitel 3.1
+3 °C	Beschädigung und Reduktion der Menge und Diversität von Algen, geringe Zunahme an Mollusken und Krebstieren, Verringerung von Fisch-Spezies	Biscayne Bay, Florida (USA), verschieden thermale Kraftwerke	(Levin et al. 1972) Kapitel 5.2
	Verringerung von Kaltwasser-Fisch-Spezies, Erhöhung der Summe an Fisch-Spezies, Strukturveränderungen der Makroinvertebraten-Gemeinschaft	Zehn Flüsse in Michigan (USA), Stausee (Oberflächenwasser-Auslass), Erhöhung der mittleren Sommertemperatur bis 5.5 °C (durchschnittlich 2.7 °C)	(Lessard and Hayes 2003) Kapitel 3.2
	Keine nachteiligen Auswirkungen auf aquatische Ökosysteme	Ebro-Fluss, Katalonien (Spanien), thermisches Kraftwerk, Erhöhung der Sommertemperatur um 2-4 °C (jährlicher Mittelwert 3 °C)	(Prats et al. 2010) Kapitel 5.1
+4 °C	Absterben oder starke Reduktion vieler Tiere und Pflanzen	Biscayne Bay, Florida (USA), verschiedene thermale Kraftwerke	(Levin et al. 1972) Kapitel 5.2

+7°C	Erhöhung der Anzahl und der Spezies an Fischen	Trent-Fluss, Leicestershire (UK), thermisches Kraftwerk, Temperaturerhöhung um 4-12 °C (Mittelwert 7 °C), Maximaltemperatur ≤ 28 °C	(Sadler 1980) Kapitel 5.1
+ 10°C	Meidung oder Ansammlung von Fischen im Gebiet des Auslasses (abhängig von der Spezies, Jahres- und Tageszeit)	Monona-See, Wisconsin (USA), thermisches Kraftwerk, maximale Sommertemperatur bis 35 °C	(Brauer et al. 1974), (Neill and Magnuson 1974), Kapitel 5.2
	Dreimal höhere Primärproduktion, Reduktion der Epifauna und Verschwinden einzelner Spezies in den wärmsten Monaten	Abflusskanal, Maryland (USA), thermisches Kraftwerk, Temperaturerhöhung um 12.7 °C im Winter und 6.4 °C im Sommer	(Levin et al. 1972) Kapitel 5.1
+ 11°C	Gewichtsverlust um 20% bei Welsen	Abflusskanal, Connecticut (USA), thermisches Kraftwerk	(Levin et al. 1972) Kapitel 5.1

ANHANG

Thermal pollution and its impacts on lakes and rivers: a review

Contents

1 Introduction

Human activity affects natural water bodies in various ways. Among the various changes induced by mankind is the temperature of the natural waters. Water temperature is a key factor in aquatic ecosystems. Therefore, there is a need to understand and predict the changes due to anthropogenic water temperature alterations, which potentially may have negative effects on natural systems. Such man-made impacts on natural temperature regimes we will call – in agreement with the existing literature - thermal pollution. The effects of water temperature alterations were first noted in mid nineteen hundreds with increasing industry and energy production and the subsequent use of water (Langford 1990). Since the mid-1960s, the term "thermal pollution" has been in general use (Langford 1990). Problems associated with temperature extremes have been relatively commonplace in continental Europe (Malcolm et al. 2008). In recent years, however, they have increased again in northern rivers which are internationally important for their populations of Atlantic salmon (Malcolm et al. 2008). Due to a changing climate, such problems are expected to increase in the course of this century.

Most physical and chemical processes of water are temperature dependent (Younus et al. 2000, Preece and Jones 2002). Water temperature mediates biological activity in aquatic environment (Cincotta and Stauffer 1984, Babenzien and Babenzien 1985, Langford 1990, Younus et al. 2000, Prats et al. 2007), and it affects the growth, reproduction and distribution of fish (Sinokrot and Stefan 1993, Younus et al. 2000, Malcolm et al. 2008, Broadmeadow et al. 2011). Because of these tight connections between water temperature and biological processes, anthropogenic temperature alterations are expected to have substantial ecological implications (Frutiger 2004b). Changes in aquatic systems might also impact humans. For instance, they can cause a decline of water quality, which is especially important for drinking water supply and recreation. Fisheries are often the first to be adversely affected as high water temperatures might limit fish habitat and increase mortality (Caissie 2006).

Human activity globally alters natural water temperature by changing climate conditions. The on-going climate change is most likely predominantly man-made (IPCC 2007). There are many publications on the effects of on-going or future climate change on water temperatures of natural water bodies (Hondzo and Stefan 1993, Webb and Nobilis 1994, Blenckner et al. 2002, Brooks and Zastrow 2002, Flanagan et al. 2003, Mooij et al. 2005, Wilhelm et al. 2006, Komatsu et al. 2007, Prats et al. 2007, Delpla et al. 2009, Fang and Stefan 2009, Kirillin 2010, Schmid et al. 2011, Van Vliet et al. 2011). Climate change might impact water temperatures directly by increasing the energy input into water bodies and by changing water levels due to alterations in precipitation regimes. For example, (Prats et al. 2007) observed in the Ebro River in Spain in the period 1955-2000 a temperature increase of 2.3 °C (corresponding to an increase of 0.05 °C per year). This temperature alteration was negatively correlated with a decrease in discharge and positively correlated with an increase in air temperature (Prats et al. 2007). Burkhardt-Holm et al. (2005) found an increase of about 1.5 °C in streams in Switzerland over the past two decades, which is the result of climate change and of a more transparent atmosphere due to the reduction of air polutuon.

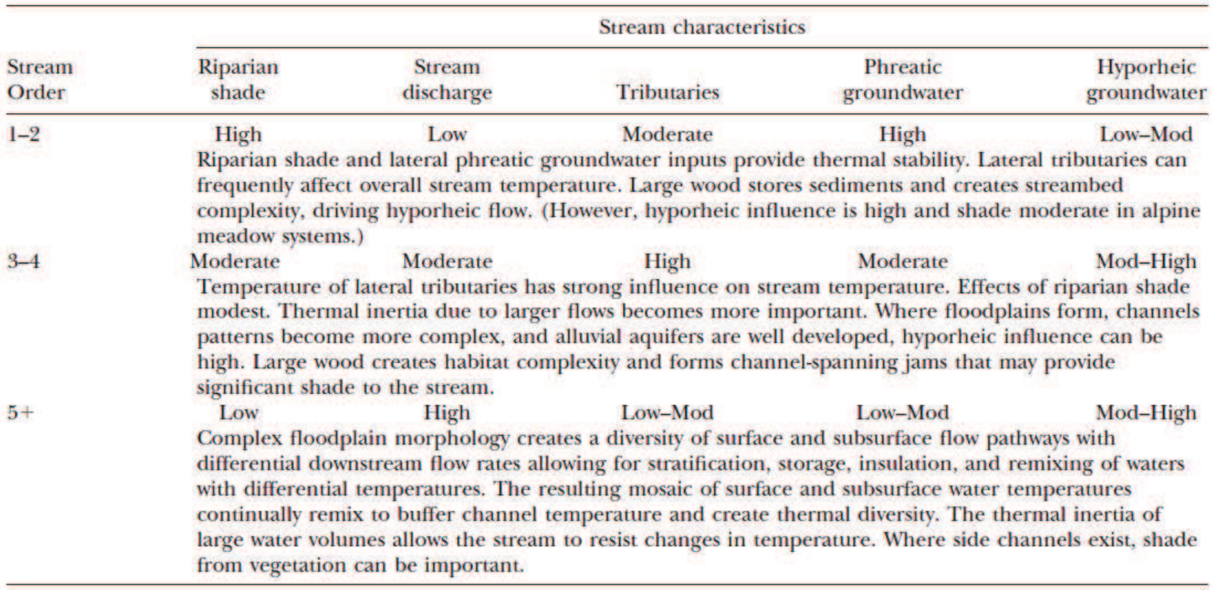
This work will not cover the effects of a changing climate in detail. Instead, it focuses on sources of anthropogenic thermal pollution which impact directly specific water bodies. However, assessments of changes in water temperatures cannot be made without considering climate change. The background water temperatures are generally expected to increase (Mohseni et al. 1999, Maderich et al. 2008), intensifying potential thermal stress on aquatic ecosystems. Many water bodies have reached the upper temperature limits of key species (often salmonids), and with additional warming, temperature limits will be exceeded (Malcolm et al. 2008, Broadmeadow et al. 2011). Because of the threat of global climate change the accurate prediction of water temperatures has become of renewed interest (Sinokrot and Stefan 1993) and it is necessary to implement new regulations that force industries to assess the thermal impact on aquatic environment (Maderich et al. 2008).

According to Prats et al. (2010) the most important sources of water temperature modifications caused by humans are often reservoirs and the use of water for cooling systems. Langford (1990) notes, that changes in riparian shading due to forestation or deforestation is a major factor affecting water temperatures too. Caissie (2006) mentions the same main sources of thermal pollution but includes the reduction of river flow. Other anthropogenic sources that cause alterations of water temperatures include sewage, agricultural drainage, urban run-off (Langford 1990), water withdrawals or channel engineering (Poole and Berman 2001). These effects, however, are generally of minor importance. Therefore the effects of **reservoirs, riparian shading, thermal effluents** and **stream discharge** are reviewed in the following.

Thermal pollutions of different sources might influence each other, which leads to possible damped or intensified effects. For instance, in the Ebro River in Spain the cooling effects of an upstream reservoir and the heating effects of a nuclear power compensated each other during summer (Prats et al. 2010). In winter in contrast, warming effects of released reservoir-water and warming by discharged cooling water added up (Prats et al. 2010). There are many possible interactions of thermal pollutions, which should be considered when assessing the effects of an individual local thermal pollutant.

Without human influence, river temperature forcing parameters can be classified into atmospheric conditions, groundwater inflow, topography, stream discharge and streambed (Caissie 2006). Energy exchange with the atmosphere and water body occurs at the water-air boundary layer by incoming short wave radiation, incoming and outgoing long wave radiation, latent heat exchange (especially by evaporation) and sensible heat flux (Webb and Zhang 1997). Normally, heat exchange with the atmosphere is the dominant factor controlling energy content of a stream (e.g. Webb and Zhang 1997). Due to the strong dependence on atmospheric conditions, water temperatures in streams follow two cycles: a seasonal cycle and a diurnal cycle, in which the diurnal cycles are superimposed on the seasonal cycle (Sinokrot and Stefan 1993). Records of stream water temperatures and air temperatures show that stream water temperature follows the air temperature closely on a seasonal time scale (Sinokrot and Stefan 1993). The importance of different temperature drivers depend on the river size as shown in Table 1. In lakes, the most important energy fluxes occur by heat exchanges with the atmosphere and water in- and outflow. If not very shallow, lakes are thermally stratified with one or two annual episodes of full mixing (for most lakes in temperate climates). Internal mixing processes are important for the temperature distribution in a lake. Human impacts to the thermal regime of water bodies can occur by modifying the natural forcing factors or by releasing water at temperatures different from the natural ones (Prats et al. 2010).

Table 1. *Relative influence of stream characteristics on temperature in small, medium and large streams. Sorve of information: (Poole and Berman 2001).*



	Stream characteristics				
Stream Order	Riparian shade	Stream discharge	Tributaries	Phreatic groundwater	Hyporheic groundwater
1–2	High	Low	Moderate	High	Low–Mod
	Riparian shade and lateral phreatic groundwater inputs provide thermal stability. Lateral tributaries can frequently affect overall stream temperature. Large wood stores sediments and creates streambed complexity, driving hyporheic flow. (However, hyporheic influence is high and shade moderate in alpine meadow systems.)				
3–4	Moderate	Moderate	High	Moderate	Mod–High
	Temperature of lateral tributaries has strong influence on stream temperature. Effects of riparian shade modest. Thermal inertia due to larger flows becomes more important. Where floodplains form, channels patterns become more complex, and alluvial aquifers are well developed, hyporheic influence can be high. Large wood creates habitat complexity and forms channel-spanning jams that may provide significant shade to the stream.				
5+	Low	High	Low–Mod	Low–Mod	Mod–High
	Complex floodplain morphology creates a diversity of surface and subsurface flow pathways with differential downstream flow rates allowing for stratification, storage, insulation, and remixing of waters with differential temperatures. The resulting mosaic of surface and subsurface water temperatures continually remix to buffer channel temperature and create thermal diversity. The thermal inertia of large water volumes allows the stream to resist changes in temperature. Where side channels exist, shade from vegetation can be important.				

2 Stream discharge

There are many sources for anthropogenic flow alterations, but reservoir operations, water diversion and irrigation (especially in dry and warm climates) may have the greatest impact on flow. If the runoff is reduced, the water body in the river becomes thinner and subsequently the river water gets more sensitive to any temperature driver, while increased runoff reduces the sensitivity to temperature changes. Anthropogenic influence on discharge highly interacts with other sources of thermal pollution. Hence, in the following chapters, the effects low discharge will be discussed again.

Due to their low thermal capacity, small streams are highly vulnerable to increase in man-made heat input (Caissie 2006). The same holds true for river with reduced discharge (**Figure 1**), temperature fluctuations are much higher during periods of low discharge. (Webb and Nobilis 1994) analysed the trends for individual months and found that the Danube at Linz was most sensitive to a changing climate in the autumn period of low flows when the thermal capacity of the river is at its lowest.

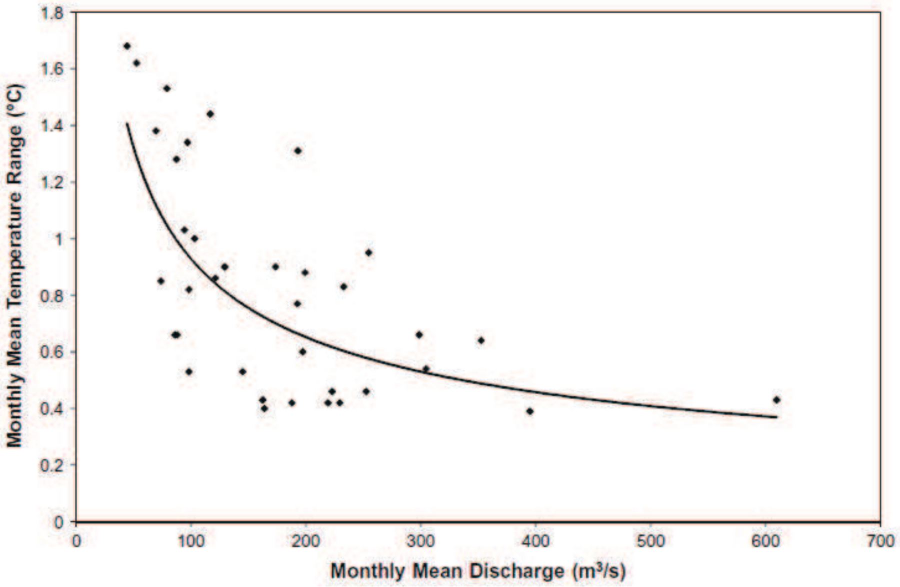


Figure 1. *Relationship between the monthly mean temperature range and the monthly mean discharge of the Ebro River at Escatrón (Spain). Measurements were taken from February 1997 to January 2000 (Prats et al. 2010).*

Large reductions in flow make rivers downstream vulnerable to temperature extremes as the stream's assimilative capacity for heat gets reduced (Poole and Berman 2001). When there is little water left in the river bed, water temperatures are higher than normal during summer and lower than normal during winter with higher daily temperature amplitudes (Frutiger 2004a). Wider daily water temperature ranges are observed at tributaries compared to those of the main stream (Prats et al. 2010). The diel temperature magnitude is typically inversely proportional to flow rate (Constantz et al. 1994). In addition to meteorological forcing, other temperature relevant factors like inflows or riparian shading get more important at little flow rates. According to Frutiger (2004b), reduced discharge might lead to the loss of species which are unable to tolerate the more pronounced temperature peaks. Hydropower abstraction reduces flow massively (Meier et al., 2003) and can cause locally massive temperature changes (Meier and Wüest, 2004).

3 Reservoirs

Water temperatures of rivers are altered when they pass through a reservoir. Therefore, the effects on downstream temperatures depend on actual condition of the reservoir stratification, depth of the outlet and reservoir operation. Generally, seasonal temperature amplitudes in the downstream river are decreased while fast temperature variations are increased. If the reservoir is used for energy production, water is usually not released after the dam but far downstream at lower elevations. Furthermore, water is released when the demand for electricity is high, i.e. mainly during workdays. Hence, reservoir operation might lead to spatial and temporal reduction of river discharge with the corresponding effects on water temperature (chapter 2).

A sudden release of great water masses might cause a fast temperature change of several ^{o}C downstream river. The increase of discharge is called hydro-peaking, the corresponding temperature effect thermo-peaking. Depending on the season and reservoir type, water release might cause warm- or cold water pollution. The different reservoir operations, affecting the thermal regime of the downstream river, can be classified into hypolimnetic water release, surface water release and/or thermo-peaking. Thermo-peaking might occur at hypolimnetic as well as surface water release.

River temperature alterations in areas with hydropower production are widespread. For instance in Switzerland, the temperature regime of most major streams is severely altered (Frutiger 2004a). Beside temperature effects, flow changes influence aquatic life as well. Invertebrate species richness typically has been found to decrease below dams, although total abundance may increase (Jackson et al. 2007). Certain mayfly species are sensitive to flow regulation and so may disappear below dams, while other taxa, such as oligochaetes that are tolerant of flow change, often reach high abundance (Jackson et al. 2007). Furthermore, the water quality variables of release reservoir water can be different than the ones in the uninfluenced stream (Lessard and Hayes 2003). The geographic extent of thermal pollution downstream of a dam depends on discharge rate, river flowing depth, local climate (Sherman et al. 2007), the contribution of tributaries and the temperature difference between reservoir release and natural stream flow (Preece and Jones 2002). According to (Sherman et al. 2007) there is a large number of dams in South-eastern Australia which cause above all cold water pollution, affecting many hundreds of kilometers of downstream river. (Preece and Jones 2002) found that in an average year, water temperatures of the Namoi River in South-east Australia were restored to within 1 °C of the pre-dam condition within 100 km downstream from the dam (**Figure 2** and **Figure 3**). To reduce the temperature effects on the downstream river, sometimes multiple-outlet structure were built to provide a selective withdrawal from different depths in a reservoir (Sherman et al. 2007).

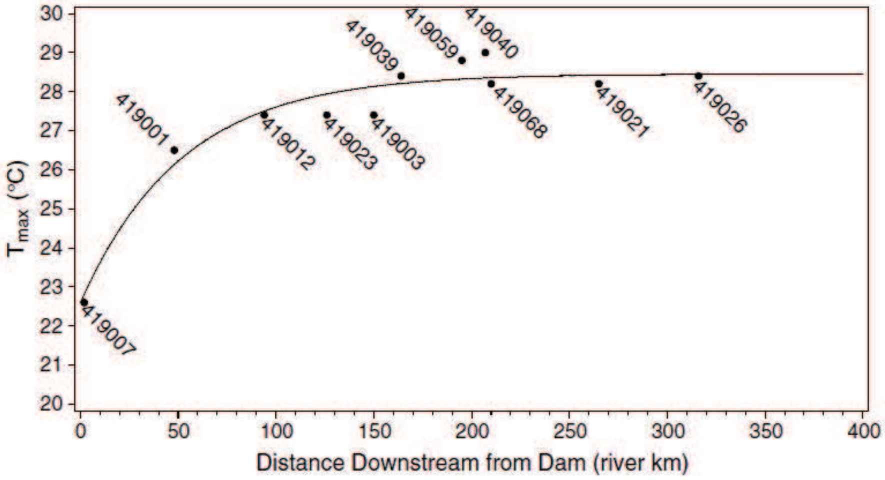


Figure 2. *Maximum water temperature of the Namoi River downstream from Keepit Dam, South-east Australia. Points and numbers show the measurement points downstream. The solid line indicates a negative exponential growth curve fitted against distance from the dam. Negative exponential growth seems to be a good approach of the recovery of the temperatures after a thermal disturbance (Preece and Jones 2002).* Note the long distances for re-adjustment.

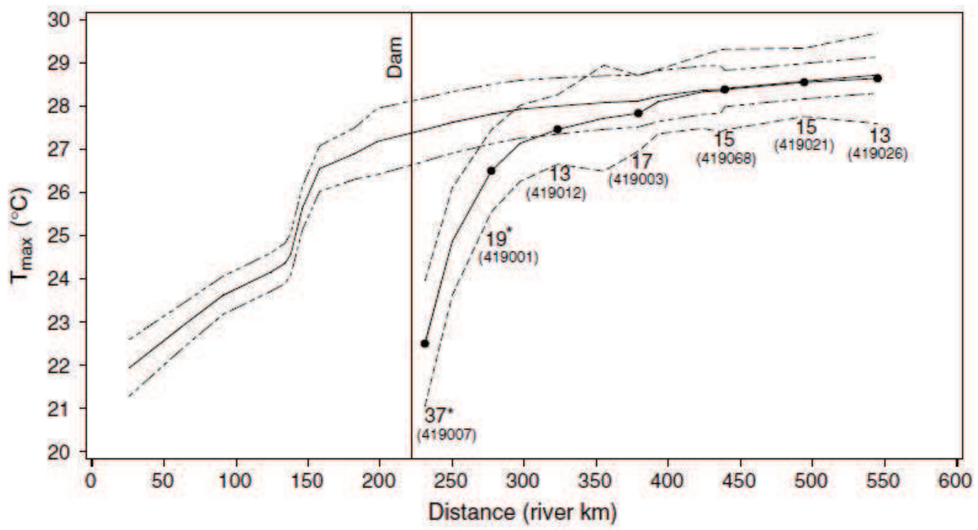


Figure 3. *Maximum temperatures of Namoi river. Both downstream values and pre-dam (Keepit Dam) reference conditions are modelled. Broken lines are 95% confidence intervals of the respective mean fitted values. The downstream measurement stations are marked (●) (Preece and Jones 2002).*

3.1 Hypolimnetic water release

In general, seasonal temperature is more constant downstream of a stratified reservoir with hypolimnetic release: summer temperature are lower, winter temperature higher than without reservoir (**Figure 4**; (Langford 1990, Webb and Walling 1993, Preece and Jones 2002, Frutiger 2004a, Jackson et al. 2007). The absence of extreme temperatures below reservoirs might cause the loss of certain species which need peak temperatures to trigger specific processes in their life cycle (Frutiger 2004b). Typically, water released from greater depths in the hypolimnion of a reservoir has a temperature of around 4 °C (e.g. Frutiger 2004a). Even though the diurnal and seasonal regimes may be extensively altered, the mean annual temperature may not be greatly modified (Langford 1990). In some cases, however, the cooling effect by reservoir water release is dominant, leading to a substantial annual heat deficit (Frutiger 2004a) which favours cold stenotherm species (Frutiger 2004b). The deeper the outlet, the colder the temperature because the sun's shortwave radiation does not penetrate sufficiently far to warm the reservoir at the level of the outlet (Sherman et al. 2007). Cold water pollution in Australian rivers was found to favour alien cold water species such as trout, giving them a competitive advantage over native warm water species like the native Murray cod (Sherman et al. 2007).

An instructive example of temperature effects caused by hypolimnetic water release is shown by (Preece and Jones 2002). They studied the effects of the Keepit Dam on the Namoi River in South-east Australia. The temperature modifications by the dam pose a major threat to the aquatic biota of the lowland streams, which can stretch over long distances of the rivers (Preece and Jones 2002). The effect was greatest immediately downstream from the dam where the annual maximum daily temperature was ~5 °C lower and occurred three weeks later than the pre-dam conditions. The cooling effect was sufficient to disrupt thermal spawning cues for selected Australian native fish species. The magnitude of disturbance progressively diminished with distance from the dam. Beside the daily and seasonal reduction of the temperature amplitude, the time of maximum temperatures is shifted as well. Just below the Keepit dam, the maximum annual temperature occurred in February, several weeks later than natural (Preece and Jones 2002).

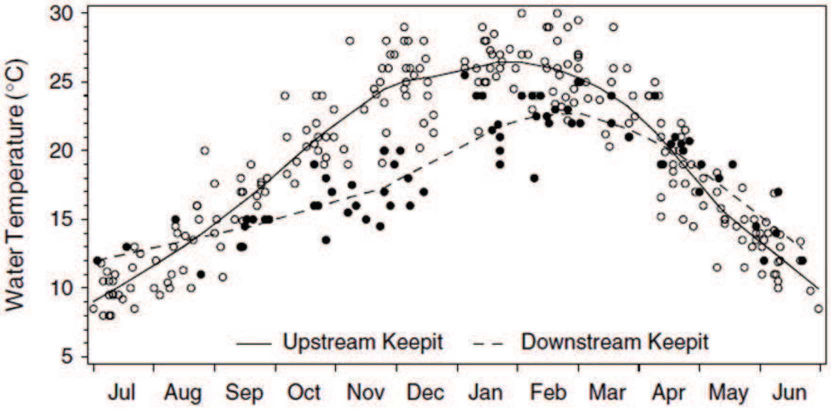
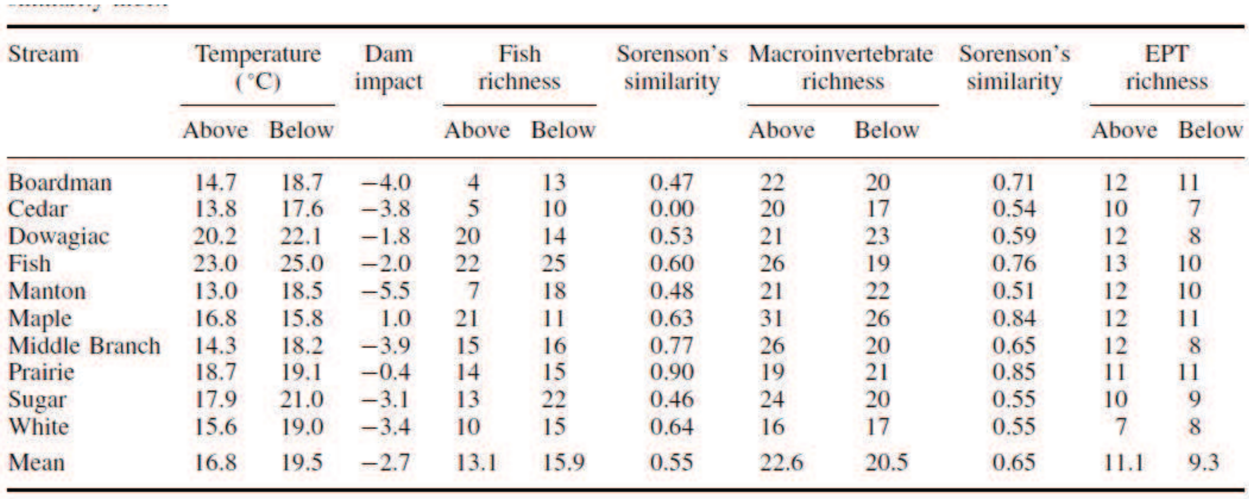


Figure 4. *Water temperature of the Namoi River upstream (○) and downstream (●) from Keepit Dam. Lines are fitted LOESS curves. Data shows typical temperature alterations of a reservoir with hypolimnetic water release, cooling downstream summer temperatures and warming winter temperatures with a delay of annual maximum temperature (Preece and Jones 2002).*

3.2 Surface water release

While most of the big reservoirs release water at great depths in the hypolimnion, some small dams release surface water. The effect of such dams on downstream thermal regimes is a major habitat concern for many cold-water systems (Lessard and Hayes 2003). Surface release leads generally to higher temperatures in the downstream river (Hamblin and McAdam 2003, Lessard and Hayes 2003). However, at their study site (10 streams in Michigan, USA, measurements 1998 and 1999), (Lessard and Hayes 2003) did not find year-round temperature alterations. Only the mean summer temperature was significantly different between up- and downstream sections away from what would be predicted for unregulated streams (Table 2; (Lessard and Hayes 2003). These increases in temperature were maintained at least 2 – 3 km below the dams with shifts in the macro-invertebrate community, increased fish species richness and reductions in brown trout, brook trout and slimy sculpin population densities (Lessard and Hayes 2003). The higher temperatures downstream coincided with lower densities of several cold-water fish species than upstream (Lessard and Hayes 2003). However, (Lessard and Hayes 2003) observed an increased overall fish species richness downstream, above all at high reservoir impact and therefore high increase in water temperature. Fish species richness was most correlated with mean summer temperature, but conductivity, total phosphorus, stream discharge, and modal substrate size were also strongly correlated with richness. (Lessard and Hayes 2003) assume that aggregation of fish species attempting to move upstream may have played a role for the increased fish richness, but they consider the mean summer temperature as the most important factor. To protect downstream rivers from adverse temperature effects by surface release of reservoirs, (Hamblin and McAdam 2003) suggest selective withdrawal of hypolimnetic water added to the surface released surface water.

Table 2. *Mean summer temperature alteration and effect on fish and macro-invertebrate richness in ten streams in Michigan (USA) caused by small reservoirs with surface outflow. In all but in one case there is a temperature increase (up to 5.5 °C). Dam impact describes temperature above minus temperature below. Changes in fish and macro-invertebrates community composition are shown by the Sorenson's similarity index (Lessard and Hayes 2003).*



Stream	Temperature (°C)		Dam impact	Fish richness		Sorenson's similarity	Macroinvertebrate richness		Sorenson's similarity	EPT richness	
	Above	Below		Above	Below		Above	Below		Above	Below
Boardman	14.7	18.7	−4.0	4	13	0.47	22	20	0.71	12	11
Cedar	13.8	17.6	−3.8	5	10	0.00	20	17	0.54	10	7
Dowagiac	20.2	22.1	−1.8	20	14	0.53	21	23	0.59	12	8
Fish	23.0	25.0	−2.0	22	25	0.60	26	19	0.76	13	10
Manton	13.0	18.5	−5.5	7	18	0.48	21	22	0.51	12	10
Maple	16.8	15.8	1.0	21	11	0.63	31	26	0.84	12	11
Middle Branch	14.3	18.2	−3.9	15	16	0.77	26	20	0.65	12	8
Prairie	18.7	19.1	−0.4	14	15	0.90	19	21	0.85	11	11
Sugar	17.9	21.0	−3.1	13	22	0.46	24	20	0.55	10	9
White	15.6	19.0	−3.4	10	15	0.64	16	17	0.55	7	8
Mean	16.8	19.5	−2.7	13.1	15.9	0.55	22.6	20.5	0.65	11.1	9.3

3.3 Thermo-peaking

According to (Lowney 2000), the strongest influence on diurnal temperature variation in a river regulated by a large reservoir may be the temporal signature of the reservoir release itself. Hydroelectricity production is mainly dependent on electricity prices. Therefore, water discharge occur irregular with pronounced peaks. Such sudden water releases from hydropower plants can cause abrupt temperature variations, typically on a daily basis (Lowney 2000, Toffolon et al. 2010, Carolli et al. 2011). In high elevation reservoirs, temperature of released water can be several ^{o}C different from temperature of the receiving body (Toffolon et al. 2010). Downstream rivers are usually warmed up during winter (warm thermo-peaking) and cooled down in summer (cold thermo-peaking; (Toffolon et al. 2010, Zolezzi et al. 2011). In the Noce River in Northern Italy for instance, within less than half an hour the water depth can nearly double and the stream temperature might increase up to ~3 – 4 °C (Toffolon et al. 2010). At this study site, warm thermo-peaking occurs from September to January and results in additional (up to 4 °C) heating with respect to that associated with the natural diel fluctuations (Zolezzi et al. 2011). From March to July, cold thermo-peaking cools down the temperature (up to 6 °C), in contrast with the natural trend that would result in heating during the day (Zolezzi et al. 2011).

The effect of the fast temperature changes caused by reservoir release on the riverine biota and bio-chemical processes in the downstream river are not well investigated (Zolezzi et al. 2011). Most biological studies of thermal influence on fisheries concentrate on the response of particular species to step changes in temperature over long periods of time, rather than species response to changes in the diurnal temperature regime (Lowney 2000).

4 Riparian shading

Water temperatures in water bodies are controlled to a great extent by weather conditions. Parameters such as air temperature, solar radiation, relative humidity, cloud cover (less so wind speed) play an crucial role in the heat exchange between the atmosphere and rivers (Sinokrot and Stefan 1993). However, short wave radiation has been found to be the most important component of heat flux across the stream water surface and hence the stream heat budget (Sinokrot and Stefan 1993, Johnson and Jones 2000, Younus et al. 2000, Moore et al. 2005). Therefore, river shading – usually due to riparian woodland - reduces the energy entering the rivers and consequently takes influence on the heat budget. In contrast, removal of riparian vegetation amplifies the influence of short wave radiation (Johnson and Jones 2000). The riparian zone is defined by Poole and Berman (2001) as the land area influenced by stream-derived moisture. In contrast to open water bodies such as lakes, streams are fairly narrow, and if the banks are lined by trees, streams will experience significant sun shading and wind sheltering (Sinokrot and Stefan 1993, Poole and Berman 2001). Wind sheltering reduces the heat fluxes between water surface and atmosphere. Furthermore, riparian woodland reduces the nocturnal net radiation deficit by damping (reducing) the streams radiative energy loss of long wave emission (Sinokrot and Stefan 1993). In combination with the reduced evaporative heat loss due to wind sheltering (Sinokrot and Stefan 1993), less pronounced minimum temperatures on shaded compared to open sites might be observed during night (e.g. Broadmeadow et al. 2011). These combined effects can result in substantially moderated thermal regimes on all time scales in forested areas compared with more open vegetation (Malcolm et al. 2004). However, the effect of damped minimum temperature is minor. Therefore, annual mean temperature at shaded sites is typically slightly lower than at open sites. Temperature influence of riparian shading can be measured by comparing stream temperatures before and after forest clearance (**Figure 5**; Johnson and Jones 2000, Moore et al. 2005) as well as comparing woodland sites with open (e.g. moorland) sites (Malcolm et al. 2008, Imholt et al. 2010, Broadmeadow et al. 2011).

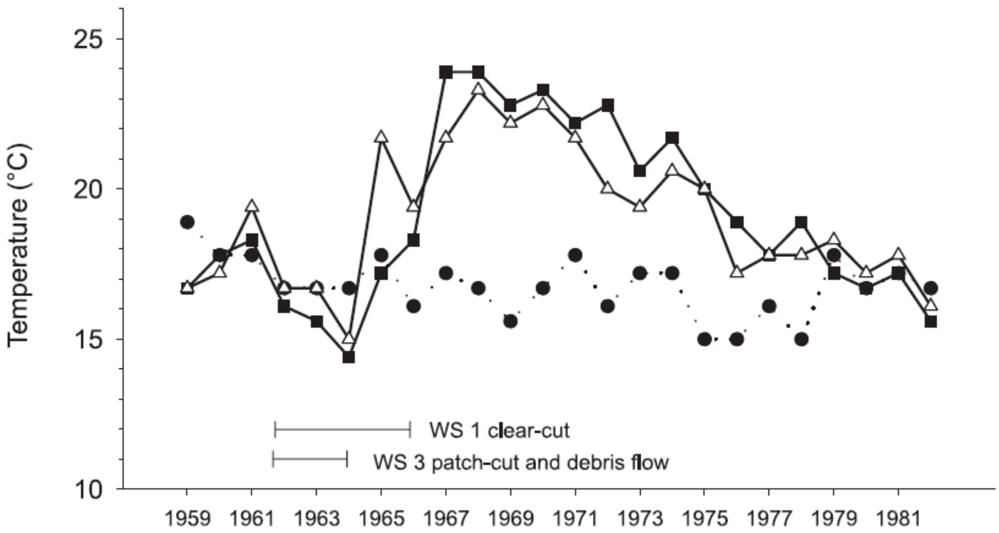


Figure 5. *Maximum summer stream temperature (June 1 – September 30, 1959 – 1982) in western Cascades, Oregon (USA). Site WS 1 (squares) was clear cut and slash burned between 1962 and 1966. WS 2 (circles) was non-harvested. WS 3 (triangles) had roads constructed between 1959 and 1961, was patch-cut during 1962-1963, and had debris flows scour the channel in 1964. The reduction or riparian shading shows pronounced effects on maximum temperatures. With re-growing vegetation, maximum temperature decreased gradually until it returned to pre-harvest levels after 15 years (Johnson and Jones 2000).*

The extent of influence of riparian shading on stream temperature depends on its characteristics as actual discharge, river width, river depth, flow direction, surrounding topography, latitude and the leave cover of trees on stream banks. Malcolm et al. (2008) found the most marked effects on a small stream (Girnock River), where the width-depth ratio is relatively high and the cover from individual tree crowns can shade a significant proportion of the stream width. In contrast, (Prats et al. 2010) found the effects of riparian shading to be a minor on temperatures for their study site at the Ebro River in Spain with stream widths of 50 – 200 m. Confirming these findings, (Malcolm et al. 2004) state that high width-depth ratios often generate large diel temperature fluctuations and warmer day- and cooler night-time temperatures than sites with lower width to depth rations. The effect of riparian shading is strongest when there is much incoming short wave radiation. (Malcolm et al. 2008) observed an increasing effect during spring to a maximum in summer, before the effect decreased again in autumn (**Figure 6**).

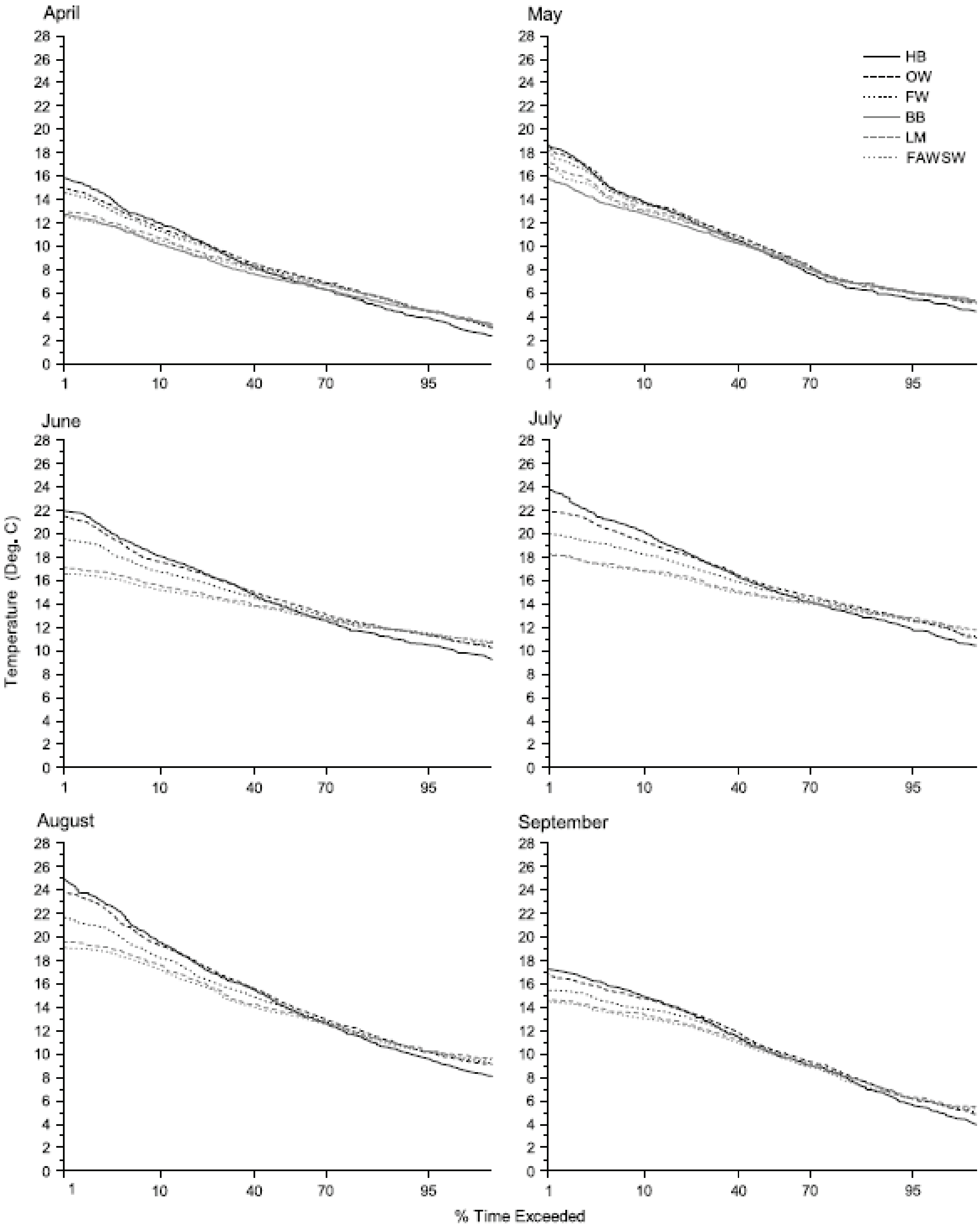


Figure 6. *Monthly temperature duration curves April-September 2003 of a river in western Scotland. HB and OW are open sites, FW, LM and FAWSW or woodland sites with increasing length of upstream woodland. BB are measurements of another tributary (Malcolm et al. 2008).*

(Malcolm et al. 2008) estimated the minimum length of upstream woodland for maximal cooling to about 1.5 – 2 km. (Broadmeadow et al. 2011) showed that even shorter distances of upstream riparian woodland might affect river temperature. They found that a 50% increase in riparian cover over 100 m, 500 m or 1 km would likely be associated with a reduction in mean temperature of ca.

1 °C. The relationships between temperature and percentage of riparian cover for 100 m, 500 m and 1 km upstream were not statistically significant (Broadmeadow et al. 2011). According to (Broadmeadow et al. 2011), inter annual temperature variations on shaded sites are smaller than on open sites. The authors measured maximum summer temperatures variations of typically 2.5 – 5°C for open compared to < 2°C for shaded sites. Annual average water temperatures in shaded or partially shaded sites are slightly lower than in open sites. (Broadmeadow et al. 2011) measured a difference of 1 °C. The difference between open and woodland sites is most apparent in the summer months, with the woodland shading apparently acting to reduce maximum temperatures and mediate temporal fluctuations (Imholt et al. 2010). Changes in riparian shading might also alter the annual temperature evolution. After logging, the largest increases in stream temperature did not occur at the usual time of maximum stream temperatures, but in early summer, which coincided with the timing of maximum solar inputs (Johnson and Jones 2000). According to (Johnson and Jones 2000), such shifts in seasonal timing may have subtle but important effects on stream biota. The authors found a coincidence between the time shift and the early development stages of many organisms, which possibly leads to different types of impacts than if temperature increases primarily occurred at times of historical maximum summer temperatures. (Broadmeadow et al. 2011) found at the two study catchments in southern England substantially lower maximum summer temperatures for shaded than for open sites (14.3 - 19.2°C and 17.0 - 23.1°C, respectively). Such differences of around 4°C seem to be typical (Imholt et al. 2010). Greater impact of riparian shading is possible though. (Johnson and Jones 2000) measured an increase of 7 °C of the maximum temperature after removal of riparian woodland, while (Harris 1977) even reported an increase of 13 °C.

Water temperature in the daily time scale depends on the heat exchange with the environment (Prats et al. 2010). This heat exchange is foremost driven by incoming short wave radiation, although air temperature has some influence too (Sinokrot and Stefan 1993, Prats et al. 2010).

Riparian woodland reduces diurnal stream temperature fluctuations (**Figure 7**; (Johnson and Jones 2000, Malcolm et al. 2004, Broadmeadow et al. 2011). In undisturbed forested streams, seasonal temperature variation in seasonal scale are greater than in diurnal scale (Johnson and Jones 2000). However, (Johnson and Jones 2000) found after clear cutting and burning the riparian woodland, the diurnal range of temperature in early summer (6–8°C) was nearly as great as the seasonal range in an unharvested site (8 °C). (Sinokrot and Stefan 1993) observed diurnal temperature fluctuations of up to 5 °C in shallow streams with little shading in the upper Midwest of the United States. (Johnson and Jones 2000) measured a pronounced increase of diurnal water temperature fluctuations in June after logging from approximately 2 to 8 °C. The daily maximum temperature might increase dramatically after a logging event. (Moore et al. 2005) describe an increase of up to 5 °C in daily maximum temperatures which was positively associated with maximum daily air temperature and negatively with discharge. Riparian shading might, beside reducing maxima temperatures, increase the minima temperatures during night (**Figure 7**, Broadmeadow et al. 2011). This due to the long wave radiation from overhanging riparian vegetation (Webb and Zhang 1997).However, this effect is rather small and compared to the reduction of maximum temperatures insignificant.

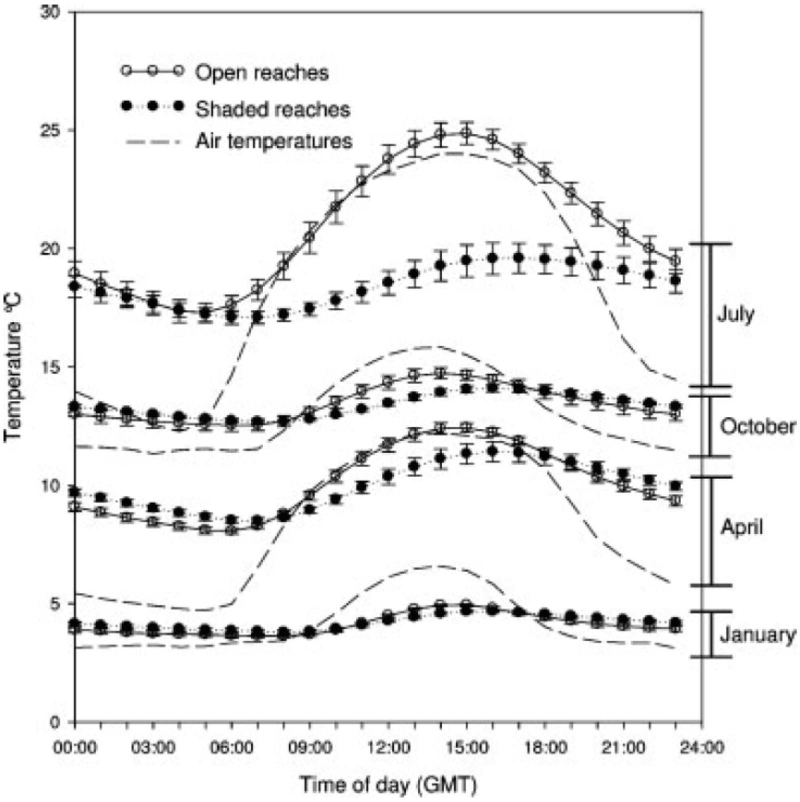


Figure 7. *Mean diel variation of water (five sites) and air temperature (one site) in a small river in south England for January, April, July and October 2006. Vertical bars show standard error (Broadmeadow et al. 2011)*

Due to its capacity to reduce maximum water temperatures (Malcolm et al. 2004, Broadmeadow et al. 2011), riparian shading is highly important for aquatic life. For example, upstream riparian shading restricts the number of days that thermal thresholds for trout are exceeded (Broadmeadow et al. 2011). At their study site in south England, (Broadmeadow et al. 2011) found that a relatively low level of shade (20–40%) was effective in keeping summer temperatures below the incipient lethal limit for brown trout, but ca. 80% shade generally prevented water temperatures exceeding the range reported for optimum growth of brown trout. The smaller temperatures in wooded sites affect the metabolism of aquatic life. Electro-fishing surveys showed that in general, fish in wooded sites grew at a slower rate than those in the open moorland (Malcolm et al. 2008). (Imholt et al. 2010) state that there are significant differences in the mean size of mayfly (*Baetis*) between moorland and woodland sites, with differences occurring primarily during the summer months. *Baetis* in moorland size were larger earlier in the year than in woodland sites, but later in the year (growth of the next generation) it got reversed (Imholt et al. 2010). Overall, the results indicate that subtle differences in growth and life cycles, linked to differences in riparian cover and stream water temperature, can occur over short distances in upland streams (Imholt et al. 2010). The increased daily temperature amplitudes after cutting riparian wood might cause stress for aquatic organism. However, the specific tolerances and responses of most organisms to rapid temperature fluctuations are not well known (Johnson and Jones 2000).

Expansion of riparian woodland was suggested as a water management tool to prevent water temperature of exceeding incipient lethal limits of organisms (especially salmonids) and to keep it adapted for native species in future (Malcolm et al. 2008, Imholt et al. 2010, Broadmeadow et al. 2011). Such measures are above all of relevance considering increasing temperature stress by climate change (Broadmeadow et al. 2011). (Broadmeadow et al. 2011) found at their study site in south England, that planting of new riparian woodland to achieve ca. 20% canopy cover along at least a 500 m reach of small streams could be effective in preventing current summer maximum water temperatures from exceeding lethal limits for salmonids and other fish. They expect higher levels of riparian woodland to be needed to address future climate warming. The authors quantified the effect of riparian shading by linear regression to a reduction of the maximum summer temperature by 2.0 – 2.3°C due to an increase of 50% in upstream riparian cover. Even though riparian shading has strongest effects on small streams, it might be beneficial for larger rivers by the cooler tributary inflows (Malcolm et al. 2008).

5 Thermal discharges

Of all sources of thermal pollution, thermal discharges affect natural water bodies most directly and obviously. For the most part, thermal discharges originate from cooling water release by thermal power plants (nuclear and fossil; (Langford 1990), leading to warm water pollution. In the future, heat pump operation might become an additional source of thermal discharges. In contrast to power plants, heat pumps can – depending on the release scheme - produce warm or cold water pollution. However, no work about cold water discharges were found, cold water pollution was only reported by release of hypolimnetic reservoir-water. Beside man-made thermal discharges, there are also natural ones in geothermal active regions like hot springs and geysers. Therefore, thermal discharges are not a new phenomenon in aquatic systems (Langford 1990). In contrast to other sources of thermal pollution, thermal discharges might lead to very local temperature alterations with high small scale temperature gradients. Normally, river water is used for cooling water cycles and there are many cases studied. However, sometimes lakes are affected by thermal discharges and with Lake Stechlin in northeast Germany there is a well-documented example on its consequences.

In the water bodies, discharged water of increased temperature forms a thermal plume. The whole plume-area can be considered as mixing-zone, where temperature decay and heat loss occur and the effluent water gets gradually diluted by cold ambient water (Langford 1990). The concept of a mixing zone is well established in the history of water quality (Langford 1990). It assumes that an effluent from a discharger requires a certain zone for mixing with the general water body (Langford 1990). Traditionally, the mixing zone has been considered a zone in which the receiving-water standards do not apply (Coutant 1999). However, it is difficult to estimate the expansion of the mixing zone due to the high complexity of processes of transport, mixing and the heat exchange with the atmosphere (Maderich et al. 2008). Mixing zones tend to be site-specific, and the size and extent is dependent on many factors including the size of the discharge in relation to the receiving water, hydrography, weather (e.g. wind, see Figure 16), operating conditions and outfall design and location (Langford 1990). In rivers mixing processes are much stronger than in lakes. Therefore, the flow velocity is one of the most important factors for the expansion of a thermal plume (**Figure 8**).

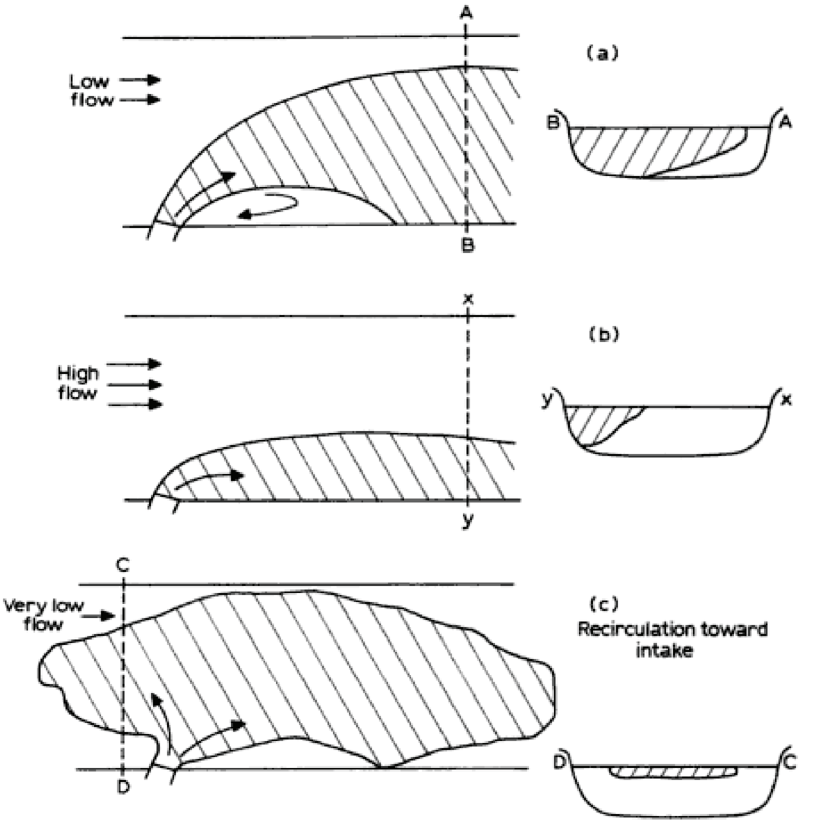


Figure 8. *Thermal plume patterns in a river at different rates of flow (Langford 1990).*

Despite the mixing processes in streams, reach of mixing can extend downstream for many kilometers (Coutant 1999, Prats et al. 2010). If the thermal discharge enters as tributary, the mixing zone often will remain discrete along its respective shoreline until mixing is forced by some topographic feature such as a bend (which imparts a spiral rolling effect that enhances mixing; (Coutant 1999). Water temperatures can thus differ markedly across a long reach of the stream. This effect might be desirable to provide a zone of passage for mobile organisms and restricting the size of any local adverse impact (Coutant 1999). However, there are sometimes several outlets along the whole river width to provide a quick dilution of the discharged water. If water temperatures are increased strongly, this may lead to thermal blockages for fish as documentted for sockeye salmon in the Columbia river (USA) and some tributaries. Often, the avoided temperatures are about 3-4°C above the preferred temperature range of the fish species (Coutant 1999).

As water temperature alterations decrease with increasing distances from the cooling water outfall, three zones are commonly defined (Maderich et al. 2008): (1) the near-field, where the transport is dominated by turbulent entrainment of the incoming buoyant jet, (2) the intermediate field, where the buoyancy force in the plume is dominant, and (3) the far-field, where the cooling water is transported passively by the ambient currents. Often the absolute amout of heat released is small compared to the entire water body and then the far-field (ambient) temperature is not changed much (Langford 1990).

Energy fluxes through the water-atmosphere-interface are the most effective way of heat exchange between a water body and its surroundings. Hence, the simplest method of heat disposal is to discharge an effluent to the receiving water directly, as near the surface as possible (Langford

1990). The main mechanisms of surface heat loss are evaporation, conduction and back radiation (Coutant 1999). (Richter et al. 1979) found evaporation to be the dominant process of heat transfer to the atmosphere. The magnitude of energy loss depends on the temperature of the water surface (e.g., back radiation is proportional to the fourth power of the absolute temperature of the surface(Coutant 1999). The rate of heat loss and mixing with surrounding water differs in individual water bodies. In cooling ponds and shallow lakes, atmospheric losses may be dominant, but in stratified waters and rivers, mixing processes and heat loss to the surrounding water may be more significant (Langford 1990).

Potential ecological effects of any thermal discharge cannot be regarded simply as a direct function of the increased water temperatures (Langford 1990). Characteristics of water are changed when passing a cooling system. For instance, cooling water might get aeration caused by turbulence in the cooling system (Langford 1990) or plankton organisms, damaged by exposure to high temperatures, may increase the phosphorus content (Koschel et al. 1985). If cooling water is not discharged at the place of its origin, discharges lead to a dislocation of water masses of mostly different water characteristics. This is relevant on horizontal levels like water transport from one water body to another (e.g. Koschel et al. 1985) as well as on vertical levels like water transport from the hypolimnion to the water surface (e.g. Neill and Magnuson 1974). Furthermore, there may be additives like biocides (mainly chlorine) in cooling water discharges (Langford 1990).

5.1 Rivers

Water temperature is arguably the most important physical property of streams and rivers (Webb 1996). It moderates many different aspects of stream and river biota (Webb 1996). Hence, the thermal regime of rivers plays an important role in the overall health of aquatic ecosystems, including water quality issues, the distribution of aquatic species within the river environment and stream productivity (Caissie 2006). Stream temperature also controls the rates of many biotic and abiotic processes (Johnson and Jones 2000). Surprisingly, in view of the vast amounts of literature dealing with thermal discharges, very few large-scale mortalities have occurred which can be unequivocally related to high temperature in any effluent (Langford 1990). (Maderich et al. 2008) confirm this finding by their conclusion, that cooling water discharges in the large rivers were not critical in terms of environmental impact. However, the recent coincidence of low volumetric flow rates and relatively high water temperatures in the warm summers of 2003 and 2004 in Europe demonstrated the limitations of once-through cooling (Maderich et al. 2008).

(Levin et al. 1972) reviewed various recent studies on ecological effects caused by thermal discharges (discharge increased up to 13°C) of power plants in the USA. They conclude that, with a few exceptions, there has not been any major damage to the aquatic environment from the heated effluents of existing power plants at this time. However, they report some ecological changes. Since the Connecticut Yankee Power Plant (Connecticut) started operation, the white and brown bullhead catfishes undergo a marked weight loss (average of 20%) in the warm water of the effluent canal (temperature increase by 11°C) despite a constant availability of food (Levin et al. 1972). At the Chalk Point fossil-fueled Steam Generating Plant on the Patuxent River (Maryand) cooling water got warmed 12.7°C during winter and 6.4°C during summer (Levin et al. 1972). Comparing biota in the intake and effluent channel showed a nearly three times as great average production in the effluent channel (Levin et al. 1972). Furthermore, in the warmest months of the year the number of epifaunal species decreased and a few species even disappeared (Levin et al. 1972).

Temperatures in the Ebro River in Spain are influenced by water release of hypolimnetic water by three reservoirs (Mequinensa, Riba-roja and Flix) and the use of water for cooling at the 5 km downstream Ascó Nuclear Power Plant (Prats et al. 2010). A concession of 72.3 m^3s^{-1} of water is granted to the power plant for cooling (minimum river flows in the area are around 100 m m^3s^{-1}) (Prats et al. 2010). The warm water discharge forms a thermal plume which floats on the surface until it gets mixed completely some kilometers downstream (Prats et al. 2010). Just below the nuclear power plant, the water temperature increases by approximately 3C° on average year round due to the effluent discharge (Prats et al. 2010). The actual value depends greatly on discharge (Figure 9): High temperature increase was observed in times of low river discharges, while temperature alterations were moderate during high discharges (Prats et al. 2010). The effects of the reservoirs and the nuclear power plant were the usual for this kind of structures and could be detected many kilometers downstream (Prats et al. 2010). In the summer, the cooling effect of the reservoirs and the warming effect of the nuclear power plant compensated each other, while in winter, the warming effect of both summed up (Prats et al. 2010). (Prats et al. 2010) also reviewed previous studies on the thermal pollution by the Ascó Nuclear Power Station during summer. Despite the warming of 2-4°C, no negative effects on the aquatic life were observed. However, it should be considered that summer temperatures of the Ebro River at the power plant are lowered due to the upstream reservoirs. Without this cooling effect, maximum temperature limits of some species could potentially have been exceeded. As in many water bodies, a general temperature increase was observed in the Ebro River (upstream the reservoirs and the nuclear power plant). Between 1955 and 2000, an increase of 2.3°C in the mean annual water temperature could be demonstrated, making a temperature increase of $0.05°Cyr^{-1}$ (Prats et al. 2010). The increase could be related to a decrease in discharge in the same period and an increase in air temperature starting in the 1970s (Prats et al. 2010).

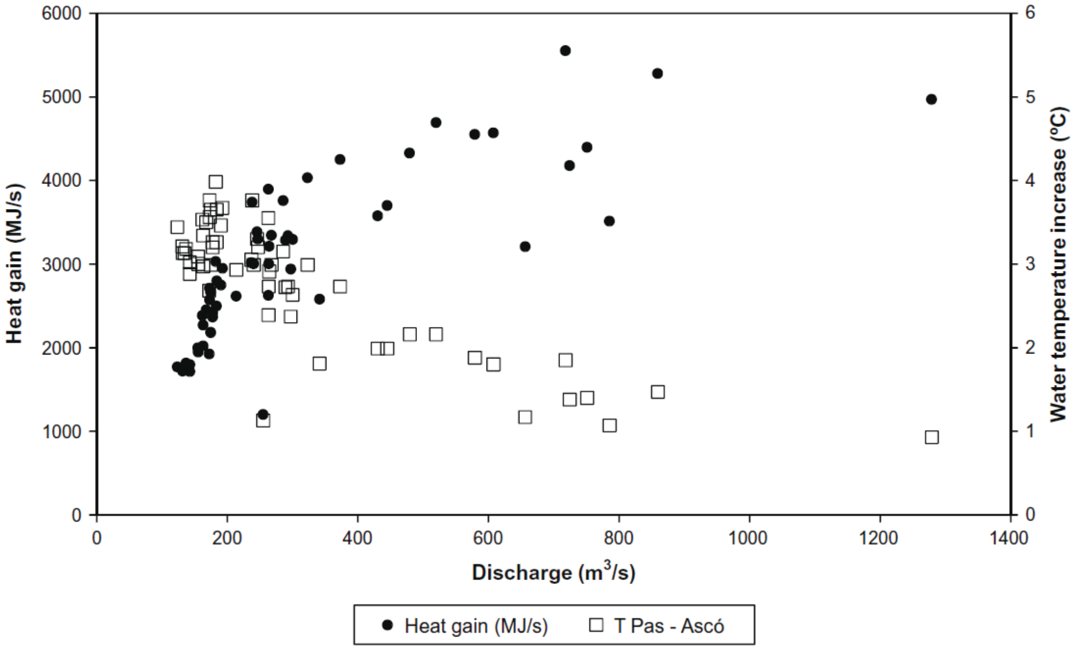


Figure 9. Heat gain (●) and alteration of the monthly mean water temperature of the Ebro River (Spain) caused by the Ascó Nuclear Power Plant (□) as a function of monthly mean discharge (data from May 1998 to March 2004) (Prats et al. 2010).

From October 1977 to September 1978, (Sadler 1980) studied the effects of the cooling water discharges of the Castle Donington Power Station on fish in the Trent River (Leicestershire, UK). The station operated for 15-19 hours per day and so temperatures downstream of the power station return to ambient for several hours each night (Sadler 1980). Nevertheless, monthly water temperatures (measured between 10 and 12 o'clock) were strongly increased during operation (**Figure 10**). The mean increase during the measurement period was 7°C, with monthly maximum and minimum temperature increases of 12°C and 4°C, respectively (Sadler 1980).

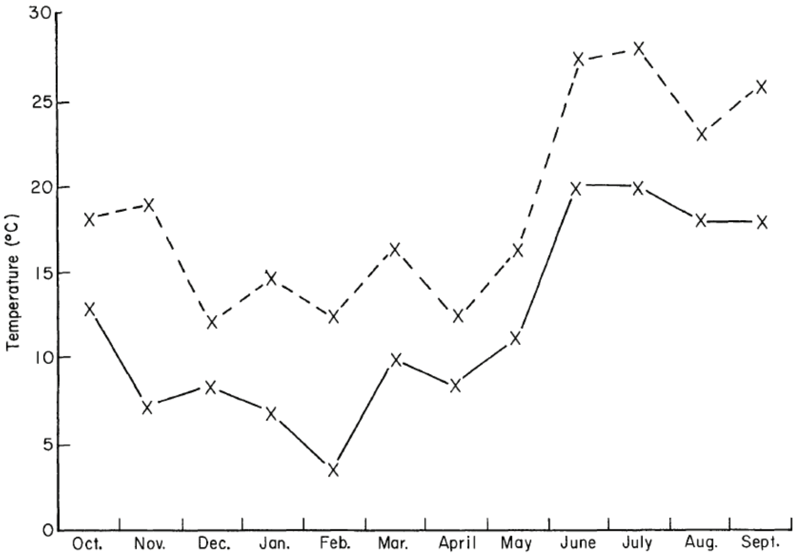


Figure 10. Water temperature measured between 10 and 12 o'clock of the River Trent downstream (x - - - x) and upstream (x——x) of Castle Donington Power Station October 1977 – September 1978 (Sadler 1980).

(Sadler 1980) compared fish occurrence of catches by seine netting up- and downriver of the Castle Donington Power Plant. The downstream sites showed consistently greater abundances of fish (**Figure 11**) and greater species diversity (**Figure 12**) than either of the upstream site. Average densities of 0.2 fish m^{-2} upstream and 0.6 fish m^{-2} downstream of the effluent were estimated for the age group of fish sampled (Sadler 1980). The maximum stream temperatures downstream reached 28°C at which temperature there was little evidence of fish avoiding the heated discharge (Sadler 1980).

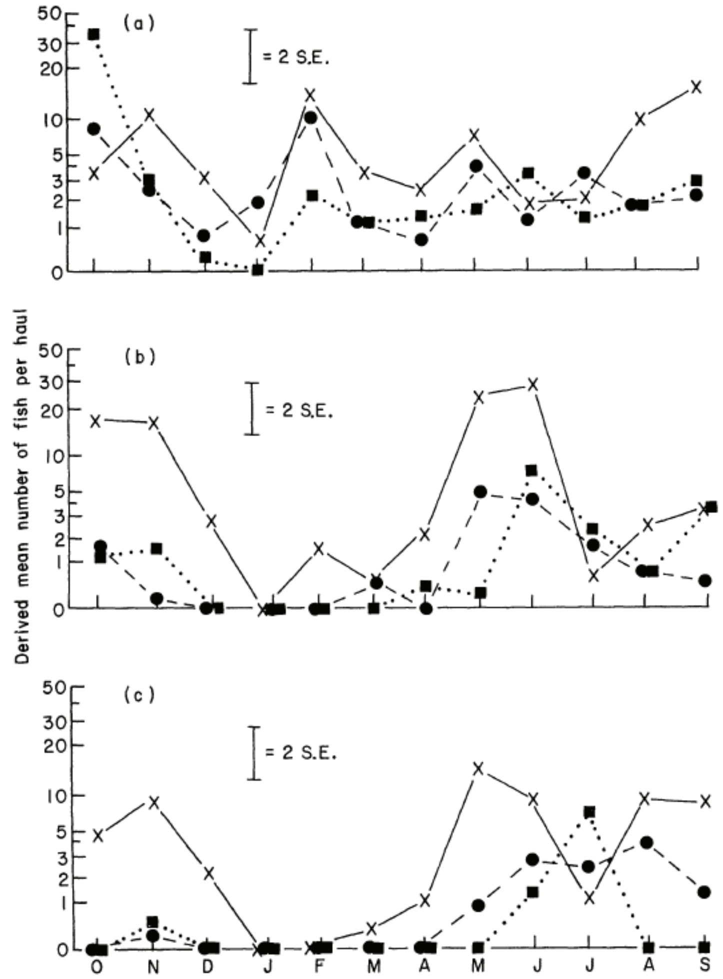


Figure 11. Abundance of (a) bleak, (b) gudgeon and (c) roach downstream (x——x) and at two upstream sites (●- - -● and ▪ · · · ▪) of Castle Donington Power Plant (October 1977 – September 1978) (Sadler 1980).

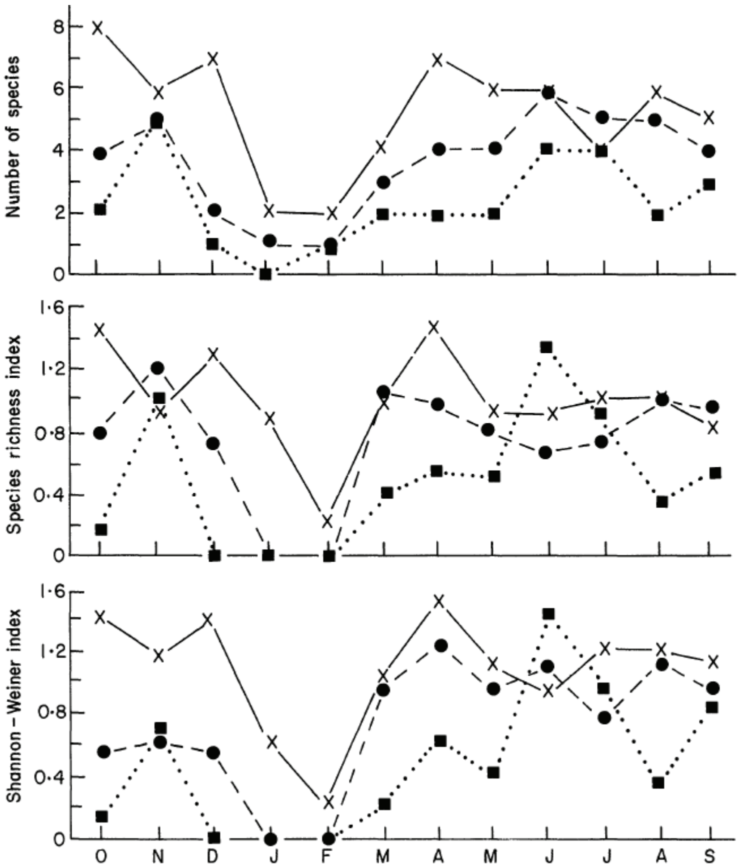


Figure 12. Diversity of fish population downstream (x——x) and at two upstream sites (●- - -● and ▪ · · · ▪) of Castle Donington Power Plant (October 1977 – September 1978) (Sadler 1980).

5.2 Lakes

For cooling water supply in power plants, lake water is far less used than river water. Therefore, there are not many studies about the effects of thermal discharges on aquatic ecology in lakes. However, there is one well-studied case, which is Lake Stechlin and the nearby Rheinsberg Nuclear Power Plant in northeastern Germany. This example on thermal pollution on a lake will be explicated in detail. Generally, the occurrence of problems due to thermal discharges in standing waters is more likely than in rivers. Since, in contrast to rivers, deeper lakes are mostly thermally stratified, the effects on temperature can be minimized by cooling water abstraction from below the thermocline and its release at the surface (Langford 1990). While this measure is capable to reduce the temperature difference between discharged and ambient water, it might cause other unwanted effects. The hypolimnetic water will usually be richer in nutrients or contaminant than the surface water, and in smaller lakes, there may be a breakdown of the normal stratification regime (Langford 1990).

At the Biscayne Bay in Florida (USA) the heated effluents of two fossil-fueled units and two nuclear power plants lead to the reduction of the diversity and abundance of algae and animals in small areas adjacent to the mouth of the effluent canal (Levin et al. 1972). In an area of 0.5 km^2, where

temperatures have risen 4°C, many plants and animals have been killed or greatly reduced in number (Levin et al. 1972). In a second zone of about 0.7 km^2 where water temperature increased by 3°C, algae have been damaged, and species diversity and abundance have been reduced. In the latter area, mollusks and crustaceans increased somewhat, but the number of fishes decreased (Levin et al. 1972).
Lake Monona is located in Wisconsin (USA) and has a surface area of 14 km^2 and a maximum depth of 22.5 m (Brauer et al. 1974, Neill and Magnuson 1974). Lake water is used as cooling water in the Blount Street fossil fuel Power Station, where it circulates through the stream condensers and gets heated up by 10°C (Brauer et al. 1974, Neill and Magnuson 1974). Water is pumped from 5 m beneath the lakes surface about 110 m offshore (Neill and Magnuson 1974) and discharged at two outfalls into the littoral zone, 200 m apart from the intake (approximately $5.9 \times l0^8$ l of water per day) (Brauer et al. 1974). The time for water to pass through each circuit is about 10 min (Brauer et al. 1974). Maximum temperatures in the effluent-outfall area approach 35°C in summer and 14°C in winter wile unheated parts of the littoral zone rarely exceeds 29°C (Neill and Magnuson 1974). Although the outfall area gets warmed by several degrees Celsius, temperatures in the greatest part of the lake are unaltered. Only about 0.3% of the water at a depth of 0.5 m is warmed more than 1°C, the percentage of total volume so warmed is much smaller (Neill and Magnuson 1974). The normally zooplankton-poor littoral zone was enriched by the discharge of water containing limnetic zooplankton that had been pumped through the power plant’s cooling system (Neill and Magnuson 1974). (Brauer et al. 1974) found the zooplankton density at the limnetic intakes to be consistently higher than in littoral reference areas, leading to 2-7 times higher densities of zooplankton near the outfalls compared to littoral reference stations. The zooplankton transported to the littoral zone increased the supply of food for fishes in the outfall (Brauer et al. 1974). Distributions of fishes within the littoral zone of Lake Monona were markedly and differentially influenced by the discharge of heated effluent (Neill and Magnuson 1974). Several fish species concentrated regularly or occasionally in the outfall area, others regularly avoided the warm water plume and some species were evenly distributed (Neill and Magnuson 1974). Concentration in or avoidance of the outfall was also dependent on season and day/night (Neill and Magnuson 1974). Temperature during summer was a major factor governing fish distribution, fish tended to be most abundant in that part of the habitat having temperatures within or nearest a species-specific preferred range of temperature determined in the laboratory (Neill and Magnuson 1974). (Neill and Magnuson 1974) also measured body temperature of fish in the area close to the warm water discharge. They found body temperatures of most fishes tended to increase with water temperature. However, fish temperatures tended to vary less than water temperatures, body temperatures of some species were virtually independent of capture temperatures (Neill and Magnuson 1974). Fish seemed to move between different water temperatures, managing to keep their body temperature in the range of their preferred temperature.

5.2.1 Lake Stechlin

Lake Stechlin is a lake in Brandenburg, north eastern Germany. Between 1966 and 1989, the lake was highly thermally polluted by a nearby nuclear power plant. This anthropogenic manipulation is often called an ecological long-term experiment (Koschel 1995, Koschel et al. 2002, Koschel and Adams

2003). The lake has been studied before, during and after power plant operation. Hence, Lake Stechlin is probably one of the best studied examples of thermal pollution on lakes. However, beside thermal pollution, there were other important factors like the induced water circulation and nutrient input by cooling water discharge (Koschel 1995). Released radioactivity by the nuclear power plant was found to be very small, it did not affect neither aquatic nor terrestrial life of the Lake Stechlin area (Koschel et al. 1985). The various effects (physical, chemical and biological) of the cooling water circulation system on Lake Stechlin are summarized in **Figure 13**.

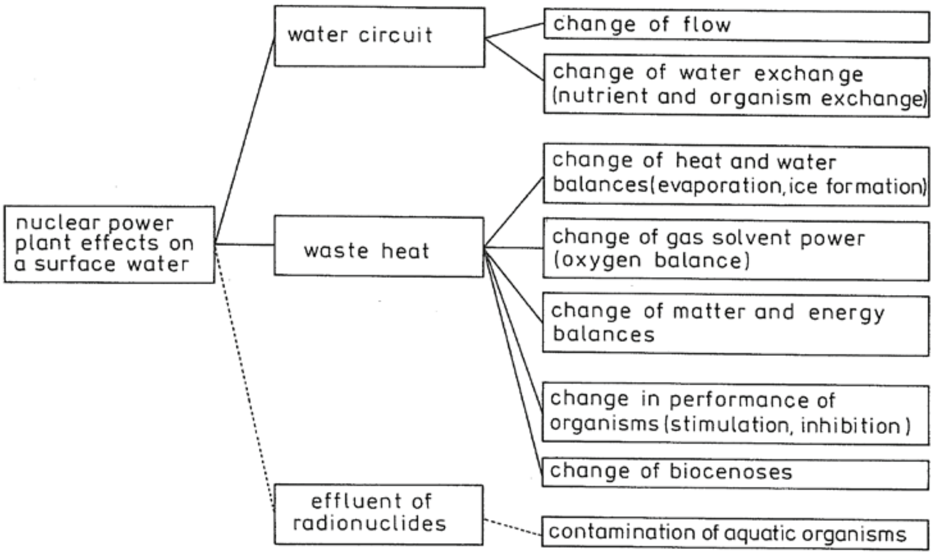


Figure 13. Summarized effects of different phases of cooling water circulation on the productivity in the Lake Stechlin system (Koschel et al. 1985).

Lake Stechlin is a typical oligotrophic and dimictic (in some years warm monomictic) lowland lake of the Northern (Baltic) Land Ridge (Koschel et al. 2002). As its surrounding lakes, Lake Stechlin is a dead ice lake without natural surface runoff (Krey 1985). The lake is fed, apart from precipitation, by ground water only (Heitmann and Schubert 1965). Lake Stechlin has a surface area of 4.25 km^2, an average and maximal depth of about 23 and 68 m, respectively, and a volume of about 97.5 × 10^6 m^3 (Richter et al. 1979, Krey 1985, Koschel and Adams 2003).

From 1966 to 1989 the nuclear power plant Rheinsberg was operated in the Lake Stechlin area (Koschel et al. 2002). Water was taken from neighboring mesotrophic Lake Nehmitz and the heated cooling water was pumped into Lake Stechlin (Koschel 1995). From Lake Stechlin, water flowed back to Lake Nehmitz trough a 1.5 km long channel which was built in 1959 (Weiler et al. 2003). The two channels, the southern part of Lake Stechlin and the northern part of Lake Nehmitz formed a cooling water circulation system (**Figure 14**). After starting power plant operation, 300’000 m^3 cooling water (according to (Casper et al. 1985a) even 400’000 m^3) were discharged daily into Lake Stechlin (Koschel 1995). The warmed cooling water reached Lake Stechlin with an increased temperature by 10°C (Richter et al. 1979, Richter and Koschel 1985).

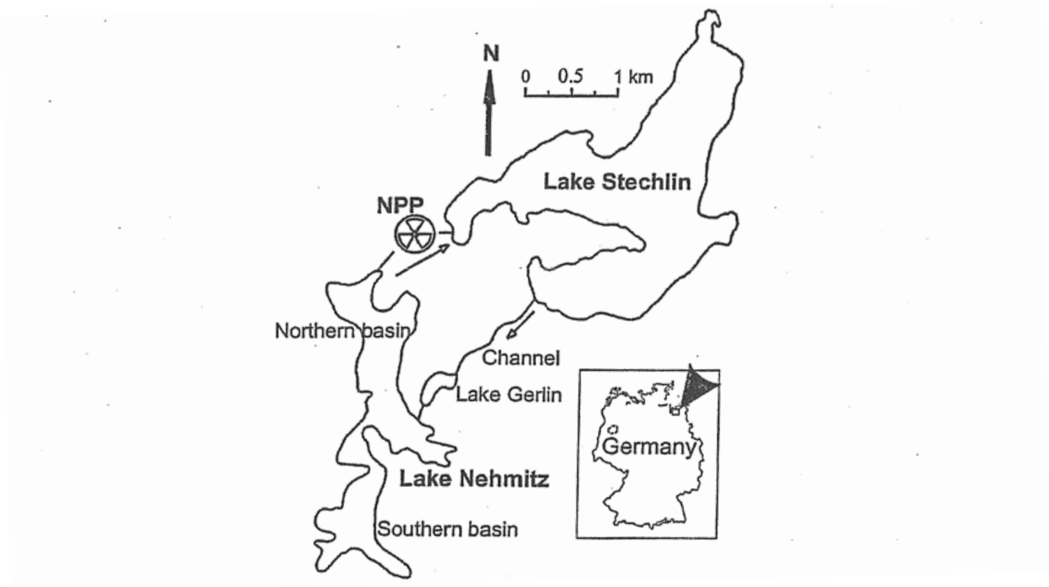


Figure 14. *Lake Stechlin, Lake Nehmitz, the Rheinsberg Nuclear Power Plant (NPP) and the flow direction of the cooling water circuit (indicated by arrows; (Schulz 2004).*

The water temperature has been higher throughout the year after power plant operation started, with particularly remarkable increases in January/February and July/August (Richter and Koschel 1985). According to (Koschel et al. 2002) (based on various publications), the mean temperature increase during power plant operation was 1°C for the whole lake (yearly average) and 2°C for the epilimnion during July/August. The mean temperature in the hypolimnion increased by 0.8°C (Richter et al. 1979). Due to the low density of the warm cooling water, the water layer at the very surface was most affected by the discharged water (**Figure 15**). At the surface, mean temperature increase fluctuated between 3°C in spring and 1°C in autumn (Richter et al. 1979). The distribution of temperature increase at the surface was dependent on the current wind conditions (**Figure 16**). During the warm time of the year, the pronounced stratification in Lake Stechlin disconnected the epilimnion from the effects of thermal discharge. Hence, temperature increase in the epilimnion is much higher, temperature alterations in the hypolimnion can be ascribed to turnover processes during cold times of the year (**Figure 17**). While the temperature increase due to the thermal discharge is high in summer at the surface, temperatures at 20 m depth are most affected in winter and spring (**Figure 18**). These measurements show that the amplitude of temperature alterations occur temporally shifted at different water layers.

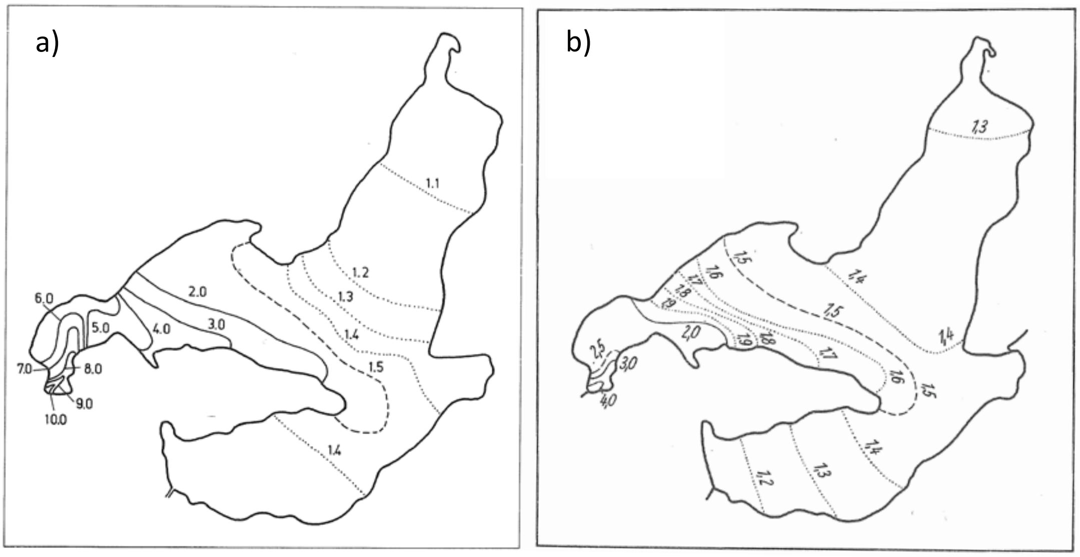


Figure 15. Mean increase of Lake Stechlins surface water temperature a) (Richter and Koschel 1985) and temperature increase in 2 m depth b) (Richter et al. 1979). Temperature increase in the west bay is higher at the surface than in 2 m depth, with rising differences toward the effluent. For the rest of the lake area, temperature alterations are about the same.

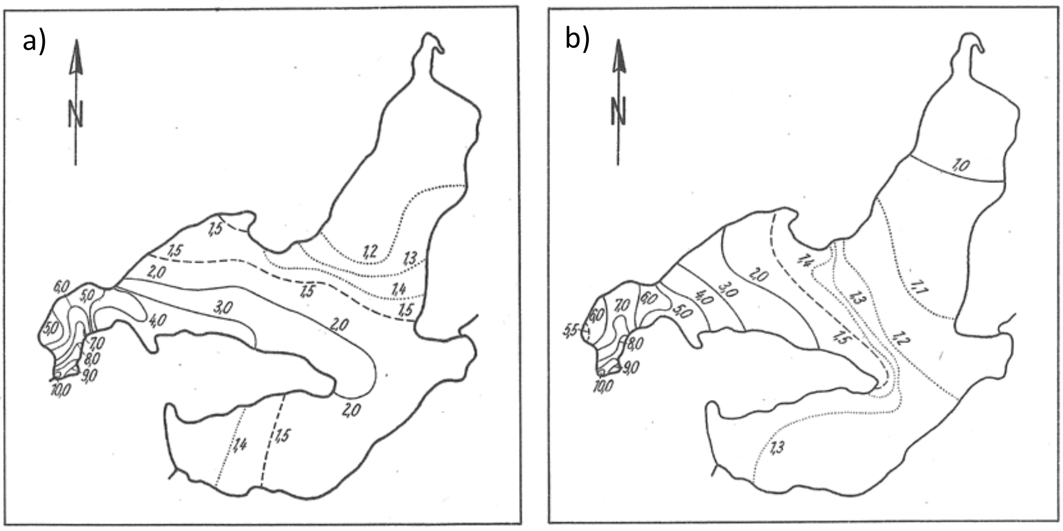


Figure 16. Mean temperature increase at the surface of Lake Stechlin after starting power plant operation with wind from the west a) and wind form the east b) (Richter et al. 1979).

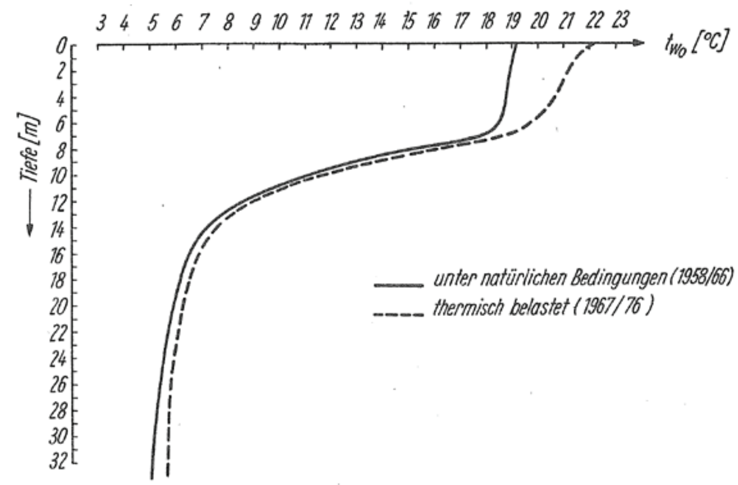


Figure 17. Mean temperature profile at August 1 under thermal pollution (1967-1976, - - -) and uninfluenced conditions (1958-1966, ——). Measurements were taken in the west part of Lake Stechlin, about in the middle between effluent and the center of the lake. At this time of the year, when lake stratification is at its maximum, epilimnion is much more affected by the thermal discharge than the deeper layers of the lake (Richter et al. 1979).

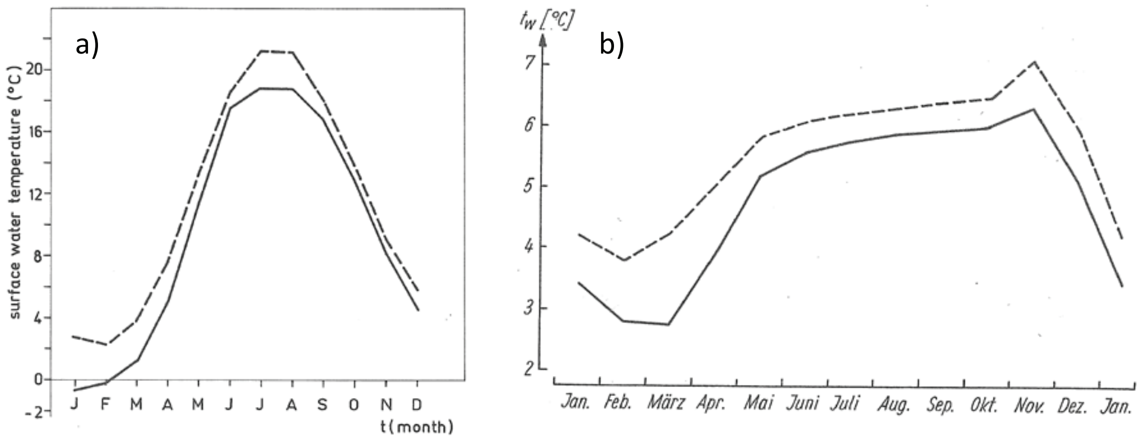


Figure 18. Mean annual course of Lake Stechlins water temperature at the surface a) (Richter et al. 1979) and in 20 m depth b) (Richter and Koschel 1985). Measurements were taken before the nuclear power plant came in operation (1958-1966, ——) and during the cooling water circulation (1967-1976, - - -).

The greatest part of the heat energy transported to Lake Stechlin by cooling water discharge was transferred to the atmosphere (Koschel et al. 1985). According to (Koschel et al. 1985), the lakes evaporation and outgoing long-wave radiation were considerably increased during power plant operation. 57% of the additional energy input was transferred to the atmosphere by evaporation, 31% by radiative heat exchange and 12% by sensible heat flux (Richter et al. 1979). The strongly increased evaporation had effects on the water budget of the lake (**Figure 19a**). Yearly evaporation after starting power plant operation increased by 27% (from 663 mm to 843 mm) (Richter and Koschel 1985). The greatest increase in evaporation occurred in summer months (**Figure 19b**), according to the peak increase in surface water temperature. Because of this substantial increases of heat loss to the atmosphere, only about 5% of the heat energy released in Lake Stechlin reached Lake Nehmitz (Richter et al. 1979).

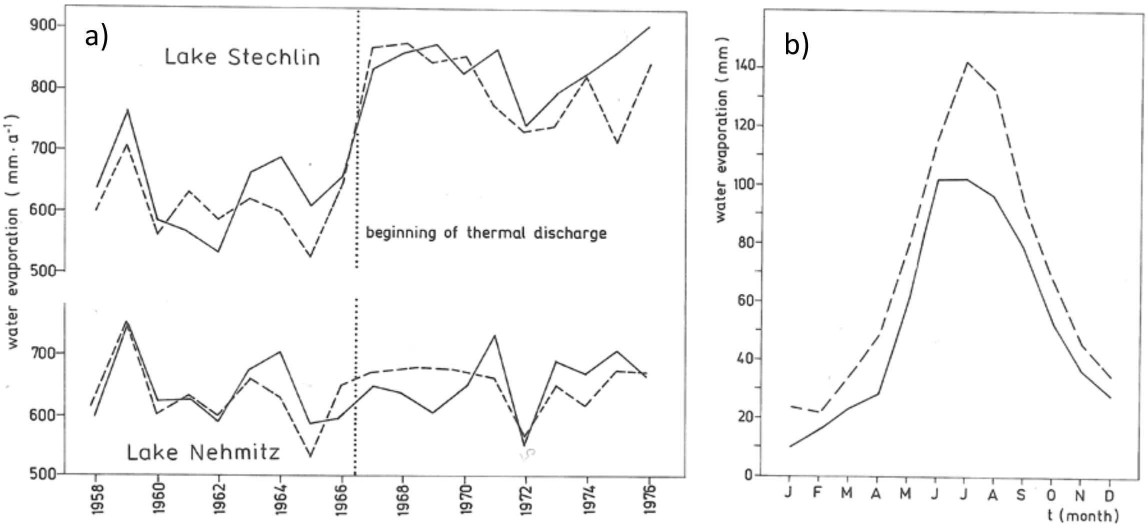


Figure 19. Increased evaporation due to the cooling water circulation of the nuclear power plant. a) shows measured (——) and by the heat balance method calculated (- - -) evaporation values in Lake Stechlin and neighboring Lake Nehmitz. The dotted line marks the starting point of power plant operation. While the effects on Lake Stechlins evaporation are considerable, Lake Nehmitz does not seem to be affected. b) shows the amount of evaporation in Lake Stechlin before (——) and during (- - -) power plant operation (Richter and Koschel 1985).

The highest increase of total energy load of the whole water mass as well as the maximum temperature increase in deep water layers occurred in winter months due to mixing processes (Richter et al. 1979, Koschel et al. 1985). In summer on the other hand, mainly the epilimnion (1/4 of the whole water mass) was warmed (Richter et al. 1979). Hence in summer, heat was transferred more rapid to the atmosphere and less energy remained in the lake (Richter et al. 1979). The autumn full circulation period was lengthened and passed directly into the spring full circulation period, a real winter stagnation phase did not set in anymore (Casper et al. 1985b, Koschel et al. 1985). Hence, during power plant operation, Lake Stechlin showed a distinct trend to monomixis (Koschel et al. 1985). Consequently, during winter months, the duration and thickness of ice cover got reduced and the lake only froze partially (Koschel et al. 1985). Only in very cold winters when, in spite of incoming cooling water, a complete ice cover was formed, stratification approached the natural conditions (Koschel et al. 1985). While in winter months the stratification of Lake Stechlin got weakened or oppressed, stratification got more pronounced in summer. The temperature difference between epi- and hypolimnion got increased, the temperature gradient of the thermocline increased from 1.5-2°Cm^{-1} to 2-2.5°Cm^{-1} (Richter et al. 1979). The thickness of the thermocline got compressed and decreased by 0.5 m (Koschel et al. 1985, Koschel et al. 2002).

Oxygen conditions in Lake Stechlin deteriorated after the cooling-water circulation system of the nuclear power plant began to function (Mothes et al. 1985). The critical time period for oxygen saturation was at the end of the summer stagnation period in the lower horizons of the hypolimnion (Mothes et al. 1985). Especially the west and south basin was affected, while there was not much impact on the north basin (Mothes et al. 1985). (Mothes et al. 1985) found a decrease of oxygen saturations from 70% to 53% in the west basin and from 60% to 44% in the south basin in November at a depth of around 30 m. A decrease of oxygen was also seen at depths of 60 m (Mothes et al. 1985). (Koschel et al. 2002) compared the oxygen concentrations at the same depth during and after power plant operation. Surprisingly, they did not find an increase in minimum oxygen concentration after shutting down power plant operation 1989, but a further small decrease. Changing climate could be a possible explanation for these findings, while sediment interactions seem to not affect oxygen, as (Benndorf et al. 1985) found the sediment of Lake Stechlin not to exert any significant effect on the pelagic zone regarding oxygen consumption and nutrient release. According to (Mothes et al. 1985), the oxygen deterioration was caused by the increased nutrient supply by water transport from mesotrophic Lake Nehmitz to the oligotrophic Lake Stechlin, and not by the waste heat of the cooling water. The waste heat rather counteracts any deterioration by reducing or totally preventing any winter ice cover and extending the time period of full circulation, leading to a maximum oxygen saturation of the water body (Mothes et al. 1985). (Babenzien and Babenzien 1985) found a decrease in oxygen contents in the thermal effluent because of the lower saturation concentration at increasing temperatures (water temperature increased by 18°C in the condenser). While this effect was without any consequences for the well oxygenated Lake Stechlin, it might be very critical in rivers (Babenzien and Babenzien 1985).

Lake Stechlin is known for its clear water and therefore it is a popular spot for scuba-diving. The thermal discharge of the nuclear power plant affected the lakes visibility. Before power plant operations (1958-1965), (Richter and Koschel 1985) found the Secchi depth to be generally higher (on average 2 m) compared to the time of operations (1958-1982). The periodical oscillation however was found to be about the same. The reduction of visibility was caused by the increase of plankton and periphyton due to higher water temperatures and nutrient inflow from Lake Nehmitz (Casper et al. 1985b).

Changes in the phosphorus load in Lake Stechlin are of great importance, as it is the primary nutrient limiting the biomass production in Lake Stechlin and most neighbouring lakes (Mothes et al. 1985). During nuclear power plant operation (sampled time period 1970-1989), phosphorus increased and after power plant operation (1990-2000) declined significantly (Koschel et al. 2002). At 60 m depth, a delayed reaction was observed (Koschel et al. 2002). The phosphorus concentration progressively increased in the deep hypolimnetic water from the 1970s to the 1990s, to reach its maximum (about four times higher concentrations than in the 1970s) in the late 1990s, about 10 years after power plant operation was stopped (Koschel et al. 2002). In oligotrophic lakes, sediments possess the ability to retain a major proportion of phosphorus input (Babenzien and Babenzien 1985). From the sediment, phosphorus can be released primarily by bacterial metabolism (Babenzien and Babenzien 1985). In Lake Stechlin, the microbial activity is very low (Babenzien and Babenzien 1985), preventing the nutrient outflow from the sediment and making it to a phosphorus-sink. During power plant operations, a great amount of total phosphorus input into Lake Stechlin originated from the incoming cooling water of mesotrophic Lake Nahmitz (Koschel et al. 1985). Orthophosphate, released from the plankton organisms damaged by the shock-like increase in water temperature of up to ~28°C passing the nuclear power plant, is another source of phosphorus, especially during summer (Koschel et al. 1985). Orthophosphate input due to heating effect amounts to 22% of the whole orthophosphate input into Lake Stechlin (Koschel et al. 1985).

During the power plant operation, the quantity and seasonal periodicity of primary production were strongly influenced by the discontinued operation of the cooling water circulation of the nuclear power plant (Koschel et al. 2002). Due to higher water temperatures, lengthening of the vegetation period and additional supply of nutritious matter, the rate of annual primary production got dangerously high (Koschel et al. 1985). Primary production showed remarkable rates as early as February and March (Koschel and Scheffler 1985). After this peak in the early year, primary production decreased until the highest rates were observed in summer (Koschel and Scheffler 1985). In years with long ice cover or a long phase without the circulation of thermally loaded cooling water, the primary production peak in February/March was not observed, which equals pre power plant conditions (Koschel and Scheffler 1985). The increase phytoplankton density was reflected in the increased turbidity of the water from 1970 to 1994 (Koschel 1995). Under natural conditions, fluctuations in the composition of the phytoplankton community correspond to the seasonal alteration of meteorological factors (Koschel and Scheffler 1985). During the period of thermal discharges however, the cooling water circulation was the dominant environmental factor (Koschel et al. 1985). When the power plant came to operation, the sudden temperature increase caused a shock like reaction of the phytoplankton community, throwing the organismic society into disorder (Koschel et al. 1985). There was a distinct shifting of the previous structure and a distinct, higher level of biomass production (Koschel et al. 1985). The development of microplanktonic diatoms (*Fragilaria crotonensis)* and nano- and ultraplanktonic green algae, especially *phytoflagellates* and *coccoid* forms, were favored (Koschel et al. 1985). Later however, a new biological equilibrium was established which, considering the composition of the phytoplankton community, partially approached the previously existing equilibrium and preserved its diatom-chrysomonaden dominance (Koschel et al. 1985). In the end, the increased phosphorus load and thermal pollution resulted more in moderately higher primary production of phytoplankton than in important changes in the population structure of phytoplankton (Koschel et al. 2002). According to (Benndorf et al. 1985), indirect effects such as feedback control of phytoplankton by zooplankton (and zooplankton by fish) predominate over direct effects of cooling water discharges on phytoplankton growth.

In areas, which were highly influenced by the warm water discharge, the start of periphyton growth began as early as January instead of April (uninfluenced conditions) (Casper et al. 1985b). The annual climax of periphyton abundance was reached in May/June instead of August/September (Casper et al. 1985b). Macrophytes showed a similar reaction to temperature alterations as phytoplankton. First reaction was seen in *Characeae* communities, of which nearly all receded (Casper et al. 1985b). In many places desolated areas without macrophytic vegetation arose (Casper et al. 1985b). Most sensitive macrtophyte species disappeared, while some, thriving in nutrient-richer water, increased suddenly (Casper et al. 1985b). In course of time, certain stabilization took place: The displacement of the zonal vegetation seemed to have ended, species which had disappeared returned while other new invading species disappeared again (Casper et al. 1985b). (Weiler et al. 2003) collected crustacean zooplankton by a plankton net which was hauled from the bottom to the surface at the deepest point of the lake. They did not find any change in species composition during or after power plant operations, the 14 species remained and no further species were added. The natural competition between the different algal groups was disturbed by the cooling water circulation (Casper et al. 1985b). The dominance alternated from year to year, depending upon the intensity and regularity of the cooling water circulation (Casper et al. 1985b).

The effects of the temperature alteration on macrozoobenthos were pronounced. The total productivity in the depths increased by about 300% (Koschel et al. 1985), while the number of species in the lake increased from 408 to 502 because of the higher nutrient content (Flössner et al. 1985a). In the west bay, where most of the additional nutrient input was deposited, a totally new population structure developed; the former sub-communities were no longer recognizable (Flössner et al. 1985a). The community of the more profundal zone of Lake Stechlin as a whole, however, did not change (Flössner et al. 1985a). In 1965, when there was a trial runoff the cooling water circulation without heat, an increase of macrozoobenthos abundance was observed (Koschel et al.

1985). These findings confirm the importance of increased nutrient supply on changes of macrozoobenthos.
Surprisingly, there are barely any studies about the effects of the warm water discharge on fish in Lake Stechlin. However, there were some observations on European Cisco in the lake. The yield of European Cisco decreased strongly between 1947 and 1982 from 8 to 1 kg ha^{-1} per year (Flössner et al. 1985b). The absence of any research in this field does not allow the formation of a definitive judgment on the factors responsible for the decrease of the yield though (Flössner et al. 1985b). Due to the cooling water circulation, the transparency in the lake declined and the lower limit of chara-meadows (*Characeae*) growth raised from 20 m to 12-14 m (Flössner et al. 1985b). These changes resulted in a loss of spawning sites, especially in the west of Lake Stechlin, where essential spawning sites of the European Cisco are (Flössner et al. 1985b). Furthermore, through the thermal inflow into the west bay, additional plankton and detritus precipitation prevented the eggs of European Cisco from developing (Flössner et al. 1985b). Beside the loss of spawning sites, overfishing may also have contributed to the decrease of European Cisco (Flössner et al. 1985b).

Many effects in Lake Stechlin caused by the cooling water circulation showed clear tendencies towards eutrophication (Figure 20), e.g. the large increase in water temperature, the increase in nutrient load, the increase in bioproduction and the observed changes of bioceonoses (Koschel et al. 1985, Koschel 1995, Koschel et al. 2002, Koschel and Adams 2003). With the cooling water circulation in operation, the phosphorus load of Lake Stechlin attained values which were typical for mesotrophic conditions (Richter and Koschel 1985). However, there were several factors working against eutrophication. The changes in stability of the lakes stratification reduced the nutrient and organic matter load of the hypolimnion (Koschel 1995) and transported great amounts of oxygen into the hypolimnion (Koschel et al. 1985). After starting power plant operation, photosynthetically-induced calcite precipitation increased in Lake Stechlin (Benndorf et al. 1985, Koschel et al. 2002, Koschel and Adams 2003). The increase of calcite precipitation provided a high autochthonous self-protection potential by co-precipitation of phosphorus, a rise in sedimentation velocity and a decrease of phosphorus-release from the sediments (Koschel et al. 2002). So the authors conclude that, despite the long lasting heat inflow by the nuclear power plant, Lake Stechlin kept its oligotrophic state (Koschel et al. 1985, Koschel et al. 2002, Padisak et al. 2003).

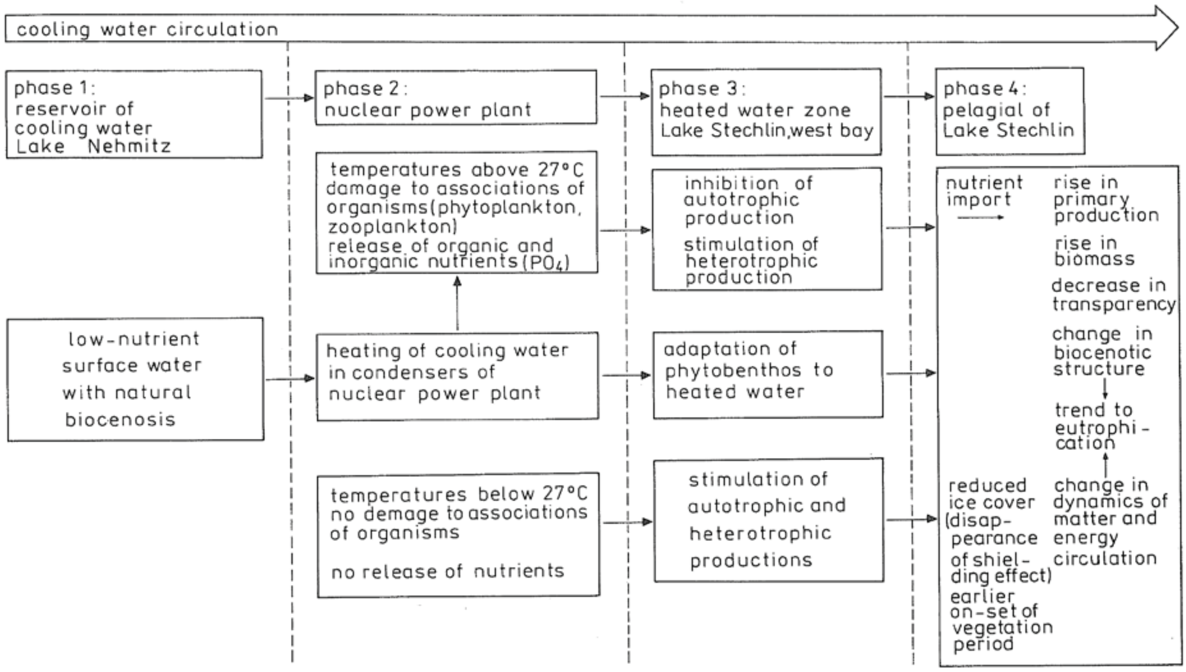


Figure 20. Impact of the Rheinsberg Nuclear Power Plant with its external cooling water circulation on Lake Stechlin. Tendency is of most processes is towards eutrophication of the lake (Koschel et al. 1985).

Effects of the warm water pollution on biology in Lake Stechlin depended highly on season. There were reverse effects observed in the cold and warm period of the year, e.g. the increase and decrease of primary production during winter and summer months, respectively. (Koschel et al. 1985) summarized the effects of the thermal pollution in Lake Stechlin on biocoenoses and lake metabolism as follows:

During the cold period of the year (<10°C):

- No damage to the organisms.
- Heightened productivity of periphyton and phytoplankton compared with the unheated lake region with roughly the same concentration of nutritious matter.
- Increasing vitality of phytobenthos, e.g., premature development of macrophytes. Water lilies (*Nymphaea alba*) already blossomed at the end of April instead at beginning June.
- Increasing vitality of macrozoobenthos, e.g., premature emergence of insects. Insects emerge regularly in winter. Without warm water influence, there is no emergence in winter.
- Heightened activity of bacteria.

During the warm months of the year (>15°C) with surface water temperatures up to 32°C:

- Damage to phyto- and zooplankton.
- Greatly decreased productivity of periphyton and phytoplankton.
- Considerable increase of activity of heterotrophic bacteria coenoses
- Release of nutritious matter fallowing damage to organisms. The concentration of orthophosphate can multiply considerably in the heated water region.

6 Reactions of aquatic biota on temperature

Very high or very low water temperature leads to mortality of aquatic organisms. Less extreme temperatures influence the physiology of organisms (e.g. survival, metabolism, growth, life-cycles). Furthermore, as mobile organisms react actively to temperature conditions, temperature is a key factor for behavior and distribution. While the temperature tolerances of single species, especially fish, are quite well known (e.g. Jobling 1981, Cincotta and Stauffer 1984, Coutant et al. 1999), the reaction of a whole ecosystem to temperature alteration is difficult to predict. In a water body and its ecosystem, internal autocatalytic or autoinhibited processes might occur (Koschel and Adams 2003). Hence, the enormous complexity of ecosystems is one of the obstacles in the causal analysis of such systems (Benndorf et al. 1985).

There are several possible classifications of temperature effects on organisms. (Fry 1967) distinguishes between the fallowing effects:

- Lethal effects (high or low temperatures which are deadly for an organism within a finite time)
- Controlling effects (sub-lethal effects which affect physiological or biochemical processes of organisms, such as growth, metabolic rate or reproduction)
- Directive effects (behavioral responses, movements or migration)

In this work, the effects are subdivided according to described classification. One chapter covers the lethal and the controlling effects, while the subsequent chapter explicates the directive effects.

6.1 Lethal limits and physiological reactions

Most animals and plants survive over a genetically predetermined temperature range (Langford 1990). This range can be modified by many factors, e.g. the acclimation temperature of the organism (Coutant 1999), but for any species the ultimate upper and lower lethal temperatures vary little from those determined genetically (Langford 1990). Of aquatic organisms, plants do have generally higher upper temperature limits than fish and other invertebrates, and Prokaryotic micro-organisms survive higher temperatures than plants (Langford 1990). Bacteria are generally regarded as the organisms most tolerant to temperature, with most forms accepting up to 85°C (Langford 1990). Hence, as a broad generalization, temperature tolerance decreases with physiological and morphological complexity among poikilotherms (Langford 1990). Based on this statement, protection of fish generally results in the protection of most invertebrate fauna (**Figure 21**) or at least an adequate fish food supply (Bush et al. 1974). Less complex organisms than invertebrates will also profit of the thermal fish protection.

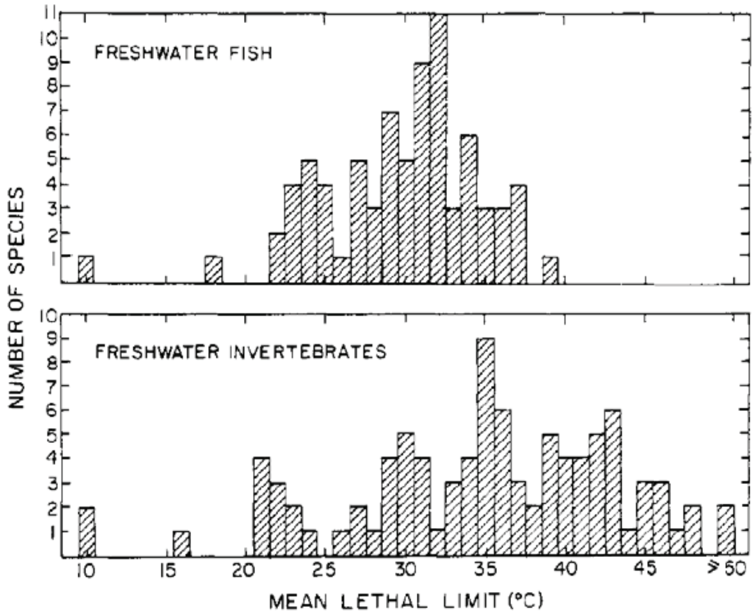


Figure 21. Number of species and mean lethal limit for freshwater fish and invertebrates (Bush et al. 1974).

(Jobling 1981) suggested to divide fish-responses to temperature into a tolerance, a resistance and a preference zone (**Figure 22**). This approach or similar once are widely used. In such classifications, temperature limits and acute preferendum depend on the acclimation temperature of the fish of a certain species. Fish acclimatized to a certain temperature will search the corresponding acute prefereundum. If fish are exposed to a temperature gradient for a longer period, they will gradually gravitate towards its temperature of final preferendum (Jobling 1981, Cincotta and Stauffer 1984). The species specific temperature of the final preferendum offers the most favorable thermal conditions for the individual, allowing optimal growth (Jobling 1981). Acclimation to a altered temperature occurs relatively rapidly in fish, usually faster than 1°C in 24 hours (Fry 1967). Upper and lower incipient lethal temperatures (VILT and LILT) represent the temperatures at which, theoretically, 50% of the population could survive indefinitely (Jobling 1981). Outside the tolerance temperatures lies the zone of resistance, within which there is a strong interaction between temperature and exposure time (Jobling 1981). Fish (especially big once) are known to be able to spend a certain time in warm water before their internal body temperature rises (Haynes et al. 1989). The upper boundary of the resistance zone is represented by the critical thermal maximum (CTM), above which survival is virtually zero (Jobling 1981).

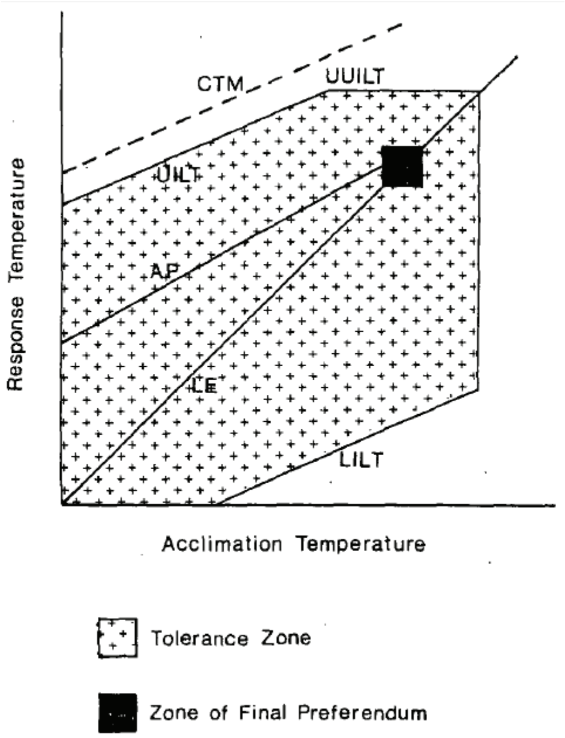


Figure 22. Diagram explaining the temperature relations of fish. The critical thermal maximum (CTM), the upper incipient lethal temperature (UILT), the lower incipient lethal temperature (LILT), the ultimate upper incipient lethal temperature (UUILT), the acute thermal preferendum (AP) and the line of equality (LE) are shown. If given a choice, fish will gradually gravitate towards the species specific temperature of final preferendum (Jobling 1981).

The concept of temperature tolerance of any organism, however, is complex and tolerance ranges for various functions are very specific (Langford 1990). Beside the acclimation temperature and the time of exposure to altered temperatures, many other factors such as life stage, prior thermal experience, water pollution, low concentration of dissolved oxygen or parasitic disease impact the tolerance zone of individual fish (Langford 1990, Coutant 1999).
Figure 23 shows the temperature dependence of two fish species (carp and brown trout). In many water, brown trout is the most temperature sensitive species (Broadmeadow et al. 2011). According to (Jobling 1981), the optimum growth occurs at or very close to the final preferendum. The figure shows that there are different temperature limits and requirements for different processes. Generally, many aquatic species have specific thermal optima for various stages of their life cycle (Preece and Jones 2002).

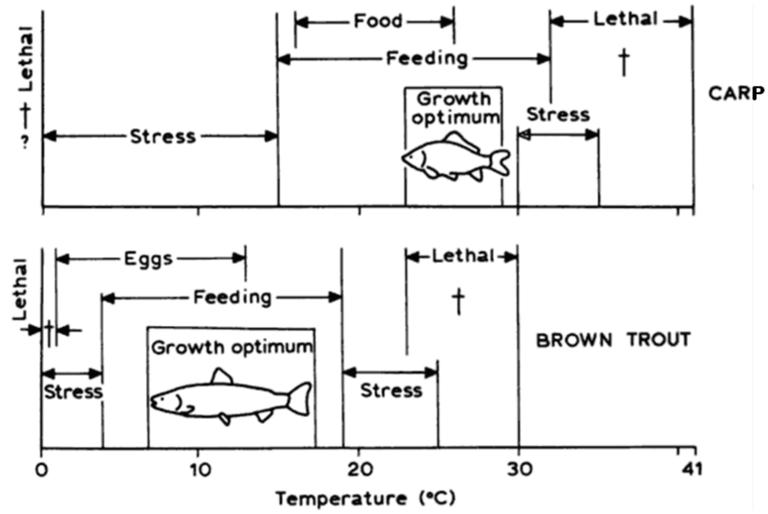


Figure 23. Comparison of thermal requirements of carp (*Cyprinus carpio*) and brown trout (*Salmo trutta*) ((Langford 1990) after (Elliott 1980)).

The range between thermal limits varies geographically between aquatic organisms. According to (Langford 1990), organisms can be broadly categorized as: cold stenotherms (narrow tolerance ranges (e.g. ± 10°C) in arctic regions), warm stenotherms (narrow tolerance ranges (± 10°C) in tropics), eurytherms (species with wide tolerance ranges (≥ 30°C), e.g. in temperate or sub-tropical regions). Hence, the effects of temperature alterations strongly depend on the temperature tolerances of the existing species and vary between different climatic regions. If and how organisms are affected by temperature alterations is highly dependent on the sensibility of the species at a certain temperature range. For instance, if temperature is increased from about 8 to 11°C, brown trout probably will not be affected; temperatures are still optimal (Figure 23). If an equal 3°C temperature increase occurs from 12 to 15°C, reproduction might be adversely affected, and an increase from 19 to 22°C could endanger a whole population. Hence, relatively high temperature alterations might not influence a species while small thermal changes at a critical temperature range might have severe consequences. However, high and maximal temperatures seem to be most crucial for fish. For example, (Lessard and Hayes 2003) found the mean summer temperature to be the best predictor for density of brook and brown trout. According to (Bush et al. 1974), lethal temperature limits for freshwater invertebrates and fish start at temperatures as low as 10°C. However, lethal limits of a greater number of species start at 21°C, while high losses, especially for fish, are to expect at temperatures around 30°C (Figure 21) (Bush et al. 1974). For warm water streams, the maximum allowable temperature with unspecified duration is about 32°C, and the same tolerance limit can be set for normal invertebrate communities (Bush et al. 1974). For cold water streams in contrast, sometimes not higher temperatures than 25°C are accepted in accordance with the upper lethal limits of brown trout (Lessard and Hayes 2003). In Lake Stechlin in northeast Germany, (Koschel et al. 1985) found the fallowing effects of high temperatures in the thermally polluted lake:

- < 25°C: no considerable physiological damage
- > 28°C: above average destruction and decrease of primary production
- ~ 30°C: phyto- and zooplankton damaged irreversibly

(Koschel et al. 1985) recommend to not heat water bodies comparable to Lake Stechlin above 27°C (nearly the same limit was discussed by other authors) due to the physiological effects on aquatic life. (Bush et al. 1974) found that in five warm water streams in the USA (upper and lower

Mississippi, Tennessee River, Delaware River, Sacramento River), at about 24 ± 2°C, fewer than one half of the fish species were within their preferred temperature range, while at 32-34°C, one half of the fish species are expected to be lost from the system. In the Columbia River, which contains a larger proportion of cold water forms, one half of the fish species would be beyond their preferred range at temperatures exceeding 20°C while one half the species would be lost at 28-30°C (Bush et al. 1974).

Increasing temperatures lead to higher metabolic rates in organisms. Following standard chemical (Q_{10}) laws, the rates of chemical reactions double with about every 10°C increase in temperature (Coutant 1999). Hence, temperature is the most important factor influencing growth rates of aquatic organisms (e.g. Langford 1990, Imholt et al. 2010). Higher temperatures require more energy from biota to sustain increased rates and processes and can deplete the energy reserves of individual fish (Thomas et al. 1986). Furthermore, oxygen demand rises as shown by (Ziarek et al. 2011) for Daphnia. If water temperature are lowered by cold water pollution, lower growth rates are observed (Sherman et al. 2007). This might especially have adverse effects on warm water species as the Australian Murray cod, making them more susceptible to predation and other interactions with introduced species (Sherman et al. 2007). However, temperature is not the only controlling factor on growth rates. There might be additional, possibly limiting factors like food supply or oxygen availability (Langford 1990). Regarding the behavior of fish to gain optimal growth, according to (Wildhaber and Lamberson 2004) the bioenergetics paradigm that fish respond to temperature as well as food based on energetic gains and losses are widely accepted.

It has been reported that the incidence and severity of viral and fungal infections of fish can be related to high water temperatures (Langford 1990). Thermal stress reduces the resistance of fish (Thomas et al. 1986). Recently, the proliferative kidney disease (PKD) which infects salmonids was a great issue in many cold freshwater bodies. According to (Burkhardt-Holm et al. 2005), PKD is one of three key factors for the decline of fish catches in Switzerland. Clinical PKD is characterized by a temperature-dependent proliferative and inflammatory response to parasite stages in the kidney (Okamura et al. 2011). Laboratory and field studies demonstrate that increasing temperatures enhance disease prevalence and severity and distribution of PKD-related mortality (Okamura et al. 2011). The temperature limit of 15°C was found to be a critical factor for PKD-disease outbreak in fish (Wahli et al. 2002) and according to (Burkhardt-Holm et al. 2005), 90% of the PKD-induced mortalities in Swiss waters occur when water temperatures surpass 15°C for two weeks or more. Furthermore, increasing temperatures can be expected to result in higher spread of the parasites and during abnormally high winter temperature, PKD can develop (Okamura et al. 2011). However, factors other than temperature influence development and severity; above all eutrophication seems to have similar effects as high water temperatures (Okamura et al. 2011).

6.2 Behavioral reactions

Temperature alterations not only affect the physiology of organisms, but also their behavior. Although cold-blooded organisms like fish are not able to change their body temperature to any extent from that of the water they occupy, most do not just passively occupy whatever temperatures are presented to them. Mobile organisms routinely seek temperatures that are physiologically most advantageous, when they have that opportunity (Coutant 1999). Naturally, aquatic animal are often exposed to changing temperatures in natural habitats, both spatially and temporally (Coutant 1999). Such spatial temperature variability allows animals to find favorable thermal conditions and to maintain bodily functions as close to optimal as possible (Coutant 1999). (Ziarek et al. 2011) demonstrated a respond of Daphnia to ambient temperature by their individual swimming behavior, resulting in different types of motion that might be interpreted as different adaptations to the environment. The authors conclude that Daphnia interacts with its environment by adapting its swimming behavior in a fashion that maximizes the fitness and success in terms of energy intake and cost reductions while minimizing predation risks. However, adaption on environment temperatures is most distinct in fish and widely observed.

More obvious than fish searching for preference temperatures is temperature avoidance (Coutant 1999). For instance, salmonids were often found to avoid temperatures > 20-22°C (Haynes et al. 1989). (Spigarelli et al. 1983) found the abundance of salmonids in Lake Michigan (close to the cooling water discharge of the Point Beach Nuclear Plant) to drop sharply as water temperatures rose above 20-21°C. (Haynes et al. 1989) concluded, that the fishes might have left the thermally impacted areas when temperatures exceeded their thermal preferendum. Besides influencing the distribution of fish, temperature avoidance might also change their migration behavior because of too high temperatures over the whole stream profile (Coutant 1999). Such cases of thermal blockages can occur, when temperatures are about 3-4°C above the preferred temperature range of the fish species (Coutant 1999).

Natural, complex habitats usually have distinct spatial variations in temperature (Coutant 1999). While lakes normally are thermal stratified, the interaction of external temperature drivers and internal river structures produce heterogeneity in stream temperature at a variety of spatial and temporal scales (Poole and Berman 2001). Coldwater areas are normally found where there are coldwater tributaries, seeps or springs, deep pools, shaded zones and cold alcoves (Coutant 1999, Ebersole et al. 2003). Various studies have noted that, during high temperature events, many aquatic species move to areas of colder water, so called thermal refuges (Coutant 1999, Caissie 2006). Therefore, the existence of thermal refuges is of great importance. While in undeveloped basins thermal heterogeneity is common, this might not be the case for artificial river beds. Hence the effects of thermal pollution are expected to be less severe in undeveloped water bodies than in unnatural once.

According to (Caissie 2006), thermal discharges can adversely affect aquatic resources by reducing the available area of suitable habitat. However, that is mainly significant for immobile organisms or if a greater area is affected. The strong spatial variations in temperature caused by cooling water discharges (Neill et al. 1972) are normally used by fish to maintain their preferred body temperature. Thermal discharge areas might be avoided or preferred by fish, depending on fish species and season (Neill and Magnuson 1974, Haynes et al. 1989). Almost all in situ studies show increased fish abundance at warm water discharges in spring, autumn and winter and decreased abundance in summer (Sadler 1980, Langford 1990). Beside favorable temperature, other reasons for fish to occupy a thermal discharge were found, such as concentration of prey species or discharge currents producing a positive rheotactic response (Haynes et al. 1989). Hence the understanding of how different fish behave in heterothermal environments is crucial in predicting the ecological impact of heated effluents (Neill et al. 1972). Even if temperature clearly is one of the main factors affecting the distribution of aquatic organisms (Coutant 1999), temperature alone might not be sufficient for statement about changes of fish distribution due to thermal pollution.

Fish respond to all aspects of the temperature regime, including the maxima and minima, seasonal and diel fluctuations, the duration of extreme thermal events and rates of change (Broadmeadow et al. 2011). While a lot of research has been done on reaction of organisms to high temperatures, very little is known about the effects of high temperature change rates. Such abrupt temperature changes occur above all due to water reservoir-water release, causing a thermopeaking event. According to (Coutant 1999), salmonids are better protected if abrupt thermal gradients are avoided. However, temperature fluctuations are a normal phenomena in alpine streams. Therefore, direct physiological effects on benthic invertebrates at temperature changes of < 2°C appear unlikely (Frutiger 2004b). According to (Frutiger 2004b), growth rates of caddisfly (*Allogamus auricollis*) in the River Ticino seems to be mainly limited by food availability and not by temperature, and thermopeaking (caused by water release of the upstream reservoir) does not have any metabolic impact on caddisfly. (Carolli et al. 2011) conducted two cold thermopeaking and two warm thermopeaking simulations by quickly cooling the water by 3–4°C during the warm season and warming the water by 2–3°C during the cold season at a rate of about 2.4×10^{-1}°C min^{-1}. This rate is very similar to those associated with thermopeaking waves in rivers of this area in northern Italy (Carolli et al. 2011). Although the achieved temperatures were not close to lethal and the change rate in the tolerability range for benthic invertebrates, taxa responded quickly and pronounced by higher drifting (Carolli et al. 2011). The more pronounced drifting occurred in warm season, when there was cold water pollution (Carolli et al. 2011). However, the effect of flow increase by reservoir-water release might be more important on invertebrate drift than the associated temperature changes. (Bosco Imbert and Perry 2000) found highly increased drifting of benthic invertebrates caused by flow increase. While stepwise flow increase lead to a selective drift response of taxa, abrupt flow increase affected the whole benthic invertebrate community and causes higher drift density (Bosco Imbert and Perry 2000). (Carolli et al. 2011) conclude that the effects of thermopeaking in addition to those of hydropeaking caused by reservoir operation might be important in structuring the benthic communities in alpine rivers.

By changing the water temperature regime, thermal pollution influences the structure of biotic communities (Prats et al. 2010). As expected, warm water pollution was generally found to favor warm water species, while the opposite effect occurs for cold water pollution. However, the community structure might respond gradually to temperature alterations. When Rheinsberg Nuclear Power Station (northeast Germany) came in operation, phytoplankton- (Koschel et al. 1985) and macrophyte- (Casper et al. 1985b) community in Lake Stechlin reacted shock-like, totally new community compositions occurred. At the same time, there was not any change in the community composition of the crustacean zooplankton (Weiler et al. 2003). However, after some time, a new equilibriums (partially approached the previous ones) of phytoplankton- and macrophyte-community composition established (Casper et al. 1985b, Koschel et al. 1985). Contradictory effects on community diversity caused by warm water pollution were observed. While (Lessard and Hayes 2003) reported an increase, (Bush et al. 1974) found a decrease in diversity. Very pronounced temperature alterations, as it may be found for instance at effluents of warm water discharge, can allow the survival of exotic species. Population of alien fish have been found at warm water discharges (Langford 1990). However, (Langford 1990) concludes, that changes in communities are mostly caused by scour or biocides and not by changed temperature, except in the warmest effluents.

7 Discussion

To estimate the effects of thermal pollution on lakes and rivers, it must be known which components of the ecosystem functions are sensitive to water temperature changes. Aquatic organisms react to all alterations of the water temperature, but the responses are diverse. **Table 3** summarizes the temperature effects for different sources of thermal pollution. Reduction of river discharge alters especially extreme temperature values on all time scales, hypolimnetic water discharge by reservoirs strongly dampens the annual temperature cycle and increases the rates of short term temperature changes (thermo-peaking), surface water release by reservoirs increases summer temperatures, reduction of riparian shading increases temperature maxima during spring and summer and thermal warm-water discharges increase temperatures all year round. The effects of the different sources of thermal pollution can interact. Especially reduction in river flow increases the sensibility to any temperature driver and therefore amplifies the effects of other anthropogenic thermal pollutions.

Table 3. *Relevance of the most important thermal pollutions on different aspects of the water temperature regime. If there is an increase instead of a reduction of river discharge or riparian shading, the reverse effect occurs. +, ++ and +++ indicate a low, medium and high increase. -, - - and - - - show equivalent decreases. For the time shift, the plus sign indicates a later, the minus sigh an earlier occurrence in the yearly cycle.* ○ *indicates no significant change.*

Aspects of the water temperature regime	**Source of thermal pollution**				
	Reduction of river discharge	Reservoir with hypolimnetic water release	Reservoir with surface-water release	Reduction of riparian shading	Thermal discharges
Annual mean temperature	+	○ to - -	+	+	+ to +++
Summer temperature	++	- - -	++	++	+ to +++
Winter temperature	- -	++	○	○	+ to +++
Maximum temperature	+++	- - -	+	+++	+ to +++
Minimum temperature	- - -	++	○	○	+ to +++
Diel temperature amplitude	+++	○	-	+++	-
Rate of diel temperature changes	+	+++	+	+	○
Time shift of the annual max/min temperatures	○	++	○	- -	○

Changes of the water temperature regime affect aquatic organisms directly (e.g. by influencing their metabolism) as well as indirectly by changing physical and chemical properties of a water body (e.g. dissolved oxygen, stratification or duration of ice cover). Therefore, it is not enough to consider direct temperature effects on aquatic ecosystems only. Furthermore there are many environmental parameters determining the reaction of an ecosystem to thermal pollution. The distribution of the thermal plume and above all the size of the affected area compared to the total size of the water body is of great importance. If only a comparable small part of a water body is affected, there will be rather changes in behavior of mobile organisms than pronounced alterations of the aquatic ecosystem. Hence, for small affected areas even pronounced temperature alterations might not have significant ecological impacts. Thermal variability is important for mobile aquatic organisms (especially fish) to maintain a favorable body temperature and to endure extreme temperatures in thermal refuges. In undeveloped basins, there is such a thermal variability. In highly anthropogenic altered basins however, this might not be the case. Hence, aquatic ecosystems in natural water bodies will react less sensitively to thermal alterations. Stress factors as contaminants or parasitic affection decrease the temperature tolerance of aquatic organisms. Also important is the existence of highly temperature controlled ecosystem parameters. For instance, if temperature in a water body is the limiting factor for primary production during winter, a substantial increase is to expect if water temperatures are warmed during the cold months of the year. Temperature tolerance varies between ecosystems. Depending on the geographical region and hence on the climate conditions, some species have a small, some a wide thermal tolerance range. The less water temperature variations occur naturally, the more severe will thermal pollution affect the ecosystem. In water bodies and its ecosystems, internal processes might amplify or damp effects caused by thermal pollution.

Given a certain change of water temperature, it might not affect aquatic organisms at all, if the alteration happens within the preferred temperature range of the organism. The same temperature change can have severe consequences for the organism though, if temperature crosses a thermal limit (especially the lethal limit). Therefore, it is essential in which temperature range a change in the temperature regime occurs. At high temperatures, lethal limits of many species are reported (Figure 21). Therefore, sensitivity on temperature increase is most pronounced at high temperatures.

It is much easier to assess maximum tolerable upper temperature limits in water bodies to protect ecosystems than to assess maximum tolerable temperature changes. Some important temperature thresholds are shown in **Table 5**. If temperatures do not exceed about 21 °C, effects on ecosystem will be rather moderate. At sites, where very temperature sensitive species are not present (e.g. no salmonids), maximum temperatures of 27 °C seem to be tolerable. Generally, in temperate and cold climates temperatures exceeding 28 °C should be avoided because of potential major damage of ecosystems.

Table 4 summarizes the observed effects of thermal pollution on aquatic life. The occurrence and severity differ strongly between the individual cases. However, stronger effects are rather seen on standing waters than on rivers. Small temperature alterations of about 0.5 °C seem not to have any effect. Despite partly pronounced temperature alterations, many authors conclude that surprisingly little effects on aquatic ecosystems were caused by thermal pollution (especially in rivers; e.g. Langford 1990, Levin et al. 1972, Maderich et al. 2008, Prats et al. 2010). Other factors might be more important than temperature itself. In case of thermal effluents for example, (Langford 1990) expects the controls on outfall siting and design and the use of biocides to have grater effects on ecosystems than water temperature increase.

In many countries, upper temperature limits and maximum limits for temperature alterations have been assessed by law to protect aquatic live. In Switzerland, maximum temperature increases of 3 °C (in trout waters 1.5 °C) with upper limits of 25 °C are tolerable. In the European Union, the same values for temperature increase are used with upper limits of 21.5 °C in salmon waters and 28 °C in cyprinid waters. Such thermal limits are arguable though, particularly in waters where dilution is high and mixing rapid (Langford 1990). According to (Langford 1990), many temperature limits and restrictions in maximum temperature rises (such as the described) are ecologically unnecessary and economically indefensible in many waters. (Coutant 1999), on the other hand, suggests simple, stringent standards applied at index locations based on general beneficial uses, followed by flexibility in how the standards are met.

In consideration of the reported cases of thermal pollution and its effects on ecosystems, the temperature limits in Swiss and EU law seem to be rather restrictive. Following the findings of this work, temperature rises of up to about 4 °C at a temperature range below 21 °C might be accepted. At 21-28 °C, temperature increase of 2-3 °C is tolerable. 28 °C should not be exceeded because of a too high risk of ecological damages. Ecosystems in lakes react far more sensible to thermal pollution. In case of a large scale temperature change, a maximum temperature increase of 1 °C should be accepted. If only a small part of the lake area is affected, far higher temperature increases might not cause any major adverse effect. However, such tolerable temperature limits have to be adjusted to local circumstances. For instance, if the naturalness of water body is low or if the water is contaminated, the acceptable temperature changes must be lowered. On the other hand, if temperature alterations occur during a very short time only, higher upper limits and temperature alterations might be tolerable. Generally, cold water pollution is less problematic than warm water pollution. Concerning the general temperature increase in many water bodies of during the last decades, effects of cold water pollution might approach the previous temperature regime. The best way to assess acceptable thermal limits would be the development of a simple model which includes and combines the most important factors.

Table 4. *Effects caused by temperature alterations (effects). The location, the source of the thermal pollution and some additional comments on the studied case are shown (comments). The data source is also report (source).*

ΔT (°C)	Effects	Comments	Source
-5°C	Major threat to aquatic biota, disturbance of spawning of native fish species	Namoi River, New South Wales (Australia), reservoir (hypolimnetic water release)	(Preece and Jones 2002)
-3-4°C	Higher drifting rates of benthic invertebrates	Abrupt temperature decrease (0.24 °C min^{-1}), typical for thermo-peaking (abrupt reservoir release)	(Carolli et al. 2011)
-2°C	No effect on majority of aquatic organisms	Alpine streams, reservoir (thermo-peaking due to abrupt water release)	(Frutiger 2004b)
-0.5°C	No or insignificant effects on ecology	General rule	(Langford 1990)
+0.5°C	No or insignificant effects on ecology	General rule	(Langford 1990)
+1°C	Increase of primary production, increase in abundance of macrozoobenthos, changes in community structures, alterations and time shift in life cycles, increased bacteria activity, trends towards eutrophication	Lake Stechlin, Brandenburg (Germany), thermal power plant, local temperature increase up to 10 °C, effects of increased nutrient input involved	various authors

+2°C	No effect on majority of quatic organisms	Alpine streams, reservoir (thermo-peaking due to abrupt water release)	(Frutiger 2004b)
+2-3°C	Higher drifting rates of benthic invertebrates	Abrupt temperature increase (0.24 °C min^{-1}), typical for thermo-peaking (abrupt reservoir release)	(Carolli et al. 2011)
+3°C	Damage and reduction of abundance and diversity of algae, small increase in mollusks and crustaceans, decrease of fish species	Biscayne Bay, Florida (USA), various thermal power plants	(Levin et al. 1972)
	Decrease in cold water fish species, increase in total fish species, shifts in macroinvertebrate community	10 rivers in Michigan (USA), reservoir (surface water release), mean summer temperature increase up to 5.5 °C (mean 2.7 °C)	(Lessard and Hayes 2003)
	No negative effects observed on aquatic life	Ebro river, Catalonia (Spain), thermal power plant, summer-temperature increase 2-4 °C (annual mean 3 °C)	(Prats et al. 2010)
+4°C	Killing or great reduction of many animals and plants	Biscayne Bay, Florida (USA), various thermal power plants	(Levin et al. 1972)
+7°C	Increase of abundance and species diversity of fish	Trent river, Leicestershire (UK), thermal power plant, temperature increase 4-12 °C (mean 7 °C), maximum temperature ≤ 28 °C	(Sadler 1980)
+ 10°C	Fish avoided or concentrated in the outfall area, depending on species, season and daytime	Lake Monona, Wisconsin (USA), thermal power plant, maximum summer temperature up to 35 °C	(Brauer et al. 1974), (Neill and Magnuson 1974)
	Three times higher average production, decrease of epifaunal abundance and disappearance of a few species in warmest months	Effluent channel, Maryland (USA), thermal power plant, temperature increase 12.7 °C in winter and 6.4 °C in summer	(Levin et al. 1972)
+ 11°C	Weight loss of catfish by 20%	Effluent channel, Connecticut (USA), thermal power plant	(Levin et al. 1972)

Table 5. *Some temperature limits for ecologically important aspects (effects) in general or for specific sites (comments). The data source is also reporte (source).*

T (°C)	Effects	Comments	Source
15°C	Threshiold for massive increase in risk for of PKD-outbreaks and PKD-induced mortalities for salmonids	90% of all brown trout PKD-induced mortalities in Swiss waters occur when temperatures surpass 15 °C for two weeks or more	(Burkhardt-Holm et al. 2005), (Wahli et al. 2002)
20-22°C	Threshold above which salmonids avoid areas with higher temperature	Outfall area of Point Beach Power Station, Lake Michigan (USA)	(Haynes et al. 1989), (Spigarelli et al. 1983)
21°C	Threshold for lethal limits for a larger number of fish species		(Bush et al. 1974)
25°C	Lethal limit for brown trout	In many waters, brown trout is the most temperature-sensitive species	(Langford 1990, Lessard and Hayes 2003)
	Up to this temperature no considerable physiological damage of organisms	Lake Stechlin, northeast Germany	(Koschel et al. 1985)
27°C	Further heating cannot be recommended; massive ecological consequences	Lake Stechlin, northeast Germany and comparable waters	(Koschel et al. 1985)
28°C	Destruction of organisms and decrease of primary production sets in	Lake Stechlin, northeast Germany	(Koschel et al. 1985)
	Many fish in European rivers are able to tolerate this temperature without harm		(Langford 1990)
28-30°C	One half of fish species lost	Columbia River, USA	(Bush et al. 1974)
30°C	A great number of fish species reach their upper lethal limits		(Bush et al. 1974)
	Irreversible damage of phyto- and zooplankton	Lake Stechlin, northeast Germany	(Koschel et al. 1985)
32°C	Maximum allowable temperature with unspecific duration for warm water streams, tolerance limit for normal invertebrate communities		(Bush et al. 1974)
32-34°C	One half of fish species lost in warm water streams	Upper and lower Mississippi, Tennessee River, Delaware River, Sacramento River (all USA)	(Bush et al. 1974)

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