Article

Are coldwater fish populations of the United States actually being protected by temperature standards?

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Abstract

Governmental water quality agencies are faced with identifying water quality goals to protect aquatic biota, applying the best available science in development of water quality standards and associated management principles, implementing water quality laws so that there is consistency with goals, developing guidance documents for applying science to law, monitoring, and enforcement. Deviations in this path from goals to standards to enforcement, however, are common across countries as well as among States within the United States and can result in failure to protect the aquatic biota.

The Clean Water Act (CWA) is the key US law for water quality protection. Its goal is to 'restore and maintain the chemical, physical, and biological integrity of the Nation's waters' and to fully protect the most sensitive beneficial uses. The US Environmental Protection Agency (EPA) Gold Book guidance for development of protective water temperature standards, dating from 1973, still recommends the use of MWAT (Maximum Weekly Average Temperature) as an index for assigning protective chronic temperature standards to coldwater fisheries. MWAT, applied according to EPA guidance, is typically used in conjunction with an acute upper limit. Unfortunately, MWAT is a criterion that is not protective, as can be shown by reference to several case studies on salmonids. Use of MWAT at a basin scale can result in considerable reduction in available salmonid rearing area and can, in many cases, be little better than recommending the upper incipient lethal temperature as a standard. Although MWAT is not used by many US States in standards, it is problematic in that it is cited as the official EPA model for a protective standard. The conceptual use of MWAT highlights some critical problems in application of the Clean Water Act and its associated federal regulations for protection of coldwater fishes, such as the concepts of full protection, protection of the most sensitive species, restoration of water quality, and support of species' viability at a basin scale.

Full implementation of the CWA in support of salmonids' thermal requirements has been patchy, with some States taking criteria development, monitoring, listing, and TMDL (Total Maximum Daily Load) development seriously and others virtually ignoring the problem. In addition, Section 316 of the CWA conflicts with the basic goals of the CWA by giving deference to the thermoelectric power industry to discharge heated effluent under a process where variances granted supersede water quality-based limits. This is exacerbated by EPA 316 guidance permitting evaluation

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of biological trends amidst shifting or uncertain baselines in a limited set of RIS (Representative Important Species), rather than application of best available science to protect both the most sensitive species and the entire aquatic community that is reflective of high quality habitat conditions. Designing water temperature standards to be fully protective and supportive of species viability (abundance, productivity, spatial structure, and diversity) benefits from application of concepts of optimum growth and survival temperatures distributed at a basin scale with reference to the natural thermal potential along a river continuum. Some notable successes in standards development found in the US Pacific Northwest are offered as future national models.

Keywords: Water temperature; US Clean Water Act; salmonids; maximum weekly average temperature; thermal effects; chronic temperature; balanced indigenous community; natural thermal potential; incipient lethal; growth optimum; distribution limit; productivity.

Introduction

The US Clean Water Act (CWA*), enacted in 1972, is the cornerstone of surface water quality protection in the United States. The European Commission (2000) has established a water quality directive for members of the European Union with similar, ambitious goals for protection and restoration of good water quality status. Given the shared technical problems in setting meaningful standards and translating laws into effective tools for protecting aquatic communities, it is worth evaluating in detail some critical flaws in the Clean Water Act and its technical interpretation by the US Environmental Protection Agency (EPA), as well as individual US States, that thwart realisation of its goals. This review, hopefully, can lead to construction of more consistent and effective laws, technical guidance, and more technically defensible standards by US States.

Over the course of development and modification of the Clean Water Act, different perspectives on required levels of protection of aquatic resources and appropriate methodology for monitoring or indexing the water quality have emerged in different sections of the law and guidance documents. For example, 'full protection' of the designated beneficial use and protection of the most sensitive use, where there are multiple uses, is required, but in Sections 316(a) and 316(b), the use of Representative Important Species (RIS) is suggested as a means to monitor only a portion of the full community as an expression of the whole (EPA, 1977a, b). Also, the CWA states that provisions in Section 316 (the section of the Act relating to thermal discharges) take precedence over other requirements of the CWA (Code of Federal Regulations: 40 CFR §131) and even antidegradation policy (provisions to preserve and protect the existing designated uses and to minimise degradation) must be consistent with Section 316 (40 CFR §131.12). Since the early days of the CWA, States have given and renewed variances for heated discharge from power plants with scant consideration for long-term shifts in full aquatic diversity and species composition (Baum, 2004), despite the fact that control of point-source, and not non-point source, thermal pollution remains the key regulatory tool with strong permitting authority available to protect aquatic life. The CWA specifically leaves water quantity allocation to State authority (United States Code: 33 USC § 1251), although the linkage between water temperature compliance and streamflow is critical. It seems evident that different parts of the law have been interpreted in divergent ways that have become distanced from the original intent and, in many cases, technical conflicts and inconsistencies are built into the law. Advances in scientific understanding of species thermal ecology and aquatic community ecology, and development of sensitive community monitoring

* See Glossary on pp. 198-199.

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methodologies, suggest that better means are available to avoid the chronic and cumulative effects of thermal and other water quality impairments, and that greater clarity can be brought to future revisions of the CWA.

The current EPA national guidance (EPA, 2006) for water temperature still defaults to the EPA 'Gold Book' (EPA, 1986). This guidance originated from a National Academy of Sciences (NAS) methodology (EPA, 1973) for acute, time-dependent exposures and maximum weekly average temperatures (MWAT) for various seasons, including the growth (summer) and reproductive (spring and fall) periods, adjusted to local conditions for chronic exposure. This guidance was later repeated in the Red Book (EPA, 1976) and then the Gold Book (EPA, 1986). The MWAT index for temperatures providing protection against chronic thermal stress was derived from biological criteria (growth optimum and ultimate upper incipient lethal temperature). However, this index was then to be monitored in the field as a physical MWAT. Given that it is an average temperature, it is different physically from the growth optimum, which is an instantaneous or maximum temperature. Despite evidence available after release of the NAS methodology that this method incorporated substantial biological flaws, it has remained as a US model for protective temperature standards. I am aware of no scientific justification of its merits in protecting aquatic biota.

The overall purpose of this review is to examine some selected critical weaknesses in the technical guidance on temperature standards associated with the US Clean Water Act that thwart efforts to meet the intent of this law. It is hoped that this process will reveal how more effective biologically- and physically-based criteria can be built into temperature standards to protect the most sensitive species, as well as aquatic communities, on a watershed spatial scale. Objectives of this paper are to:

1. view a case history on point-source heat discharge in the Connecticut River as a means of illustrating the technical problems set up by old EPA technical guidance documents;
2. view several case histories illustrating why the EPA's Gold Book technical guidance for use of MWAT is inadequate for providing full protection; and
3. recommend methods to fully protect beneficial uses that are more reliable than those provided by using either MWAT or Representative Important Species and are less likely to result in increases in basin temperature distribution via uncontrolled cumulative effects.

Evaluation of a case history of water quality protection on the Connecticut River

The Connecticut River (mainstem and tributaries) is a good test case for demonstrating the regulatory problems in achieving recovery of coldwater fish. It is a large river system (29 100 km^2 and 655 km in length), spanning four States – Vermont (VT), New Hampshire (NH), Massachusetts (MA) and Connecticut (CT) (Jackson et al., 2005) – and once supported large runs of Atlantic salmon (*Salmo salar*), estimated to number at least 109 000 to 181 000 returning adults (NOAA & USFWS, 1999; Fay et al., 2006) based on the proportion of total US historical habitat found in the Connecticut River and estimated total US adult returns. This run comprised the southern boundary of the North American distribution of the species (Juanes et al., 2004), based on the location of the river mouth, with the historical distribution within the river itself spanning nearly 615 km of mainstem length from the mouth at Old Saybrook, CT to Beechers Falls, VT (Gephard & McMenamy, 2004). In 1798 this run was extirpated due to construction of a dam at Turners Falls, CT. However, eradication of the native Atlantic salmon run had been initiated with overfishing and large-scale habitat destruction, sedimentation, and extensive mill dam construction on tributaries (Gephard & McMenamy, 2004). Habitat degradation of the Connecticut River watershed has also been documented extensively for its significant impairment to brook trout (*Salvelinus fontinalis*) viability (EBTJV, 2006). Factors attributed to the decline of brook trout in the four States (VT, NH, MA, and CT) are high water temperatures, riparian habitat modification, sedimentation, dam inundation, and introduction of exotic species (EBTJV, 2006). Although water temperatures in the mid-Connecticut River are reported as averaging 21.1–26.7 °C, with periodic maxima of 32.2 °C (CRASC, 1998), these New England States have been extremely reluctant to admit to thermal pollution impacts. This is despite the extent of known thermal impacts on tributaries (EBTJV, 2006) or cumulative thermal impacts that have been reported from the major Atlantic salmon rivers of Maine (Fay et al., 2006).

Loss of the Atlantic salmon from the Connecticut River was met with hatchery stocking of fry from the Penobscot River, Maine in the 1860s, although extensive salmon harvest and lack of fish passage at the river's many dams led to failure of this programme (CRASC, 1998). With the advent of the Anadromous Fisheries Conservation Act in 1965 and the CWA in 1967, the four States plus the USFWS (US Fish and Wildlife Service) and NMFS (National Marine Fisheries Service, a division within the National Oceanic and Atmospheric Administration) began a new restoration programme in 1967 that has continued to the present. A compact among the States and federal agencies created the Connecticut River Atlantic Salmon Commission (CRASC). This effort has focused on widespread stocking of tributaries and installation of adult and juvenile passage facilities.

The mainstem Connecticut River between Gilman, VT and W. Stewartstown, NH supports 13 100 m^2 of Atlantic salmon habitat between rkm 484 (Gilman Dam) and 594 (Canaan Dam) (CRASC, 1998). Gephard & McMenamy (2004) stated that 'it seems likely that little of the mainstem Connecticut River downstream of the present-day site of the Ryegate Dam (rkm 440) historically supported Atlantic salmon rearing habitat'. This contrasts with the Penobscot River in Maine, which has a watershed area of similar size (22 300 km^2, compared to 29 100 km^2 for the Connecticut River), but supports Atlantic salmon spawning and rearing in the mainstem river all the way downstream to Orono, ME (USFWS & Maine ASC, 2006). In the Connecticut River basin, the major loss of salmon was attributed to construction of sawmills and gristmills (CRASC, 1998). The enormous geomorphic impact of water-powered mills throughout watersheds of New England and the mid-Atlantic region has only recently been revealed (Walter & Merritts, 2008), with the four States of the Connecticut River basin having among the highest density of mill dams by 1840. These dams produced increases in local anthropogenic base levels and typically spanned entire first- to third-order valleys. The effect was to trap extensive fine sediment deposits, obliterating the abundant valley wetlands (Walter & Merritts, 2008). Recent removal of the mill dams has resulted in high eroding banks in incised channels where fine sediment transport is high. It is entirely conceivable that the combination of high fine sediment levels and lack of the wetland interaction with the stream network (greater than 70 % of the network stream length being first- to third-order) has resulted in poor spawning and egg incubation conditions due to high levels of fine sediment in channel substrata, and poor summer rearing due to high water temperatures and altered summer flow regimes. Given this evidence on the extensive cumulative habitat degradation in the basin, it can easily be surmised that the mainstem Connecticut River was likely, before this degradation, to have been used extensively as rearing habitat.

The mainstem Connecticut River in the four States does not provide optimum summer rearing temperatures for native coldwater species, even though Vermont classifies the mainstem as a coldwater habitat under its water quality standards (Vermont NRB, 2008). Massachusetts considers the mainstem Connecticut River not to be a coldwater fishery, despite the occurrence of Atlantic salmon (Cohen, 2007). It requires that salmonids spawn in the river to warrant concern for water temperature. New Hampshire does not place the Connecticut River in whole or in part on the list of rivers fully supporting aquatic life (New Hampshire DES, 2004a). NMFS & USFWS (2005) identified mainstem and tributary water temperature as a threat to ESA (Endangered Species Act)-listed Atlantic salmon in Maine rivers. Even the Penobscot River mainstem has been noted as having significant adult kills from mean weekly temperatures of ≥ 23 °C (Fay et al., 2006). Despite the fact that Atlantic salmon conduct spawning migrations into August in the Connecticut River (CRASC, 1997) and there is an extensive effort to re-establish salmon in the basin, even adults are not protected with summer

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coldwater criteria. This point was overlooked in a recent court ruling that granted an additional temperature variance to Entergy Vermont Yankee nuclear power plant (VEC, 2008). This ruling emphasised that most adults are taken out downstream of the power plant discharge for hatchery production, but failed to consider sublethal effects in adult or smolt passage, downstream travel of thermal effects, or thermal restoration of the river. Establishing an upper summer temperature limit of 29.4 °C in the Connecticut River (VEC, 2008), and allowing thermal variances, does not address the biological observation that adult migration success decreases at temperatures above 23 °C (Fay et al., 2006). Under Section 316(a) only the 'Balanced Indigenous Population' (BIP) is considered in effects of the thermal discharge (VEC, 2008), not standard water quality criteria.

A thorough temperature TMDL (Total Maximum Daily Load), based upon a state-of-the-art temperature model and modelled vegetation and channel reconstruction, coupled with an in-depth review of historical documentation of fish use of the river prior to significant cumulative warming effects (if possible), would help resolve uncertainty concerning a river's natural thermal potential. Unfortunately, means are not yet available to model accurately the temperature restoration results of anticipated wetland recovery and improved connectivity of hyporheic flows with the floodplain; however, thermal modelling of riparian and channel morphological recovery is more easily handled. A TMDL will not be conducted in the Connecticut River, however, because it has never been listed as impaired for water temperature (TMDLs only being required for waters listed as impaired on the 303(d) list).

Eastern brook trout (*Salvelinus fontinalis*) is the native char species that was endemic throughout this basin and widely in New England and the mid-Atlantic States. Brook trout co-evolved with Atlantic salmon and were found in all Atlantic salmon rivers (NMFS & USFWS, 2005). Brook trout have been detected in the mainstem Connecticut River in small numbers (Merriman & Thorpe, 1976; Normandeau, 2005), but in the tributaries the local populations have exhibited extensive loss due to habitat degradation (EBTJV, 2006). One can infer from the scepticism of some regional biologists about rearing of Atlantic salmon juveniles in the mainstem Connecticut River (Gephard & McMenamy, 2004) that brook trout, likewise, would not rear in the mainstem. However, it is also known that, historically, sea-run brook trout commonly migrated throughout major rivers of New England (Moring, 2005) and probably did so in the Connecticut River (Gephard & McMenamy, 2004). Brown trout in the Connecticut River mainstem in Connecticut are commonly sampled by electrofishing near tributary mouths. It is presumed they are foraging in the mainstem (Jacobs et al., 2004). As a minimum, the obligate coldwater fish species of the Connecticut River utilised the tributaries extensively, together with the mainstem during adult and smolt migration stages; but it also seems highly likely that extensive reaches of the mainstem supported at least marginal rearing or holding opportunities in cold refuges, similar to the use of these habitats in large western US rivers by bull trout (*Salvelinus confluentus*) during summer (McPhail & Baxter, 1996; Faler et al., 2005) and the past use of large Montana rivers by western cutthroat (Sloat, 2005).

Against the backdrop of historic and current coldwater species' use of the river, the existing management of water temperature via State water quality regulations can be evaluated. Vermont has just four streams listed for water temperature impairment in the entire State (Vermont DEC, 2006). Only one of these streams is of any significance in the Connecticut River (West River), but it is listed only for a 10-mile reach for impaired secondary contact recreation (fishing and boating). Support of aquatic life was not mentioned as an impaired beneficial use (Vermont DEC, 2006), despite the significance of the West River in Atlantic salmon restoration (CRASC, 1998). In its 305(b) report (the statutory report on water quality) the State of Vermont (Vermont ANR, 2008) acknowledges the importance of maintaining riparian vegetation along its streams to prevent thermal deterioration of its waters by stating 'there are no strategic State-wide requirements that riparian landowners must maintain a minimum width of vegetation along bodies of water as there are in other States.' The Connecticut River mainstem was not listed for temperature or any other pollutant despite high levels of mercury, nutrients, and toxins in many of the tributaries. In New Hampshire, the 303(d) list (the statutory list of impaired/threatened surface waters where a TMDL is required) contains no listings at all for temperature (New Hampshire DES, 2004b). New Hampshire assigns to Class A waters (its highest class) a temperature standard of 'no anthropogenic change' and for Class B a standard stating 'no change that would "appreciably interfere" with the assigned use' (New Hampshire DES, 2009), despite having standards for temperature. New Hampshire does list streams on the basis of dissolved oxygen (DO) and pH; however, it designates that all State waters should support aquatic life. This confers an expectation of providing suitable 'chemical and physical conditions for supporting a balanced, integrated and adaptive community of aquatic organisms' (New Hampshire DES, 2005). New Hampshire uses benthic macroinvertebrates, or fish and benthic macroinvertebrates, as core indicators when deciding the 303(d) listings. Additional criteria include DO, pH, benthic IBI (Index of Biological Integrity), habitat assessment (e.g. epifaunal substratum, pool characteristics, sediment deposition, channel flow status, channel sinuosity, channel alteration, bank stability, vegetative protection, and buffer width), toxic substances in water and sediment, and exotic macrophytes. The State has two IBI thresholds that must be achieved for a water body to provide full support: ≥ 65 in the northern half of the State and ≥ 54 in the southern half. The NH guidance is ambiguous in its definition of full support, however, by stating that 'degrees of full support' include good and marginal ratings. Consequently, ratings of < 65 and < 54, respectively, imply non-support. There is no large river IBI to cover the mainstem Connecticut River itself. The habitat assessment is relatively comprehensive for physical habitat condition, but full support of beneficial uses can be claimed provided no more than one of the habitat condition factors receives a score of ≤ 10 out of 20. This implies that if 9 of the 10 habitat factors receive a score of 11 out of a maximum score of 20, while one receives a score of 5, for example, the stream can be claimed to provide full support. A more rational approach would be that if the vegetative cover were rated even as 11 out of 20 (ostensibly, indicating a function of 55 %), there should be a presumption that water temperatures, and possibly sediment deposition and other habitat factors, are not achieving desirable conditions. Obviously the bar for water quality as well as habitat conditions is set very low. And if such degraded habitat conditions are linked to reference biotic condition, the stage is set for a downward shifting baseline condition. In NH there were only 26 tributary stream segments identified as fully supporting aquatic life, out of a total of 3169 segments (or 163 miles out of 9612 miles) (New Hampshire DES, 2004a). Despite this, there are only three segments on the Connecticut River mainstem itself identified as not fully supporting aquatic life and needing a TMDL. These cases were attributed to aluminium pollution and non-native aquatic plants.

In Massachusetts there are only five river segments in the entire State for which thermal modification was listed as a cause of 303(d) listing. These segments all had additional causes for listing, such as metals, nutrients, suspended solids, pathogens, turbidity, oil and grease, and organic enrichment. There was not a single Connecticut River segment listed for thermal modification (Massachusetts DEP, 2005). Biological assessment, toxicity bioassays, and chemistry/water tests are used to evaluate whether the State's surface waters fully support aquatic life (Massachusetts DEP, 2005). The biological assessment is based on a combination of EPA Rapid Bioassessment (RBP III, see Barbour et al., 1999), fish community, habitat and flow, macrophyte, and algal bloom data. Fish communities, flows and macrophytes are evaluated by an unspecified 'best professional judgment' process. The chemistry/water assessment comprises a combination of DO, pH, temperature, and toxic chemical monitoring. The temperature criterion is maximum daily mean over a month period. A listing is permissible only when 10 % of the measurements exceed the standard. When a standard is based on a single maximum daily mean value over a month, presumably one could gather high values for July and August in a single year, but it would require multiple years of data collection for the 10 % criterion to have meaning. This means that the number of years of

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data required should be specified. Also, the statistic should be based on continuous daily monitoring for a maximum daily mean to be protective. Massachusetts DEP (2005) references the State's 1996 water temperature standards in its 2004 303(d) and 305(b) report to the EPA, even though these have since been revised. The most recent water quality standards indicate that coldwater fishery use is protected by a temperature standard of ≤ 20 °C, calculated as a 7DADM value (7-Day Average of the Daily Maximum). Despite the conflict between standards and the monitoring plan, the State has listed very few stream segments or miles as impaired, and no listings are provided in the entire Connecticut River mainstem or tributaries in the State. The State specifies use of a 'weight of evidence' approach to assessing the protection of aquatic life, implying that temperature may not necessarily be used when assessing water quality (Massachusetts DEP, 2005). In the Connecticut River mainstem, the only listings presented by MA include the following causes: pathogens, organics, suspended solids, and flow alterations. It is not likely that temperature impairment would be alleviated by addressing these other factors. If a mainstem segment had been listed for temperature it is unknown, but unlikely, that there would be an upstream, tributary-based evaluation of means to address the problem, unless a basinwide and multi-State TMDL were initiated.

In Connecticut there is only a single waterbody listed for water temperature standards exceedance (Muddy River). The Connecticut River has no listings for temperature violations in the mainstem or tributaries. There is no waterbody within the State for which a temperature TMDL is identified as being required (Connecticut DEP, 2006).

Despite the virtual total lack of recognition of warm water as a limiting factor to Atlantic salmon in the Connecticut River basin, elevated water temperature has been frequently cited as a key factor in the reduction of Atlantic salmon viability in the various Maine rivers (NMFS & USFWS, 2005). What appears as a most likely conclusion is that the long history of habitat alteration in the Connecticut River has led to an uncritically accepted 'conventional wisdom' of the low recovery potential for the river. This 'logic' works heavily against recovery of native salmonid species in the basin.

Shifting baselines

Because all four States containing the Connecticut River watershed have no biocriteria for the mainstem and because no mainstem segments are listed for temperature impairment, possibly because there is no baseline for historic temperatures or historic fish populations, the native coldwater fishes and even the coolwater fishes are increasingly threatened. The prospect of addressing widespread habitat alterations via the CWA will occur only if accompanied by a TMDL or a basin restoration plan linked to a TMDL. Incremental point-source thermal pollution is likely to continue, given that significant point-source variances have been granted readily and repeatedly (e.g. Vermont Yankee nuclear power plant). Given the deference to the thermoelectric power industry enshrined in Section 316(a) of the CWA, water quality standards and TMDLs are largely neutered. The fish community is evaluated using purely 'best professional judgment' according to the ambiguous guidance provided by EPA (1977a) to protect a 'Balanced Indigenous Population'. The States have no standards for what constitutes this balance, and exotic fish have been willingly adopted as 'indigenous' by the State fishery agencies, even though introduced members such as smallmouth bass constitute a significant predation threat to rare cyprinids (Jackson & Mandrak, 2002) as well as to coldwater species restoration.

Lack of a minimally perturbed fish community reference point for a large river such as the Connecticut River facilitates acceptance of a shifting baseline as the standard against which to measure impacts of new thermal discharges (Pinnegar & Engelhard, 2008). If we visualise a fish community composed of select salmonid, percid and centrarchid species that represent cold-, cool-, and warmwater preferences respectively, their distributions could overlap extensively throughout a drainage system such as the Connecticut River (Kitchell et al., 1977). Progressive river warming would cause shifts in relative abundances of the members of these thermal guilds, but each would maintain presence throughout the mainstem course. If the salmonids were present with minimal abundance because of cumulative warming, further warming would probably be difficult to detect via biological sampling alone, given statistical problems in sampling rare species. The EPA (1977a) guidance on protecting RIS emphasises the importance of detecting shifts in species composition and relative abundance. The minimum detectable effect (Ham & Pearsons, 2000) coupled with known high levels of variance in anadromous and resident species' annual abundances (Bisson et al., 2008), under what might be considered dynamic equilibrium habitat conditions and shifts in age structures, make early detection of significant biological change difficult. Biologically important changes in species composition and abundance that accompany progressive thermal increases may occur before they are detected and attributed to a thermal shift; indeed, species composition and abundance may need to change drastically before the changes are detected, attributed to a thermal shift rather than normal inter-annual variation, and reversed. Unfortunately, given a conservative estimate of the coefficient of variation of anadromous salmonid populations in Pacific Northwest streams (CV $\geq$ 50 %), a 35-year pre-and post-treatment monitoring programme would be required to detect a 30 % change in population abundance ($p \leq 0.10$) (Bisson et al., 2008). Protection of the rare species is vital in biocriteria development, but is not guaranteed by selection of a minimal number of RIS that emphasise primarily the abundant and economically dominant members.

The challenge in aquatic resource management today is in setting a baseline against which to establish limits to allowable change. Until there is significant monitoring or survey work done, baselines can constantly shift (Pinnegar & Engelhard, 2008). This process sets a progressively lower expectation. The continuum of fish community types that would be found under historic conditions in a stream system can constantly shift headward as a stream is developed, with generalised stream heating occurring from cumulative sources. This results in a compression of coldwater fish distribution and relegation to headwater zones where the most cold-tolerant members of these species may maintain relative advantages due to greater tolerance of cold waters and high channel gradients. At the same time, the warmwater fish guilds are able to considerably expand their range. Depending upon the point in this developmental sequence when reference conditions are established and current conditions of developed streams assessed, the baseline could have shifted dramatically. There are a variety of means for inferring a historic baseline for regulating stream temperatures. If there are suitable reference conditions for streams of the size undergoing greater development, this is a good option. Reference conditions that reflect natural potential conditions (cf. 'natural thermal potential'; ODEQ, 2009), however, are often lacking, especially for larger streams. Temperature monitoring in reference streams provides a good regional target for maximum temperatures relative to stream size and elevation. Historic collections of fish can be used to infer expected zonation in terms of percentage composition, but frequently, fish collections document primarily presence/absence. Consequently, from community composition it may not be easy to denote dominance or abundance, and thereby, former core use areas. Data on potential natural vegetation distribution, vegetation height and canopy cover, stream size and orientation relative to topography, documented changes in channel geomorphology, and the current channel and vegetation status make it possible to reconstruct historic temperature regimes using state-of-the-art mechanistic temperature models (Boyd & Kasper, 2002, 2003). Statistical models can also be used with reference streams to extrapolate temperature characteristics to developed streams based on classification criteria (Donato, 2002; EPA, 2003a; Risley et al., 2003).

Problems with application of Representative Important Species and Balanced, Indigenous Community concepts

In order to secure a variance for discharge of heated effluents into rivers, operators of electric power generation plants (e.g. nuclear power plants) must conduct a 316(a) 'demonstration' that the effects of a proposed thermal

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discharge adequately 'assure[s] the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife in and on the body of water into which the discharge is to be made…' (33 USC §1326(a), 40 CFR §125.73). Under 316(b), they may also need to consider the application of best available technology to the structure of cooling water intakes. In addition, EPA guidance requires selection of 'Representative Important Species' (RIS) whose presence or abundance may be causally linked to the heated discharge (EPA, 1977a). In the EPA guidance, these species should be a combination of threatened or endangered species, commercially or recreationally important species, the most thermally sensitive species, and species significant in the food chain. Sections 316(a) and 316(b) of the CWA are very brief, but the amplification of these in EPA guidance (EPA, 1977a, b) introduced what appears to be exceptional ambiguity in the definition of 'balanced indigenous community' (BIC) so that it is disconnected from modern, rigorous EPA concepts of biotic integrity. This view that Section 316 guidance is in need of updating is supported by EPA (1992):

> *It should be noted that WQS [i.e. water quality standards] in many States are not based on the extensive data and modern scientific theories that have become available since the standards originally were issued. Largely because of the availability of Section 316(a) of the CWA, which enables permittees to perform site-specific evaluations in lieu of applying WQS, many States have not chosen to update their thermal WQS with the new data and procedures that have become available since that time. (EPA, 1992).*

The Vermont Yankee Nuclear Power Station's permit amendment process serves as a case study to illustrate problems in the CWA and application of EPA guidance concerning the impact of heated effluents on fish communities. The Vermont Yankee (VY) nuclear power station is located at Vernon, VT, near the southern Vermont border where the Connecticut River crosses into Massachusetts (rkm 229.5). It began operation in 1972 and used closed cycle condenser cooling in order not to discharge heated effluent to the river. In 1974, VY was permitted experimental thermal discharge to the river to study the potential biological impacts. The first 316(a) demonstration was published in 1978 (Binkerd et al., 1978), supporting VY's proposal to allow winter (15 October–15 May) Connecticut River temperatures downstream of the powerhouse and dam to be increased by up to 7.4 °C to a maximum river temperature of 18.3 °C by operating the reactor cooling system in open-cycle mode. A maximum rate of river heating, not exceeding 2.8 °C per hour, was also proposed. Proposals in this demonstration were adopted by Vermont in an NPDES (National Pollutant Discharge Elimination System) permit in 1978 (Normandeau, 2004). In 1986, VY was granted an NPDES permit that allowed a variance for increasing the river temperature in summer by 0.56 °C. Another demonstration was published in 1990 (Downey et al., 1990) that supported alternative thermal limits, relative to ambient water temperature, for the summer period (16 May–14 October). The thermal limits endorsed in the demonstration report were adopted in the 1991, 1996, 2001, 2003, and 2004 NPDES permits. These permits provided VY an allowable summertime increase of 2.8 °C when temperatures were less than 12.8 °C, with stepped increases down to an allowed 1.1 °C increase for ambient river temperatures above 17.2 °C (Table 1). In 2003, a new demonstration was submitted (Normandeau, 2003) requesting an additional increase (0.56 °C) to summer temperatures to allow additional open-cycle mode cooling when ambient river temperatures were between 12.8 °C and 17.2 °C. An increase of 1.1 °C was also requested above ambient temperatures greater than 25.6 °C. In the subsequent revision to this demonstration (Normandeau, 2004), based on review by the Vermont Agency of Natural Resources (VANR) and USFWS, it was decided that the proposed increases should apply only to 16 June–14 October in the summer period so that the existing Atlantic salmon smolt migration would not be subject to this increase. The 2006 NPDES permit adopted by VANR for the VY power station included the proposed temperature increases from the 2003 and 2004 demonstrations for the 16 June–14 October period, with the added requirement of a temperature cap of 29.4 °C. After a series of legal challenges, the Vermont Environmental Court sustained the summertime temperature increases and cap but

Table 1. Magnitude of temperature variances granted for the Vermont Yankee Nuclear Power Station in Vernon Pool, Vermont after various 316(a) demonstrations and the recent court decision on a variance application (VEC, 2008). Copies of State NPDES permits were obtained from the Vermont Agency of Natural Resources (VANR) by the Vermont Law School. [1.] Renewed in 1996, 2001, 2003, and 2004. [2.] Temperature standards set through action of the Vermont Environmental Court proceedings (VEC, 2008).

Summer operating period							
	1978 NPDES permit	1986 NPDES permit	1991 NPDES permit[1]	2006 Amended NPDES permit		2008 VEC[2] decision	2008 VEC decision
	16 May–14 Oct	16 May–14 Oct	16 May–14 Oct	16 May–15 June	16 June–14 Oct	16 May–7 July	8 July–14 Oct
	closed cycle	allows 0.56 °C increase in summer			cap of 29.4 °C for entire period	cap of 24.8 °C for 16 June–7 July	cap of 29.4 °C for entire period
Temperature range (°C):	Allowed temperature increase (°C):						
> 25.6					1.1		1.1
> 24.8							
> 17.2 and ≤ 25.6					1.7		1.7
> 17.2			1.1	1.1		1.1	
> 15 and ≤ 17.2			1.7	1.7	2.2	1.7	2.2
< 15					2.8		
> 12.8 and ≤ 15			2.2	2.2		2.2	2.8
< 12.8			2.8	2.8		2.8	2.8

Winter operating period

Permitting conditions for the winter operating period (15 Oct–15 May) throughout this series of years have been that the temperature below Vernon Dam and the powerhouse would not exceed 18.3 °C, the rate of temperature change would be less than 2.8 °C/hr, and the total temperature increase above ambient below Vernon Dam would not exceed 7.4 °C.

restricted them to 8 July–14 October to provide further protection to Atlantic salmon smolt migration (VEC, 2008). It also imposed a 24.8 °C temperature cap for 16 June–7 July.

The chronology of 316(a) demonstrations and NPDES permits in the Connecticut River for operation of Vermont Yankee nuclear power station reveals a consistent effort to increase the thermal limits. These increases make no allowance for future behavioural extension of the migration period under thermal recovery or the possibility of mainstem juvenile rearing (VEC, 2008). When ambient (i.e. temperatures upstream of the power station) are between 17.2 °C and 25.6 °C, the most recent permit allows an increase of 1.7 °C from 8 July–14 October despite evidence of mid-summer adult salmon migration (CRASC, 1997), making the entire chronology of summertime temperature increases counterproductive. It is difficult to argue that the temperature increases allowed in the 16 May–7 July period are neutral in their effect on either Atlantic salmon smolts or American shad adults. Between 1993 and 1997 the date of 95 % completion of Atlantic salmon smolt emigration occurred just prior to 1 June (McCormick et al., 1999). In the 228-km migration from the mouth of the Connecticut River to Vernon Dam, American shad expend between 35 % and 60 % of their energy (Leonard & McCormick, 1999). Rate of energy expenditure in the last 30 km of travel to Vernon Dam is highly energy intensive for female shad, which appear to reach bioenergetic thresholds in warm years. This energy depletion appears to affect iteroparity in shad (Leonard & McCormick, 1999), and may likewise affect Atlantic salmon. At a minimum, these variances do not fully protect seasonal uses of salmonids in the river (Atlantic

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salmon (*Salmo salar*), rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*), or brook trout (*Salvelinus fontinalis*)), and they also do not fully protect adult migration or kelt downstream migration or lead to restoration. These variances were sought solely as a means to avoid using power to run the existing closed cycle cooling towers, which would be a logical, technically-based means of protecting water quality.

Rather than relying on WQBELs (Water Quality-Based Effluent Limits) or TBELs (Technology-Based Effluent Limits) (Kibler & Kasturi, 2007), Section 316(a) injects a concern for a representative fraction of the entire fish community into the NPDES process. Over the course of licensing the VY power plant, which began operation in 1972, approximately five years of baseline biological data were collected. The first pre-operational report (Webster-Martin, 1968) merely reported miscellaneous fish captures. A later pre-operational report tabulated numbers and biomass of fish collections by species made in the vicinity of Vernon, Vermont and south of Vernon Dam (Webster-Martin, 1971). A total of 24 species were noted. Several prominent species of the Connecticut River were not listed in the first pre-operational report: Atlantic salmon (*Salmo salar*), American shad (*Alosa sapidissima*), northern pike (*Esox lucius*), sea lamprey (*Petromyzon marinus*), blueback herring (*Alosa aestivalis*) and mimic shiner (*Notropis volucellus*). Appearance after 1981 of Atlantic salmon, American shad, sea lamprey, and blueback herring was probably attributable to installation of fishways at Turners Falls Dam and Vernon Dam (Normandeau, 2004). One species identified in the first pre-operational report, brook trout (*Salvelinus fontinalis*), was not found in the second but was observed in 2002 (Normandeau, 2004).

Binkerd et al. (1978) increased the total fish species list to 31 and also reported numbers and biomass from all fish collection methods combined (seining, gill netting, minnow traps, fyke nets and electrofishing). Unfortunately, numbers and weights reported combined all data by species for 1968–1977, mixing the pre-operational and post-operational periods. Consequently, there was no analysis of early shifts in community composition.

In the 1990 demonstration report (Downey et al., 1990), a comparison of the biomass and abundance of species downstream of Vernon Dam (i.e. downstream of the VY plant) for the pre-operational dataset of 1971 (Webster-Martin, 1971) against the 1981–1989 data (Downey et al., 1990), based on a combination of trap net, gill net, seine, and electrofishing, showed that for 23 species in common, there were significant shifts in percentage biomass for certain species. Most notably, white sucker (*Catostomus commersoni*) increased from 12.1 % to 44.9 %, smallmouth bass (*Micropterus dolomieu*) increased from 6.1 % to 13.0 %, carp decreased from 30.5 % to 8.7 %, and yellow perch (*Perca flavescens*) declined from 21.0 % to 4.0 % of total fish biomass. The 1971 pre-operational biomass accounted for 95.6 % of the total 1990 biomass, despite the difference in total numbers of species reported. The trends in fish species from pre-operational conditions to post-operation could possibly be attributable to changing fish sampling methods. But given the lack of comparison of pre- versus post-operational conditions with a consistent sampling protocol, it is inconceivable that a claim of 'no prior appreciable harm' could be established. Post-facto claims of no prior appreciable harm based on sampling with a 12-year 'relatively' consistent sampling protocol from 1991 to 2002 (Normandeau, 2004), conducted entirely during operational conditions, could possibly detect trends related to continuous operational conditions but pre- versus post- comparisons are not particularly meaningful given the minimal pre-operational sampling period and different methods used between the two sampling periods.

A 316(a) demonstration report was produced in 2004 by Normandeau Associates and submitted by Entergy Nuclear Vermont Yankee to VANR and the NRC (Nuclear Regulatory Commission, which is part of the US Department of Energy and is responsible for nuclear power plant permitting). This demonstration (Normandeau, 2004) summarised 12 years (1991–2002) of ecological studies conducted by Normandeau on the Connecticut River. Fish samples were taken by electrofishing in a reservoir created by Vernon Dam and in a riffle below the dam. Vernon Dam is located at rkm 228.3 and is 1.2 km downstream of the Vermont Yankee (VY) plant discharge.

The monitoring reach in the reservoir above the VY station is exclusively pool habitat and extends to 5.8 km above the discharge. The monitoring stations below the discharge point include one pool transect and various downstream riffle sites, extending 7.7 km downstream. A statistical summary of 12 years of electrofishing samples was conducted by comparing abundance trends in lower Vernon pool to the Vernon Dam tailrace. Kendall-Tau b correlation coefficients (Kendall, 1970) were computed for each individual RIS to assess whether or not abundances showed statistically significant increasing, stable or decreasing trends. Of the nine RIS in lower Vernon Pool, eight had decreasing trends (three were significant, $P < 0.05$); in the Vernon Dam tailrace, all nine RIS had decreasing trends (two were significant, $P < 0.05$) (Normandeau, 2006).

Designation of RIS in the 1978 316(a) demonstration featured eight fish species. Atlantic salmon were the representative species for the most thermally sensitive (Table 2). Smallmouth bass and walleye (*Sander vitreus*) were exotic species included as RIS due to their perceived sport value. In 1985, Vermont changed the designation of the Connecticut River from a warmwater to a coldwater fishery in its water quality standards. The 1990 demonstration deleted shortnose sturgeon (*Acipenser brevirostrum*) and replaced white sucker by white perch (*Morone americana*). The 2003 demonstration (Normandeau, 2003) added gizzard shad (*Dorosoma cepedianum*), America eel (*Anguilla rostrata*) and sea lamprey to the RIS (Table 2). The 2004 demonstration (Normandeau, 2004) replaced white perch with white sucker because white perch abundance, which had comprised up to 25 % of total fish abundance in the early 1980s (Downey et al., 1990), became severely diminished in the 1990s (< 1.4 %: Normandeau, 2004). Before this, however, white perch had declined from 6.3 % to 2.5 % of total catch downstream of Vernon Dam between 1971 (pre-operation) and 1981–1989 (post-operation), on the basis of percentage biomass. Presumably biomass had declined even further by 1991–2002 as had abundance, although it was not reported. American eel and sea lamprey, although valued components of the native community, also had very low abundance and were deleted as RIS along with gizzard shad at the request of VANR. The 2004 demonstration added fallfish (*Semotilus corporalis*, a native cyprinid) and largemouth bass (*Micropterus salmoides*, another exotic) to the RIS (Table 2). The 2004 demonstration deleted all consideration of biomass in electrofishing samples and reported only trends in numbers, unlike previous demonstrations. The numbers were also devoid of important age class differentiation. However, the 'ecological studies' report

Table 2. Representative Important Species selected in various 316(a) demonstrations for the Connecticut River in the vicinity of Vernon, Vermont. All UUILT values are those tabulated by Normandeau (2004) from various cited literature; * cited by Normandeau (2004) from Stanley & Trial (1995), who cited Elliott (1991); ** cited by Normandeau (2004) from Moss (1970) as the single value available from the literature.

1978	1990	2003	2004	UUILT
Atlantic salmon	Atlantic salmon	Atlantic salmon	Atlantic salmon	*27.8
American shad	American shad	American shad	American shad	**32.2
Shortnose sturgeon				
White sucker	White perch	White perch	White sucker	31.1
Spottail shiner	Spottail shiner	Spottail shiner	Spottail shiner	35.0
Walleye	Walleye	Walleye	Walleye	31.7
Yellow Perch	Yellow Perch	Yellow Perch	Yellow Perch	32.2
Smallmouth bass	Smallmouth bass	Smallmouth bass	Smallmouth bass	36.7
		Gizzard shad	Fallfish	32.2
		American eel	Largemouth bass	35.0
		Sea lamprey		

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produced for the same year (Normandeau, 2004) indicated relative percentage biomass in the electrofishing catches for 2003 downstream of Vernon Dam. These data show that from the 1971 pre-operational study (Webster-Martin, 1971), to the 1981–1989 period reported in the 1990 demonstration (Downey et al., 1990), to the 2003 'ecological study' (Normandeau (2004), based on electrofishing), smallmouth bass went from 6.1 %, to 13.0 %, to 52.1 % of the total biomass collected. Bluegill (*Lepomis macrochirus*) biomass went from 1.5 %, to 3.6 %, to 10.1 %. Yellow perch went from 21.0 %, to 4.0 % to 0.7 % of total biomass in these same periods. These data indicate what appears to be a shift in the community toward warmwater tolerant species. Total electrofishing effort in 2003 amounted to only 2.7 hours in the riffle habitat downstream of Vernon Dam. Despite all this evidence, the 2004 demonstration (Normandeau, 2004) claims for each RIS a finding of 'no appreciable harm' for the 1991–2002 period, which is claimed to be the existing baseline for consideration of additional summer thermal variances. The statistically significant, decreasing trend in white sucker was simply attributed to food web dynamics and not thermal discharge.

The currently proposed RIS for the Connecticut River at Vermont Yankee power station, increasingly emphasise warmwater fish. Normandeau (2004) identified smallmouth bass as a coolwater fish even though it is most often referred to in the literature as a warmwater fish. The UUILT (Ultimate Upper Incipient Lethal Temperature, see McCullough (1999)) reported by Normandeau (2004) for smallmouth bass (36.6 °C) was higher than that reported for largemouth bass (35 °C), which is always considered as a warmwater species. A UUILT of 36 °C is probably more realistic for largemouth bass (Fields et al., 1987). Normandeau (2004) also tabulated a UUILT for Atlantic salmon of 27.8 °C that is traced to Elliott (1991), although this value is applicable strictly to juvenile salmon. Adult salmonids have lower UUILT values than juveniles (McCullough 1999) and consequently require additional protection. Normandeau (2004) tabulated a high UUILT for American shad (32.5 °C) from Moss (1970) (although the details were actually given in Marcy et al. (1972)). The methodology described in Marcy et al. (1972) provided a thermal effect value (32.2 °C) more akin to a ULT (i.e. Upper Lethal Temperature; a temperature of near-instantaneous death), which would be greater even than a CTM (Critical Thermal Maximum, see Elliott, 1991) value and not representative of a UUILT. It indicated the temperature required to kill 100 % of a test group in 4 to 6 minutes when temperatures are raised rapidly from 24–28 °C to 32.5 °C, not the 50 % mortality in 7 days typical of UUILT studies. These misinterpretations of UUILT values lend inappropriate credence to the idea that coolwater fish fare well under warm conditions at any life stage and that there is less downside to temperatures favouring warmwater fishes, rather than the most sensitive. Brown trout (*Salmo trutta*) have UUILT values at least 3 °C lower than those of Atlantic salmon juveniles (Elliott, 1991). Despite the observed presence of low numbers of brown trout and rainbow trout in the mainstem in non-summer periods, these species were not designated as RIS to represent the coldwater fish community.

The 316(a) demonstration produced by Normandeau (2004) in support of a thermal variance is instructive in representing numerous pitfalls in the interpretation of EPA (1977a) guidance on conducting demonstration projects, to justify a conclusion of maintaining a Balanced Indigenous Community. The demonstration fails to assure its CWA goal by the following oversights in treating RIS, and the exclusive use of RIS, in general, as opposed to holistic community analysis:

1. The lack of valid UUILT values for certain species and substitution of values for UUILT that are inaccurate (Bogardus, 1981);
2. Mislabelling species as coolwater when they are warmwater which creates a more favourable image of 'balance';
3. Deleting species as RIS when their abundances become severely diminished;
4. Not considering species that are key native species in favour of exotic species;
5. Not considering key native species whose abundances are depleted;
6. Omitting native and introduced coldwater species as RIS that are present in low abundances.

Focusing analysis of the fish community only in terms of effects on a highly selective set of RIS does not represent the Balanced Indigenous Community, or the thermal sensitivity of its most sensitive members.

In the Connecticut River with Vermont Yankee's thermal variance permit history, the continual shifts in RIS composition seem to allow the causation of declines in species abundance to be obscured. Native species in extremely low numbers can be either excluded outright for statistical reasons or for simply not being officially listed in the Endangered Species Act (ESA) (Normandeau, 2004). In the case of Atlantic salmon, which was the nominal coldwater species representing the most thermally sensitive, their numbers were so low that this RIS member was not really used as a biotic indicator of power plant thermal pollution. American shad also had low abundance that was attributed to dam passage problems rather than heat discharge. Entergy and VANR did not opt for protection of eastern brook trout and its representation in the RIS of the mainstem Connecticut River, even though brook trout were collected seasonally in the mainstem in very low numbers near tributary mouths. Numerous other native fishes were collected in numbers as low as were the brook trout. Progressive shifts over time in RIS composition toward more heat tolerant species allowed the 'balance' to be more easily framed as stability in numbers of tolerant species. The Kendall tau b statistical analysis of abundance trends discarded the results from Atlantic salmon as not applicable because numbers were so low, and biomass was omitted from consideration in trends for all species. Consequently, there was no representative of the coldwater fish community. Atlantic salmon was the most sensitive species in the RIS (EPA, 1977a), and the Vermont water quality designation for the river is as a coldwater fishery. In terms of RIS performance, there was no formal method to define the 'balance' existing in the mainstem.

Over the period of operation of the VY plant, there were numerous changes in fish sampling gear, making long-term trends in fish abundance impossible to track. At each 316(a) demonstration, there was a conclusion that there was 'no prior appreciable harm' to the fish community. Even the most recent demonstration (Normandeau, 2004) claimed that on the basis of 12 years of electrofishing (a period of 'relatively consistent' sampling methodology, see Normandeau (2004)) a conclusion of no prior appreciable harm was reached. However, the trends demonstrated over this 12-year period, especially when represented primarily by cool- and warmwater fish species, may reveal little biotic response to a prior, continuous period of exposure to thermal effluent if the greatest declines in most sensitive species had already occurred before the sampling programme commenced.

In conclusion, the series of reports from a pre-operational report to the three successive 316(a) demonstrations, reveals a variety of critical failures in conducting these CWA Type I demonstrations. Given that the procedures used in the permit application process at Vermont Yankee are facilitated by the deference given to 316 rules and EPA guidance on Section 316, and there are so many other power plants with similarly long histories of successive variances and complex record of monitoring, it is likely to be representative of problems encountered nationally in implementing Section 316. Technical problems with the 316 demonstrations can be itemised as follows:

1. RIS can be changed at will by State approval, obscuring the fate of those species that decline, and relieving operators of the need to find causation.
2. Local increasing trends in abundance can easily be credited to improving water quality, even when the trend is in a warmwater tolerant fish.
3. Local decreasing trends can be blamed on regionally decreasing fish abundance trends, even though the cause for regional declines can also be regional cumulative thermal increases.
4. Substantial pre-operational baselines are rarely extensive or complete and facilitate being able to claim that existing conditions are really the baseline.
5. With every 316(a) demonstration that is submitted and variance approved, there is an accompanying assumption of no prior appreciable harm which then gives the appearance of a clean slate upon which to add increased thermal load.

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6. Changes in sampling techniques between pre-operational and post-operational periods for any point-source industry may be inevitable over a period of three decades, but this also provides a rationale for abandoning the earlier samples that provide connection to earlier population sizes. This detaches a trend analysis from the initial conditions prior to the pollution source. Note: the burden of proof is the applicant's responsibility, but when consistent methods are not available across the years, and with changes in monitoring companies and funding levels, the true changes in communities may be overlooked.
7. Given an inadequate pre-operational dataset, the available pre-operational data can then be joined with the early post-operational data, giving an 'early' operational snapshot. Any radical changes in populations that might occur at the outset of operations can then be further obscured.
8. Over time, effort in sampling may decline drastically, further increasing statistical variance.
9. With an extensive, complex history of demonstrations, and auxiliary studies that are not fully integrated into demonstrations, regulatory agencies such as EPA or State environmental agencies become incapable of devoting the time necessary to understand how existing water quality and biotic integrity have deviated from historic conditions.
10. Comparisons in electrofishing samples taken above versus below the thermal discharge, which could be instructive, become debatable when the habitats in these two locations are significantly different.

It is obvious that in a river with a lengthy history of human development such as the Connecticut River, the impact from the VY plant is but one among many. While the intent of the CWA is to restore the chemical, physical, and biological integrity of the river as a means to ensure a Balanced Indigenous Community, reducing community analysis to an examination of presence/absence and trends in a limited RIS does not address community sustainability and health in a meaningful way. Furthermore, representing the full community by selecting a limited number of species that span thermal requirements from cold to warm water does not aid in protecting the most sensitive species. Normandeau (2004), for instance, claims that selection of largemouth bass as an RIS provides good representation for brook trout despite the large differences in thermal tolerance. Allowed thermal increments to the river from the VY plant are calculated from the ambient baseline temperature established 5.6 km upstream of the plant. This framework is the worst possible system if restoring a natural community composition to the river, or preventing the regressive action of mounting cumulative effects, are the goals for water quality regulation. Non-point source thermal impacts must also be accounted for in a cumulative effects analysis, but this is not feasible in the Connecticut River currently because the four States involved have listed no tributaries or mainstem segments as impaired, which is a necessary precursor to trigger a TMDL.

Technical problems with community analysis and trends in the Clean Water Act

Numerous recommendations have been made to EPA concerning modifications to guidance on conducting 316(a) and 316(b) demonstrations (e.g. Dixon et al., 2003). Barnthouse et al. (2003) recommended a three-part analysis of Adverse Environmental Impact (AEI). This analysis comprised examination of the Balanced Indigenous Community, first simply as presence–absence data reflecting species richness trends (overall and area-specific richness). The second part evaluates trends in RIS abundance. The third part makes use of standard recruit–spawner analysis to evaluate trends in population productivity for RIS species, where mortality from power plant impacts would be considered in the same way as fishing mortalities. This implies that an overall human-caused mortality level is acceptable so long as the population can be considered sustainable and density-dependent compensation occurs (Super & Gordon, 2003). However, long-term habitat and water quality degradation can lead to significant shifts in long-term R/S (recruit per spawner) curves (Schaller et al., 1999; Petrosky et al., 2001). Application of this method would require a significant monitoring effort on numerous species, applied well in advance of plant operation. Major depression in the R/S function may result in significant reductions in population productivity followed by achievement of relative stability and higher extinction risk at a diminished abundance level. In terms of population risk, smaller populations that occupy historic habitats less fully would be at higher threat of extinction and can suffer greater loss of genetic and life history diversity (McElhany et al., 2000). Increase in urbanisation and level of water quality degradation are linked to reduction in salmonid viability (Regetz, 2002). These effects are not captured in simplistic BIC analysis of richness. While some authors recommend updating EPA 316 demonstration guidance by emphasising population and community level indices (van Winkle et al., 2003), others point out that the higher level indices do not offer great sensitivity in discriminating effects of thermal regime changes (Strange et al., 2003). Shifts in R/S indicators require rigorous long-term monitoring and do not offer either rapid assessment of progressive impact or a convenient means of employing precautionary principles to prevent damage before it occurs; examination of the potential impact on individuals (Strange et al., 2003) or groups of test organisms representing a population (such as in thermal effects laboratory testing) presents a much more direct approach for predicting and avoiding impact. Also, individual-based parameters provide greater statistical power in revealing treatment effects, due to their lower levels of variability compared to population-based parameters (Osenberg et al., 1994).

Application of the individual-based approach as advocated here is predicated upon the assumption that by fully protecting the most sensitive members of the community, the other members will also be capable of fully exploiting their historic habitats in their natural positions on the river continuum. Optimum growth rates of the sensitive species are a direct, biologically-based indicator of water quality suitability for sustaining healthy populations of fish sharing a thermal guild. Application of such indices of population health and community balance from individual-based thermal effects could have been used in the Connecticut River. For example, the increase in biomass of smallmouth bass from pre- to post-operation of the VY plant may have some association with the substantial increase in winter temperatures from power plant effluent. In 1981 the temperature ranges experienced for the entire month of January at Station 7 (upstream ambient control) and Station 3 (downstream of the VY plant) were 0 °C to 0.1 °C and 0 °C to 5.7 °C respectively (Aquatec, 1982). The increase in smallmouth bass dominance, increase in year-class strength, and reduction in population variability could easily be attributed to the much warmer winter temperatures permitted below VY (Horning & Pearson, 1973; Shuter & Post, 1990; Scheller et al., 1999). Likewise, summer growth temperatures near the optimum for warmwater species could also increase overall population status and increase predation rates on coldwater species, although simultaneous reduction in availability of preferred food sources could reduce growth rates (Shuter & Post, 1990). The decline in yellow perch biomass from pre-operation (1971) to post-operation (1981–89) may also be anticipated from their year-round residence in the mainstem during the warm summer period and their relatively low optimum growth temperature (22 °C: Kitchell et al., 1977). Given that summer temperatures already far exceed the yellow perch optimum growth temperature, a further increase of 0.56 °C would be counter-productive, but may not be sufficient itself to cause either a change in local species richness or a statistically detectable shift in R/S indicators (recommended by Barnthouse et al. (2003) as a key index).

Atlantic salmon adult upstream migration through the mainstem during the summer (CRASC, 1997) would be further threatened by mainstem temperature increases. The fact that adults are more thermally sensitive than juveniles should be taken into account. Adult American shad continue their upstream migration into July, but suffer increased bioenergetic stress and mortality at high temperatures that decrease the probability of iteroparity (Leonard & McCormick, 1999). A similar impact on Atlantic salmon can be assumed. Vermont ANR permitted a 1.7 °C temperature increase during 16 June–7 July, with the idea that it would protect shad if passage conditions at the dam below the VY discharge were limited to 24.8 °C (VEC, 2008). Without providing a complete thermal biological evaluation by species and life stage, suffice to say that there are many reasons not to support continued existing or new thermal variances on the Connecticut River in the interest of restoring biotic integrity and reducing sublethal impacts.

In addition to individual-based analysis of thermal effects on species and life stages, a holistic approach to monitoring long-term trends to ensure maintenance of a Balanced Indigenous Community should incorporate far more than presence/absence data and species richness. Declining trends should not necessarily be attributed uncritically to region-wide trends, nor should increasing trends be cited as proof of no impact from a point-source thermal discharge (Barnthouse et al., 2003) – improvement in water quality and habitat conditions at a basin scale may obscure the level of impairment produced by a point-source discharge. Likewise, Coutant (2000) suggests considering high juvenile mortality as part of a normative compensatory (density-dependent) response that can improve population age structure and overall health, despite the fact that human-induced mortality might be added to natural mortality loads. Coutant (2000) recommends judging the 'normative' ecosystem by community composition, trends in key functions, and shifts in range of variation. Although these criteria are important, they are not typically the most sensitive early warning indicators (Schindler, 1987; Minshall, 1996), and further, reducing a community to RIS trends is also insensitive to important species substitutions within guilds and dynamics of rare and sensitive species.

Long-term community trends evaluated by comprehensive multi-metric indices that incorporate abundances of the full species assemblages would be more diagnostic of community and population health. In addition, rather than simply reviewing RIS gross abundance trends, more comprehensive population characteristics should be analysed, such as age class distribution, length at age, seasonal fat content, growth rates relative to temperature and age, condition factor, etc (Minshall, 1996). Establishing large river IBI or other comprehensive multi-metric indices can help set biotic targets for recovery and be the means of evaluating recovery progress. But, most importantly, in the processes of both setting water quality standards and evaluating the potential effects of proposed thermal variances, cumulative effects of a new discharge in relation to all other human-induced thermal loads and the multiple life cycle thermal effects on the most thermally sensitive species, must be evaluated.

The application of all the above techniques prompts a central question: what are the goals for water quality criteria? The techniques listed suggest that by using historic data or their surrogates, we can reconstruct the historic baseline. If we knew what the longitudinal pattern of temperature was historically, would that set the ideal template for stream restoration and species protection? Given that stream heating tends to accompany development, is there some level of human allowance for thermal increase above the NTP (Natural Thermal Potential) that would provide a good target, permitting development, yet fully protecting the beneficial use? Or, are States content for their regulations not to consider non-point sources for mandatory compliance, and to permit an unlimited number of point sources to contribute heat loads to rivers and progressively increase temperatures without regard to baseline or species optima? The Clean Water Act uses 1975 as a baseline date for species distributions that are to establish, at a minimum, the beneficial use expectations for stream segments. If a use was documented by that date, the use is expected to be maintained unless a State undertakes a formal process to conduct a Use Attainability Analysis, followed by changing the use and standards. Other options are to adopt a site-specific standard or a limited-use standard.

Use of Maximum Weekly Average Temperature (MWAT) as a protective standard

Colorado adopts an MWAT criterion to protect salmonids

In the 1960s to 1970s, the recognition of thermal pollution from power plant effluent led to a flurry of research on thermal effects and publication of some key documents: the Columbia River Thermal Effects Study (EPA & NMFS, 1971) and the EPA (1973) report. The latter recommended application of the water temperature criterion MWAT as a standard for fish protection and this was recently adopted as a State standard by Colorado (Todd et al., 2008). MWAT can be distinguished as either a biologically- based value calculated using a formula provided by the NAS (National Academy of Sciences), or as a physical temperature statistic. Physically, MWAT is simply the largest 7-day running mean of average daily water temperatures, generally based upon an annual period of record, whereas the biological MWAT is calculated from the formula:

$$MWAT = OT + (UUILT - OT)/3$$

where OT is Optimal Temperature.

Colorado recently adopted a biologically-based MWAT for westslope cutthroat (*Oncorhynchus clarki lewisi*) that is 17.0 °C. This value is higher than the MWAT of 15.6 °C calculated from data on UUILT and optimum growth temperature from Bear et al. (2007) using the EPA formula. The summer rearing MWAT adopted by Colorado for rainbow trout (18.2 °C: Todd et al., 2008) is also considerably higher than the value of 16.8 °C calculated from Bear et al. (2007), but is slightly lower than the value of 19 °C recommended historically by EPA (1976). Colorado searched for a chronic temperature index that would indicate the threshold of sublethal effects, but instead of taking the upper end of optimum as a maximum allowable daily temperature as recommended by EPA (2003a), it assumed that the upper end of optimum, as defined by constant temperature laboratory growth, could be interpreted directly as an MWAT (Todd et al., 2008). In cases where an upper growth optimum was not available, this biologically based MWAT was calculated by the EPA formula (EPA, 1973, 1986) using UUILT and optimum growth (Todd et al., 2008). As noted above, Colorado adopted MWATs that were higher than those calculated based on Bear et al. (2007) and also employed an assumption that an optimum zone would be achieved by use of MWAT. The latter assumption is flawed, however, as even if the upper optimum temperature is achieved in terms of an average, i.e. in terms of a physical MWAT, the upper threshold would still be exceeded in terms of the daily maximum temperature. The Colorado rainbow trout MWAT of 18.2 °C, for example, could be met with a maximum and minimum temperature of 27 °C and 9.4 °C respectively. Colorado does have an acute temperature standard in addition to the MWAT, to limit the extent of maximum daily temperature excursions, however this does not protect against chronic effects; it is also calculated as a 2-hour average, rather than a more protective instantaneous maximum. As will be demonstrated below, MWAT is not a protective standard for a variety of salmonids.

The concept of an allowable level of impairment in temperature standards

When considering allowable temperature increases, it has been customary practice to make the judgment of acute effect on the basis of the acclimation temperature and the UILT (Upper Incipient Lethal Temperature) relative to this (see EPA, 1973). Then, a conventional 2 °C safety factor is applied in an effort to limit direct thermal death. Some States have adopted fixed allowable temperature increases per source without regard for acclimation temperature and UILT, and also make no account for cumulative temperature increases. Oregon has a standard that permits maximum temperature increases of 0.3 °C above the ambient from all cumulative sources when ambient is less than, or equal to, the 7DADM biologically-based criterion for the reach. This appears to be the most biologically conservative approach to regulation of NPS (non-point source) cumulative effects, but the benefits in regulating both PS (point source) and NPS will be realised primarily after a TMDL apportions load responsibilities. However, the approach embodied in the EPA (1973) report, by implying that any level of increase not producing critical thermal death by combinations of acclimation and exposure temperatures, with a safety factor, is acceptable, gives the mistaken impression that any temperature increment not causing near-term death of a localised group of individuals of a target species is likely to be permitted if it also meets the MWAT test for chronic effects. The interest in how large a diel fluctuation or an instantaneous temperature rise can be without causing short-term mortalities may

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have relevance only in cases of power plant effluents. But the EPA (1973) computations do not indicate the effects of repeated exposure to such fluctuations. Long-term or sublethal effects are apt to be of more profound significance. In addition, the EPA (1973) methodology is deficient in accounting for impacts to an entire population distributed throughout its range while applying its criteria on a watershed scale. It is more attuned to the predicted laboratory effects on a small test group of individuals. The current and accumulated impacts on an entire population that extends from upstream to downstream historical limits in its drainage basin, depend upon the past and current allowed increments of thermal discharge at all point sources as well as the combined effects of non-point sources. If large thermal increments are permitted at one site simply because there was a large enough difference between the ambient temperature and the temperature standard at the site of discharge, and this increment would not create a critical short-term impact (according to EPA (1973) methodology), the ability for downstream segments of the population to fully utilise their habitat will be impaired. Although heat can dissipate with distance downstream, this is generally not a rapid process (Beschta et al., 1987; ODEQ, 2004) and extensive lengths of mainstem can be compromised in quality. Additional thermal loads downstream can prevent thermal recovery. Loss of habitat quality can be equated to reduction in population fitness, reflected in lower abundance and productivity, loss of life history forms, and restriction in the total extent and diversity of habitat types utilised.

The EPA (1973) concluded that by use of MWAT as a temperature criterion, growth rates would be limited to ≥ 80 % of optimum growth rate. They stated that this appeared acceptable, despite there being no quantitative studies available upon which to define level of impairment. More recently, Sullivan et al. (2000) developed a quantitative risk assessment methodology that employs fish bioenergetics (i.e. use of the Wisconsin growth model, Hanson et al., 1997) to predict growth rate and direct thermal mortality. This risk-based approach assumed that a 10 % reduction in growth rate would be acceptable in not significantly affecting the population. The growth model was then used to determine what temperature conditions would produce these levels of growth impact. They acknowledge that a 20 % reduction in growth potential is likely to affect juvenile size at emigration, which could affect marine survival. They modelled growth rate based on assumed food supply levels under field conditions and determined 7DADM temperatures of 16.5 °C, 22.5 °C and 30 °C corresponding to 10 %, 20 % and 40 % reduction from maximum growth, respectively, for coho salmon (*Oncorhynchus kisutch*). The 40 % growth limitation temperature was identical to the temperature producing the geographic limit (Eaton et al., 1995a, as cited by Sullivan et al., 2000). This is evidence that growth reduction does not need to be 100 % for a species distribution to reach extinction.

Sullivan et al. (2000) stated that the temperature standards developed in the past 25 years have been integrated from surveys of scientific literature, but have not been developed on a quantitative basis that lends itself to hypothesis generation. Assuming that growth rate impairment alone is a sufficient basis for generating a standard, that the level of impairment is deemed acceptable, and that the Wisconsin model is technically adequate, their framework presents a means to adjust the level of risk based on the combination of growth rate reduction and direct thermal death. Despite the quantitative basis for this analysis, however, there are questions about the computational adequacy of the Wisconsin model (e.g. Ney, 1993). This model has its own set of assumptions and unverified physiological issues. In addition, there are legal concerns about allowable levels of degradation.

Other factors listed by Sullivan et al. (2000) as clouding the quantitative risk-based approach, were acclimation temperatures, daily temperature fluctuations, food availability, management impacts on food availability, interaction of temperature and other pollutants, uncertainties concerning most sensitive species, and ecological interactions among fish species. UILT values established under constant temperature exposures have typically been used in establishing temperature standards. However, response to fluctuating temperatures and the ability of fish to recover during low temperature parts of the diel temperature cycle inject uncertainty into the level of protection. For example, Johnstone & Rahel (2003) reported a UILT of 24.2 °C for Bonneville cutthroat trout (*Oncorhynchus clarki utah*). However, in a fluctuating temperature regime of 16 °C to 26 °C (with a 6-h exposure to temperatures greater than or equal to the UILT) these trout had no mortality in a 7 d period. Increasing the cycle to 18 °C to 28 °C caused a significant mortality. These results indicate a limited ability of cutthroat to recover in a fluctuating regime after exposure to peak temperatures, but the authors noted evidence of impaired growth that could lead to population effects under the moderate fluctuating regime. Golden (1975, 1976) found very similar responses to peak temperatures in a cyclic regime for coastal cutthroat trout (*Oncorhynchus clarki clarki*). He found that mortality occurred with as little as 8 hours constant exposure at 27.3 °C. Even more interesting is that in a fluctuating temperature regime of 13 °C to 27.5 °C, with a prior acclimation in a 13 °C to 23 °C cyclic regime, 50 % mortality was produced in only 1.25 cycles. When the diel exposure cycle was increased to 13 °C to 28 °C, only 0.75 diel cycles were required to produce 100 % mortality. The UUILT was 25.0 °C for this species (Golden, 1975), similar to most reports for salmonids as well as that of Johnstone & Rahel (2003). The study by Golden (1975), however, indicated that mortality was a cumulative effect of multiple exposures to a cycle.

Sullivan et al. (2000) recommended growth modelling under expected field food availability as a means of calculating biological risk. They noted, as did Johnstone & Rahel (2003), that sublethal effects such as feeding impairment, behavioural avoidance and refuge seeking, as well as competitive interactions, could cause additional, unmodelled growth stress. This would suggest that a 10 % growth impact level is not sufficient to protect the population. Sullivan et al. (2000) estimated that the 7DADM temperatures that would result in a ≥ 10 % growth loss for coho are ≤ 13 °C and ≥ 16.5 °C. The 7DADM temperatures that would result in a ≥ 20 % growth loss for coho are ≤ 9 °C and ≥ 22.8 °C. These 7DADM temperatures that produce a 10 % growth loss are comparable to a physical MWAT of ≤ 13 °C and ≥ 14.5 °C (Sullivan et al., 2000). However, the biological MWAT for coho was reported to be 18.3 °C. If a physical MWAT of 14.5 °C causes a 10 % reduction in growth, one has to question what an MWAT of 18.3 °C, calculated using the EPA (1973) formula, hopes to achieve. Various case studies can show more fully the failure of MWAT to protect salmonids.

Case study 1: rainbow trout (Oncorhynchus mykiss)

The use of MWAT can be criticised in its application both at an individual stream reach level and at an entire watershed scale. At the scale of a stream reach, the use of MWAT as a temperature standard has an impact of greater than 20 % on growth rates and production, as illustrated by the work of Hokanson et al. (1977) and Jobling (1997). If applied at the level of the watershed, the potential for impact to the entire community is magnified further due to the methods that are most likely to be used for implementing the standard. Much of the confusion about, and resistance to, protective standards comes from a lack of understanding of their application at watershed scales. The concept of optimum temperature sounds either too protective or too unrealistic to some people. They argue that throughout the range of a fish species, individuals do not find optimum conditions everywhere. Temperatures typically increase as elevation decreases downstream, and species tend to have their lower distribution limit constrained by high temperature and increasing levels of competition and predation from species more suited to warmer waters. This would apply equally to the historic range of a species, with an elevation along the river continuum above which temperatures naturally do not exceed the optimum and downstream of which the temperature maxima do exceed the optimum but there is no means, short of refrigeration (or release of cold water from impoundments), to cool the water. So, because under even historical conditions a biological optimum was not the physical limit to the geographic range of the species in the stream system, the proponents of this line of argument reject the entire concept, without noting that means exist

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for establishing temperature targets on a basin scale in a way that maintains the entire thermal profile supporting all life history stages of the species (see EPA, 2003a; Poole et al., 2004). These same proponents also argue that by using sensitive species' optimum temperatures, other species in the thermal guild are not being protected. Under historic thermal regimes existing at a basin scale, all historic native species realised their ability to co-exist and use habitat extensively. Historic coldwater habitats were not limited only to headwater areas as is presently common but were spatially extensive. Temperature increases in basins, and shortening of suitable thermal habitats, are the primary culprits in favouring species with warmwater needs as opposed to coldwater needs.

Hokanson et al. (1977) advised caution in using short-term exposure experiments to calculate long-term exposures, such as with MWAT. They reported for *O. mykiss* that, given a physiological optimum of 16 °C to 18 °C and a UILT of 25.6 °C with a 16 °C acclimation, one would calculate an MWAT of 19 °C and a maximum temperature (applying the 2 °C safety factor of EPA, 1973) of 24 °C for short-term exposure. (Note that Bear et al. (2007) reported a UUILT for rainbow trout of 24.3 °C.) Measurement of rainbow trout growth showed that at a fluctuating temperature of 22 °C ± 3.8 °C, specific growth rate was zero and the mortality rate during the first 7d was 42.8 % d^{-1}. For experiments with a fluctuating regime within the optimum range 15.5 °C–17.3 °C, average specific mortality was 0.36 % d^{-1}. Combining data on specific growth and mortality rates, the authors were able to predict yield for a hypothetical population under various temperature regimes. A rainbow trout population would exhibit zero increase in biomass (maintenance) over a 40-d period at a constant temperature of 23 °C, as well as at a fluctuating temperature with a mean of 21 °C ± 3.8 °C, because growth balances mortality in both thermal regimes. Several sources report temperatures of 21 °C to 23 °C as the upper limit of rainbow trout distribution in the field (Hokanson et al., 1977), and a maximum of approximately 22 °C to 24 °C for salmonid distribution (McCullough, 1999).

With this laboratory information and corroborating field information, Hokanson et al. (1977) recommended a mean weekly temperature of 17 °C ± 2 °C for rainbow trout so that maximum yield is not reduced more than 27 % under normal fluctuating temperature regimes. Production was shown to be substantially reduced at maximum temperatures just above the physiological optimum. This paper has great significance. It was published five years after the National Academy of Sciences recommended the use of MWAT to establish prolonged exposure temperatures. The NAS acknowledged that growth rate should be expressed as net biomass gain or net growth. Yield is that portion of the population available for harvest and typically linked to production (Nielsen & Richardson, 1996); the remainder is lost as mortality, which can be substantial if temperatures are high. Also, if temperatures are high, much of the energy assimilated from food is lost as excessive metabolism. If the biological MWAT is 19 °C, and yield is reduced 27 % from maximum at a mean weekly temperature of 17 °C ± 2 °C (where 17 °C is the constant temperature physiological optimum), it is obvious that MWAT is not protective.

Jones et al. (2006) reported rainbow trout distribution in northeastern Georgia streams in relation to temperature, which was attributed to a combination of factors such as riparian buffer cover, stream length, and elevation. Probability of occurrence of young trout (comprising rainbow trout and brown trout < 15 cm), reflecting reproductive success, was related to water temperatures (7DADM). Water temperatures < 19.5 °C, 19.5 °C to 21.5 °C, and > 21.5 °C were related to probabilities of occurrence of > 90 %, 50 % to 90 %, and 0 % to 50 %, respectively. Because the probabilities of occurrence were those of young trout, which tend to be more tolerant of warm water than for adults of their species, the probabilities would express an overly optimistic view of salmonid distribution. It is clear from this study that a biological MWAT of 19 °C would not be protective, given the expected associated maximum temperatures. Rather, depending upon the relationship between 7DADM and MWAT, it is likely that the probability of occurrence would be in the 0 % to 50 % range.

Donato (2002) studied the temperature statistics of 183 'relatively undisturbed' stream sites throughout the Salmon and Clearwater basins in central Idaho from 15 July–10 September, 2000. Hourly temperature measures were converted to statistics such as MDMT (Maximum Daily Maximum Temperature), MWAT (Maximum Weekly Average Temperature), and percentage of hourly readings during the 58-day period that exceeded 22 °C. From the data provided in Donato (2002), a regression of MDMT on MWAT was calculated as MDMT = 0.9236 MWAT + 5.509 (R2 = 0.8295, n = 183). For an MWAT of 19 °C (corresponding to the biological MWAT of rainbow trout, *O. mykiss*), the predicted value of MDMT is 23.1 °C ± 3.0 °C (95 % C.I.). A plot of percentage of all days > 22 °C on MWAT shows that at 19 °C MWAT there were between 0 % and 5 % of all hours in the 58-day period having temperatures > 22 °C. There was a straight-line increase in percentage hours exceeding 22 °C as MWAT increased from 19 °C to 25 °C. At an MWAT of 25 °C, the percentage exceeding 22 °C was 45 %. The biological MWAT for rainbow trout then carries with it, even in 'relatively undisturbed' streams, a maximum summer temperature of 23.1 °C to 26.1 °C (based on 95 % confidence limits) and up to 5 % of the 58-day period exceeding 22 °C. Of course, even a one week period in this temperature range may be sufficient to limit the distribution of rainbow trout (or steelhead trout, *O. mykiss*). The mean MWAT for 99 second and third order streams of the Salmon and Clearwater was 15.4 °C, which corresponded to an MWMT (Maximum Weekly Maximum Temperature) of 18.7 °C (Table 3). Based on the thermal classification applied by Jones et al. (2006), these streams would have high probabilities of *O. mykiss* occurrence. One would have to question, though, whether these minimally disturbed streams in Idaho actually represent NTP (Natural Thermal Potential) conditions given that even road densities of 2 and 4 km·km^{-2} tend to increase the MWAT of streams by 1.25 °C and 3.25 °C, respectively (Nelitz et al., 2007).

Additional support for the thermal limits to rainbow trout distribution comes from Wehrly et al. (2007). For Michigan and Wisconsin trout streams they showed that the 95th percentile maximum daily mean temperature at sites where rainbow trout were found was about 23.3 °C as a 7 d exposure, while the 95th percentile maximum daily maximum temperature was about 25.5 °C. In this study and numerous others reviewed, the maximum temperatures associated with distribution limits of rainbow, brook, and brown trout ranged from 24 °C to 26.3 °C, essentially equivalent to the UUILTs of these species (McCullough, 1999). If the biological MWAT of rainbow trout is 19 °C, incurring a maximum temperature up to 26 °C, this MWAT is not protective. The point of the predictions of 95th percentile distribution temperatures is to determine a value just below the upper threshold beyond which a species cannot exist. Although this limit and its correlates are not good guidance for biologically-based standards for salmonid temperature optima (or suitable conditions), there is reason to believe that these values could be even lower. The extensive glacial till in Michigan and Wisconsin provides significant groundwater to streams in the region, which could provide frequent cold refugia (Gaffield et al., 2007) that could be up to 5 °C to 7.5 °C cooler than ambient temperatures (Picard et al., 2003), altering conclusions based on ambient temperatures.

It is often argued that studies such as that of Donato (2002) reveal that even minimally disturbed streams have high MWAT or MDMT

Table 3. Temperature statistics for 183 streams of the Salmon and Clearwater river basins, central Idaho (Donato, 2002).

	Stream temperature metric		
	MDMT	MWMT	MWAT
	2nd and 3rd order streams		
No. of sites	99	98	99
Maximum (°C)	29.5	28.9	23.4
Mean (°C)	19.7	18.7	15.4
	4th and 5th order streams		
No. of sites	73	73	73
Maximum (°C)	31.5	30.7	25.4
Mean (°C)	22.6	21.7	18.3
	6th and 7th order streams		
No. of sites	11	11	11
Maximum (°C)	27.7	27.1	25.0
Mean (°C)	24.3	23.7	22.5

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values, so it would not be feasible for more developed streams to provide high levels of support for coldwater fishes. Donato (2002) states that her study in the Clearwater and Salmon drainages of Idaho was based on minimally disturbed or undisturbed streams, not significantly affected by streambed disturbance, diversion, or riparian vegetation removal. Also, this study was based on data from a single summer having high air temperatures and low streamflows. Despite the contention that with the above caveats the statistical model represents NTP conditions, the database includes numerous significantly disturbed streams, which would significantly increase the mean and maximum values for their respective stream orders. GIS analysis of road densities in selected watersheds within this USGS (US Geological Survey) database reveals that road densities are at least as high as 3.45 km·km^{-2} (Table 4). An extensive study conducted in the Columbia River basin showed that of watersheds considered to be of low forest integrity, the majority had what was considered to be at least moderate road density (> 0.17 km·km^{-2}) (Gucinski et al., 2001). The USGS database includes three sample points in the Lemhi drainage basin. The upper Lemhi watershed (upstream of Lemhi, Idaho) has a drainage density of 1.41 km·km^{-2}. In addition, the entire watershed has 2950 points of irrigation diversion, 191 stream alteration permits, and has 37 % of the non-riparian watershed area affected by livestock grazing (NPCC, 2004a). Cattle grazing affects the mainstem river, causing riparian vegetation loss, bank instability, and sedimentation. Irrigation return flows, reduced summer flows, impaired riparian condition, and high levels of sedimentation from roads, grazing, and floodplain development cumulatively produce elevated stream temperatures. Despite what might be considered moderate road densities, overall watershed impacts lead to significant deviation from NTP conditions (NPCC, 2004a, b).

A follow-up study of 34 'least-disturbed' streams in the Salmon River basin of central Idaho (Ott & Maret, 2003) attempted to draw conclusions such as: streams in Idaho frequently exceed State temperature standards even when they are found in high quality watersheds; also, these streams continue to support their beneficial uses. No evidence was provided to indicate how well beneficial uses were protected other than to imply maintaining species presence. This study included streams such as Valley Creek, which has a combination of impairments such as a history of livestock grazing, water diversions, channel alterations, and sedimentation (NOAA, 2004). The lower Valley Creek site has an average summer diel temperature variation of 11.1 °C, a maximum diel variation of 18.7 °C, an MWMT of 22.6 °C, and an MWAT of 17.1 °C. Under these conditions, there is a large difference between MWMT and MWAT (i.e. 5.5 °C). Even Sulphur Creek, which is found in a wilderness area and has no sign of human habitation, has water diversions and is recovering from a history of livestock grazing (NOAA, 2004). In summary, statistical models that purport to represent NTP conditions may hide a multitude of anthropogenic impacts that lead

Table 4. Calculation of road densities within selected Idaho watersheds considered by Donato (2002) to be reference watersheds for measurement of stream temperature reflecting natural potential.

Watershed	Stream order	Road length (km)	Watershed area (km^2)	Road density (km/km^2)	Stream length (km)	Scale
Jim Ford	4	326.2	261.1	1.25	56.6	100k
Lawyer	4	528.6	553.7	0.95	83.0	100k
Lemhi (entire)	5	1611.7	3248.4	0.50	787.2	100k
Lemhi (upper)	4	1151.1	816.0	1.41	152.2	100k
Pete King	3	246.2	71.3	3.45	269.5	24k
Potlatch, East Fork	3	237.3	160.7	1.48	416.2	24k
Skookumchuck	3	231.9	84.9	2.73	112.0	100k
Whitebird, North Fork	3	256.5	85.4	3.00	95.6	100k
Whitebird, South Fork	3	255.8	92.8	2.76	117.5	100k

to improper conclusions that high MWATs are the norm for streams in the heart of coldwater fish native ranges.

Rainbow trout Realised Thermal Niche

The thermal niches of 16 species of aquatic vertebrates in a total of 199 first- to third-order streams throughout Oregon were surveyed using single-pass electrofishing to assess relative abundances of species in relation to the 7DADM water temperature (Huff et al., 2005). Two thermal niche indices were calculated for each species. The Realised Niche Centre (RNC) for each species is the mean temperature for the species in the stream reaches where it was found, weighted by the species relative abundance. The Realised Niche Width (RNW) is equal to the RNC plus or minus one weighted standard deviation, where the deviation of water temperature from the RNC is weighted by the species' relative abundance. In line with thermal niche theory, both indices are weighted by species' relative abundance as species tend to be most abundant near where their optimum temperature is expressed.

Thermal niches of the 16 aquatic species co-occurring throughout Oregon were defined for five geographic regions – the North Coastal basins, South Coastal, Coast Range ecoregion, Cascades ecoregion and Blue Mountains ecoregion – extending from coastal to central and northeastern Oregon. For rainbow trout the RNCs (in terms of 7DADM temperatures) ranged from 14.0 °C to 19.5 °C, depending upon the region, whereas the RNWs (upper limit of the range, 7DADM temperatures) ranged from 16.0 °C to 22.4 °C. Mean RNC and mean (upper limit) RNW were 16.0 °C and 19.7 °C, respectively, for all regions combined (Huff et al., 2005).

In the Coast Range basins, the RNC and upper RNW limits were 14.9 °C and 17.6 °C respectively. This indicates that one standard deviation is 2.7 °C for Coast Range populations and hence, assuming a normal distribution, only about 15.8 % of the population would occur at temperatures greater than 17.6 °C (the upper RNW limit, as a 7DADM value). The biological MWAT for rainbow trout obtained from EPA (1986, i.e. the 'Gold Book') is 19.0 °C. Converting this MWAT to a 7DADM value can be approximated using various sources. Temperature index conversion equations for small Idaho streams from Hillman & Essig (1998) provide a 7DADM value of 22.3 °C for a biological MWAT of 19.0 °C, with Dunham et al. (2001) indicating a value of 22.2 °C. Such temperatures would be 2.7 standard deviations above the RNC, meaning that an extremely small percentage of the population would be found at temperatures above the MWAT. For the Tucannon River, Washington, at rkm 56 (approximately the mid-point of the river length), an 11-year temperature database (obtained from Washington Department of Fish and Wildlife) yielded a regression equation of 7DADM = 1.0001 MWAT + 2.0997. This river site, which is probably larger than the streams reported by Hillman & Essig (1998) and Dunham et al. (2001), had a 7DADM of 21.1 °C, or close to 2 °C greater than the MWAT value, instead of approximately 3 °C.

It might be assumed that in some locales rainbow trout would be either fitter in warmer streams, or might have varied levels of competition due to a changing fish community composition. The data from Huff et al. (2005) for the Blue Mountains (Oregon) ecoregion indicate a rainbow trout RNC of 19.5 °C and a RNW of 22.3 °C (both as a 7DADM). This reveals that one standard deviation in the thermal niche is 2.9 °C, and hence one would expect only 15.8 % of the population to be found at temperatures greater than 22.3 °C, which corresponds closely to the MWAT for rainbow trout of 19.0 °C using the conversions of Hillman & Essig (1998) and Dunham et al. (2001). So, even in the Blue Mountains, a very small portion of the rainbow trout population is found at temperatures greater than the upper RNW limit, which corresponds closely to the MWAT.

The distribution of a combination of rainbow and brown trout in Montana streams revealed an upper distribution limit of approximately 20 °C MWMT (=7DADM) for all sites with these species, whereas the upper field limit for westslope cutthroat trout was about 16 °C (Sloat et al., 2002). As shown previously for all other Oregon sites studied, application of the Gold Book MWAT would be highly disadvantageous. Further, if an attempt is made to use redband trout UUILT values in lieu of standard rainbow trout UUILT values, an even

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higher biological MWAT would result. This site-specific standard, using the EPA Gold Book convention, would produce an even greater divergence between the RNW observed in the field and MWAT values taken as protective of summertime growth.

Dunham et al. (2005) conducted a statistical analysis of 1252 temperature records from streams in the Pacific Northwest and the northern Great Basin to derive correlations between important temperature metrics. Statistics were computed for summer (15 July–15 September), which was considered to be the period in which maximum temperatures are exhibited. This study revealed that the greater the average daily temperature range during the summer, the weaker the corrrelation between MWAT and MWMT (i.e. 7DADM; Fig. 1). Similar conclusions can be drawn from the data accompanying the report by Ott & Maret (2003) for 183 stream sites in the Salmon River basin, Idaho. If one were to take the EPA (1986) MWAT value for protection of rainbow trout (19 °C), it would correspond to an MWMT of 19.9 °C when there is no more than a 2 °C daily variation in summertime water temperature, but an MWMT of 22.3 °C where the average daily range is between 4 °C and 6 °C, and an MWMT of 28.0 °C where the average daily range is > 12 °C (Dunham et al., 2005; Fig. 1). This study shows clearly that the level of protection afforded by MWAT as a biological standard in the field varies greatly with the average daily temperature range. Those streams having a large daily temperature range would tend to be those that are lacking riparian cover or have highly damaged channel structure (Poole & Berman, 2001). Consequently, in streams that are most severely thermally affected, the coldwater biota would suffer the most by use of a biological MWAT as a temperature standard because of the inability of this index to adequately reflect the maxima experienced by the biota. It might be countered that this merely indicates the need to have a standard for acute as well as chronic temperatures. However, this raises the question of how high a MWMT limit should be. Should it be equal to the upper RNW limit? How would this serve to protect salmonids if applied in the field? And what biological objective would MWAT serve?

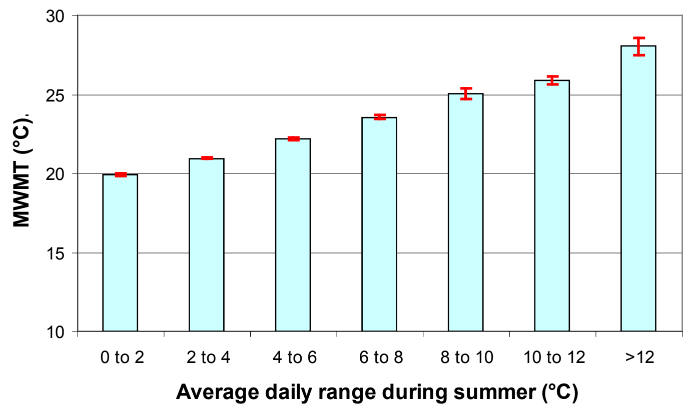

Figure 1. The variation of MWMT for an MWAT value of 19 °C as average daily water temperature range varies during the summer. Derived from data in Dunham et al. (2005).

Case study 2: Lahontan cutthroat trout (*Oncorhynchus clarki henshawi*)

Eaton et al. (1995a) created a large database that matched fish distribution in the field with average daily water temperature as a means to estimate field thermal tolerance limits for over 300 fish species. Fish collection records at specific geographic locations for a specific year were then associated with USGS water temperature data available for the same year that were found within 7.5 km (15 km according to Eaton et al., 1995b). Thermal tolerance limits for individual species were then calculated from a database of at least 1000 fish/temperature (F/T) data pairs of MWAT temperatures. From this database, the 95th percentile temperature was taken to represent the field thermal limit. For cutthroat trout, this limit was 23.2 °C.

Dunham et al. (2001) associated the distribution of Lahontan cutthroat trout (*O. clarki henshawi*) with water temperatures found within 0.3 km of fish sample locations, a much more precise spatial association even though their sample sizes were much smaller (< 100). They found that the average MWAT associated with the distribution limit for all sample streams was 17.5 °C; the maximum MWAT limiting distribution was 20.9 °C. It is likely that the discrepancy in MWAT temperatures between Eaton et al. (1995a) and Dunham et al. (2001) could be caused by a combination of factors. The larger sample size in the study by Eaton et al. (1995a) could have included more F/T associations at the extremes. MWAT temperatures found within 7.5 km (or 15 km) could easily be downstream of fish field distribution limits, or skew the temperatures due to the spatial variation in temperatures typical of stream systems (Boyd & Kasper, 2003). Also, the temporal F/T matching appears to have a resolution of about one week, within which time fish may temporarily move upstream to avoid adverse temperatures.

Dunham et al. (2003b) reported that the distribution of Lahontan cutthroat was associated with maximum daily temperatures of 18.9 °C to 28.5 °C, depending upon the stream. Distribution was predominantly linked to temperatures ≤ 26 °C. Sublethal stress induction has been reported at temperatures > 22 °C (Dickerson & Vinyard, 1999; Meeuwig, 2000) in the laboratory under optimal conditions. Consequently, Dunham et al. (2001) caution that F/T associations, even at their 0.3 km resolution, may not well-represent the exposure duration. A study of Lahontan cutthroat distribution in the Willow Creek drainage of southeastern Oregon (Talabere, 2002) showed that fish aged 1, 2 and 3 years all showed a marked decline in density with increasing temperature in free-flowing stream reaches. A 7DADM temperature of 22 °C was associated with a steep rate of decline in fish density. Density was nearly zero at a 7DADM of 24 °C. At the site corresponding to the near absence of Lahontan cutthroat (where the 7DADM was 24 °C), the mean hours/d above 22 °C, 24 °C and 26 °C was 5, 2 and < 0.5 hours respectively. In the reaches with 7DADM temperatures of 21 °C to 22 °C, mean hours/day exceeding 22 °C and 24 °C averaged about 1.5 and 0.5 h d^{-1} respectively. At these locations fish density was dramatically greater.

Other cutthroat subspecies have lower maximum field distribution limits. Westslope cutthroat (*O. clarki lewisi*) occurred in Montana streams only where maximum daily temperatures were < 16 °C and mean daily temperatures were typically < 12 °C (Sloat et al., 2002). However, the centre of distribution for all westslope cutthroat sites had daily maximum temperatures of 10 °C to 12 °C. The UUILT and optimum growth temperatures for westslope cutthroat are 24.7 °C and 13.6 °C respectively (Bear et al., 2005). Applying the EPA (1973) formula, the biological MWAT would be 17.3 °C. Although this biological MWAT is calculated for westslope cutthroat, it is nearly equal to the 17.5 °C physical MWAT associated with average field distribution limits of Lahontan cutthroat (Dunham et al., 2001). For Lahontan cutthroat, growth rates ($g \cdot g^{-1} \cdot d^{-1}$) were greatest at a constant temperature of 12 °C in comparisons of constant temperatures of 12 °C, 18 °C and 24 °C vs. fluctuating temperatures of 15 °C to 21 °C (mean of 17.5 °C) and 12 °C to 24 °C (mean of 17.2 °C) (Meeuwig et al., 2004). Growth rates under these temperature regimes were approximately 0.019, 0.017, -0.002, 0.013 and 0.008 $g \cdot g^{-1} \cdot d^{-1}$ respectively, based on a 20 g initial mass. The maximum growth rate (0.019 $g \cdot g^{-1} \cdot d^{-1}$) was comparable to the growth rate model for coho by Sullivan et al. (2000) at approximately 70 % ration. The study by Meeuwig et al. (2004) revealed that cyclic diel variation with 24 °C as a peak daily temperature produced a growth rate that was less than half that of the two coolest constant temperatures tested (12 °C and 18 °C). Also, the mean temperature of both cyclic regimes was nearly identical to the biological MWAT calculated for westslope cutthroat (17.3 °C). This MWAT would produce a 31 % to 58 % reduction in growth rate from optimum levels, given the cyclic regimes studied. These regimes represent the biological MWAT of 17.5 °C ± 2.5 °C and ± 5.5 °C – levels of fluctuation well within the range expected for many streams.

These results were similar to those of Dickerson & Vinyard (1999) who found that Lahontan cutthroat had highest growth rates at 13 °C constant temperature, although not significantly greater than at 20 °C or 22 °C. At a constant temperature of 24 °C, growth rates were barely above zero. Growth rates (7-d exposures) at 23 °C versus a fluctuating regime of 20 °C to 26 °C (mean of 23 °C) were not significantly different and were about 40 % of those at 13 °C to 20 °C. This indicates that even brief exposure in a cyclic regime to temperatures above those capable of producing high growth does not confer

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a growth advantage. Fish held at a constant 26 °C (7-d exposure) had only 64 % survival, indicating that the UUILT would be a value probably slightly less than this, similar to westslope cutthroat and most other salmonids.

Case study 3: coho salmon (*Oncorhynchus kisutch*)

It is also possible to illustrate the failure of MWAT to protect coho salmon. The growth optimum for coho fed at maximum ration (i.e. to satiation) is 15.0 °C (Edsall et al., 1999). The UUILT for coho is 25.0 °C (Brett, 1952). With the EPA (1973) formula, the biological MWAT is calculated as 18.3 °C. In tributaries of the Mattole River, California, the relationship between the physical MWAT and MWMT is MWMT = 1.363 MWAT - 4.464 (calculated from Welsh et al., 2001). Hence if MWAT is 18.3 °C, MWMT is 20.5 °C. The MWMT is identical to the 7DADM, a statistic used by EPA Region 10 (Pacific Northwest) (EPA, 2003a). The regression between MWAT and MWMT developed by Hillman & Essig (1998) for bull trout streams in Idaho predicts that if MWAT is 18.3 °C, MWMT is 21.5 °C, a result comparable to that derived using data from Welsh et al. (2001) for the Mattole River, California. In tributaries of the Mattole River, when MWMT was greater than 18.1 °C, coho were absent. All streams with MWMT < 16.3 °C had coho. This means that if the biological MWAT of 18.3 °C is used as a standard, the corresponding physical MWMT that would accompany a physical MWAT of this value would eliminate the species. It is clear that MWAT is not a protective standard, either at a site for individual coho protection or at a basin scale for population protection. Indeed, concerns over the sub-lethal and chronic effects of exposure to MWAT temperatures in salmon of the Pacific Northwest, led the EPA (2003a) to recommend a summertime (core juvenile rearing) 7DADM of 16 °C for this region, rather than the MWAT derived from the 'Gold Book' formula, to protect all coldwater salmon and trout other than bull trout (colder optimum growth temperature) and Lahontan cutthroat (warmer optimum).

Madej et al. (2006) discovered that coho in Redwood Creek, California, were confined to the lower 20-km of the river where MWMT and MWAT remained below approximately 22 °C and 20 °C, respectively, due to the transition of the mainstem into the coastal redwood fog belt. Snorkel surveys conducted in 2003, however, revealed only 29 young-of-year and 12 age 1+ coho. These fish were predominantly found in side pools separated from the main channel by gravel bars that trapped cold water from seeps or upwelling groundwater. Historically, the middle reaches of Redwood Creek were excellent habitat for coho production, but after extensive logging starting in the 1950s, the conifer cover was reduced from 86 % along the mainstem to only 15 % by 1997. The lower 20 km reach with old-growth riparian forest remained intact and accounts for retention of coho production in only 20 % of the historic coho range in Redwood Creek. Absence of coho from the middle reaches was associated with MWAT temperatures greater than 20 °C and (based on data from thermal infrared (TIR) remote sensing) with surface water temperatures greater than 24 °C. This study points out that surface temperatures are greater than temperatures measured by instream thermistors, and the fish tend to be located in the coldest thermal refugia available. The extremely weak population size and high reliance on cold refugia make linkage of coho presence to average river temperatures for entire stream zones risky. Madej et al. (2006) found that in addition to the MWMT and MWAT indices, the transition from minimal coho presence to complete absence was also linked to the longest number of consecutive hours of temperatures > 18 °C.

The EPA (2003a) standard for salmon of 16.0 °C (measured as a 7DADM) in core rearing areas is similar to the MWMT of 16.3 °C cited by Welsh et al. (2001), which those authors found to be a critical threshold for the presence of coho. With such a small difference between the growth optimum of 15 °C and the maximum critical temperature for field distribution, it is clear that regulatory procedures would be needed to ensure that temperatures are not raised much beyond the growth optimum.

Case study 4: bull trout (*Salvelinus confluentus*)

Another clear illustration of the failure of MWAT to protect fish species can be seen by examining bull trout temperature data. This example uses recent studies that provide data for growth optimum (GO), UUILT, field-based distribution limits in terms of maximum daily temperature, calculated biological MWAT (i.e. based on the NAS method), a regression of field-based MDMT vs. MWAT in the field, and another field study that related bull trout abundance and age diversity vs. MWAT. This set of data clearly indicates that the biological MWAT (the so-called protective temperature) is nothing more than the temperature associated with elimination of the species from the streams studied.

As a live example of the scientific debate in using either optimum growth temperature or MWAT as the temperature criterion, examine the case of bull trout protection in Idaho. Bull trout are the most cold-sensitive salmonids inhabiting extensive areas in Oregon, Washington, Idaho, Montana and parts of Nevada. In 1996, EPA rejected Idaho's 1994 water temperature standard of 22 °C maximum daily temperature and 19 °C as an MWAT, for inadequacy in protecting bull trout. In June 1997, Idaho proposed a bull trout standard of 12 °C as an MWAT value or 15 °C as a MWMT value (EPA, 1997). In July 1997, EPA promulgated a standard of 10 °C as a 7DADM. Oregon had also conducted a technical review of its bull trout standard and had recommended an upper temperature of 10 °C as a suitable rearing standard (Buchanan & Gregory, 1997). EPA Region 10 subsequently recommended 12 °C as a 7DADM criterion after another technical review (EPA, 2003a), and this standard was adopted by Oregon (EPA, 2003b). Idaho, however, never implemented the 10 °C standard in 1997 and has not adopted the revised 12 °C 7DADM standard either. In 1998, Idaho's Department of Environmental Quality (DEQ) made a conclusion on temperature criteria necessary to protect bull trout (Hillman & Essig, 1998):

> *…we find that MWATs of 14.7 °C to 16.7 °C and short-term (24 hour) maximum temperature criteria of 20 °C to 21 °C should adequately protect juvenile bull trout rearing habitat.*

In 2001, Idaho adopted an MWMT (7DADM) of 13 °C as its standard. In the EPA's (2003a) regional temperature criteria process, Idaho urged that 13 °C be taken as the maximum 7DADM temperature (IDEQ, 2002) and cited Gamett (1999) as support, although Gamett (2002) stated:

> *Bull trout were always present where mean temperature was less than 10.0 °C, were present at 40 % of the sites where mean temperature was between 10.0 and 12.0 °C, but were not present where mean temperature was greater than 12.0 °C.*

Idaho's current bull trout-rearing temperature standard remains as a 7DADM (MWMT) of 13 °C and is not likely to change to comply with EPA guidance unless it elects to include a review of temperature standards in the triennial review of water quality standards.

In selecting 12 °C as a 7DADM (or MWMT) standard, EPA (2003a) considered recent laboratory evidence on growth rates for bull trout, in the context of food availability for bull trout under field conditions. A laboratory study by Selong et al. (2001) showed a maximum growth rate at 13.2 °C and a UUILT of 20.9 °C which, given a growth optimum (GO) of 13.2 °C and a UUILT of 20.9 °C (Selong et al., 2001), yields a calculated biological MWAT value of 15.7 °C. However, bull trout in this study were fed to satiation on food of extremely high nutritional content and it is known (Elliott, 1981) that the optimum growth temperature declines as daily ration declines. At any fixed ration, the growth response curve is bell-shaped with a peak at the optimum temperature. The optimum zone may be relatively broad in some species, with growth rates not deviating more than 5 % over a temperature range of a few degrees (Bear et al., 2007). However, at both colder and warmer temperatures beyond the optimum range, there can be sharp declines in growth rates (Brett et al., 1969, 1982; Elliott, 1981, 1994). Temperatures lower than the optimum may produce growth rates that are comparable to a corresponding temperature greater than the optimum, but it is not necessarily the case that the biological consequences of these identical growth rates

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are comparable. At high temperatures above optimal, sublethal effects such as diminished growth rates become more pronounced, but disease and impaired reproductive success can also occur. Very low growth rates at cold temperatures could result in an additional year of freshwater-rearing in anadromous species that, on a life cycle basis, would expose the fish to extra overwinter mortality. However, if a comparably low growth rate due to high temperatures occurred, the fish may likewise need an extra summer of rearing to achieve smolt size, and consequently be subject to an extra overwinter mortality period. The difference is that these fish, rearing on the warm side of the growth peak, could accumulate sublethal stress and not be able to store sufficient levels of lipids upon entering winter to enable overwinter survival (Hokanson, 1977). With this evidence and given the low food availability considered to be likely for many trout streams (Gamett, 2002; Willey, 2004; Larsson, 2005), 11 °C or 12 °C were considered to be more appropriate standards by most members of the EPA Region 10 technical committee and the Region thus opted (EPA, 2003a) for 12 °C as a 7DADM for the Pacific Northwest to account, to a limited degree, for low food availability.

Hillman & Essig (1998) developed regression equations for pairs of key temperature statistics for Idaho streams. Their relationship between the physical MWAT in Idaho streams and the MWMT is MWMT = 1.15 MWAT +0.41; their regression associating MWMT and MDMT is MWMT = (MDMT - 0.15)/1.04. Hence, if the biological MWAT is 15.7 °C, the MWMT is 18.5 °C; and if the MDMT is 21 °C, the MWMT is 20 °C. Dunham et al. (2003a) determined from surveys in Washington that the MDMT associated with the bull trout distribution limit (probability of occurrence < 5 %) was approximately 21 °C. This implies that, if the MWMT is 20 °C, there is less than a 5 % chance of occurrence of bull trout. The biological MWAT of 15.7 °C is comparable to an MWMT of 18.5 °C. This indicates that there is very little difference between the biological MWAT (comparable to MWMT of 18.5 °C) and a temperature that results in low probability of occurrence (i.e. an MWMT of 20 °C or MDMT of 19.4 °C). Reference to Dunham et al. (2003a) indicates that the probability of occurrence ranges from 0.15 to 0.30 at this temperature. In addition, the UUILT (20.9 °C) is only slightly greater. Selong et al. (2001) determined the latter from a 60 d constant temperature exposure instead of a 7 d exposure. However, the fact that virtual elimination of the species under natural stream conditions occurs at an MWMT of 20 °C indicates that it is likely that the combination of direct thermal effects on bull trout, and the increased biotic interaction of bull trout with more tolerant species near the upper temperature limits, contribute to the elimination of the species downstream. An even steeper rate of decline than that found by Dunham et al. (2003a) in bull trout probability of occurrence, with temperatures between 12 °C and 20 °C MDMT, was determined from a survey of 34 least-disturbed streams in the Salmon River basin of Idaho (Ott & Maret, 2003).

Illustration of the problem with temperature exceeding the growth optimum

Differences between MWAT and MWMT (or 7DADM) can range from 0–2 °C to greater than 12 °C (Dunham et al., 2005). Consequently, if MWAT is adopted as a protective target, the MWMT that accompanies it can vary greatly from stream to stream. Field-based temperature limits to fish distribution are typically associated with an upper temperature limit. Despite the concept that a fluctuating temperature regime can counteract the negative effects of temperature excursions above optimum levels and approaching UUILT by night-time recovery as temperatures decline (e.g. Johnstone & Rahel, 2003), cyclic regimes with greater amplitudes of variation increasingly impair growth rates. A major concern about adopting MWAT as the standard under this cyclic temperature framework is that MWAT is not an optimum temperature by definition. Consequently, the diel thermocycles have a mean temperature higher than optimum and reach peaks that can vary greatly according to the magnitude of the differences between MWAT and MWMT.

Hokanson et al. (1977) showed the effect of fluctuating temperatures where the mean of the fluctuating regime exceeded the growth or physiological optimum. Jobling

(1997) illustrated that fish growth rates increase with constant temperature to a maximum and then decline as temperatures rise above this optimum temperature. Jobling's generalised illustration based on reviews of numerous fish species under fluctuating regimes is based on feeding to satiation. By determining growth rate under constant vs. fluctuating temperatures, where each thermocycle has a mean equal to the constant temperature and varies by ± 2 °C to ± 3 °C around the mean, the ratios of growth under cyclic/constant temperatures can be plotted against temperature (°C). Although in Jobling's conceptual figure, growth rate is maximised at approximately 20 °C, the crossover point where the growth rate ratio cyclic:constant equals 1.0 is 16 °C. At the crossover point, when the maximum diel temperature is increased from 2 °C to 3 °C above the daily mean (equal to the constant temperature growth optimum), the fluctuating temperature regimes underperform the constant temperature regime and the performance declines as the amplitude of the cycle increases. Fluctuating regimes with larger amplitudes of variation provide lower growth rates than fluctuating regimes with less variation and the magnitude of impairment increases with extent of exceedance of the growth optimum. Another way to look at this is that to achieve parity in growth rates under fluctuating and constant regimes, the fluctuating regime must have a mean less than the growth optimum at constant temperatures. This is in contrast to the common recommendations of States to adopt MWATs that are substantially higher than the growth optimum.

Graphical explanation of the use of MWAT relative to optimum temperature as a temperature standard under field conditions

The implications of the use of various temperature criteria under field conditions can be represented in a hypothetical longitudinal temperature pattern for a pristine stream (Fig. 2). For example, assume that a stream flows from its headwaters 40 km to its mouth. In the headwaters, the highest 7DADM (i.e. MWMT) temperature is approximately 9 °C; at the mouth this temperature is approximately 27 °C. The longitudinal temperature pattern increases monotonically over the 40 km mainstem but begins to level out at approximately rkm 15. Bull trout are assumed to inhabit this hypothetical watershed. The bull trout OT is 13.2 °C at full rations (Selong et al., 2001) and assumed to be 12 °C at field rations (EPA, 2003a). Given near optimum temperatures of 12 °C to 13.2 °C, the core rearing zone under historic NTP (natural thermal potential) conditions would have extended from rkm 27 to rkm 32, with a 7DADM of 20 °C at rkm 17.5 (Fig. 2). Then assume that this watershed was developed broadly over its geographic extent, resulting in the 7DADM temperature for all points along the entire mainstem length now being elevated approximately 2.5 °C above the site-specific NTP. This results in the temperature trend exceeding the 20 °C 7DADM temperature (i.e. the temperature causing virtual extirpation of bull trout) at rkm 22.5. This means that there is a 5 km loss in historic marginal rearing habitat. Of even greater concern is the fact that the optimum growth temperatures are now found only between rkm 37 and 40, whereas historically they occurred between rkm 27 and 32. Due to channel gradient, however, the mainstem between rkm 37 and 40 is unusable by bull trout.

Under historic pre-development conditions in this scenario, channel gradients as well as maximum 7DADM temperatures were optimal between rkm 27 and 32 (the core growth zone), but under current conditions, the core growth zone (i.e. the channel with optimal gradients) has temperatures exceeding optimum growth temperatures for its entire length. One might argue that maximum daily temperatures within 2.5 °C of a maximum summertime 7DADM temperature equal to the optimum growth temperature do not occur for many days per summer and not for many hours per day, so if fish can tolerate adverse growth conditions for a month or two, growth compensation can overcome the poor summertime growth. However, with the longitudinal temperature patterns depicted in Fig. 2, the current, developed condition always has at least 5 km less usable habitat than the NTP condition. Implicit in this diagram is that if we adopt 12 °C as a biologically-based standard, this standard should not be exceeded on average for the NPT condition at rkm 32 (i.e. the upstream end of the core rearing zone). Under current, developed watershed

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conditions, a 7DADM of 14.5 °C should be reached on average annually during the summer at rkm 32. But with development comes a greater likelihood of more variable annual 7DADM temperatures (Poole & Berman, 2001). Consequently, even though the upstream end of the core rearing area would reach 2.5 °C above optimum annually, there is greater probability that the annual exceedance of optimum would at times be greater than 2.5 °C.

For bull trout in this hypothetical stream under NTP conditions, the mainstem from rkm 17.5 to rkm 35 was usable during the peak in summertime water temperature conditions due to the combination of suitable channel gradient and temperature conditions. The hypothetical watershed was uniformly developed, leading to a 2.5 °C temperature increment throughout the entire mainstem length. After the watershed was developed, the lower limits of distribution were reduced by a distance of 5 km. This means that of the 17.5 km of historically occupied habitat, 28.6 % of the channel length is no longer usable. This is the lower mainstem section, which is wider and therefore contributes disproportionately to total instream habitat area.

For the annual warmest 7-day period on average, the developed watershed has a longitudinal temperature pattern with 7DADM values that are greater than the field optimum growth temperature (12 °C) as well as the optimum at full ration, at all locations physically suitable for bull trout in terms of temperature and gradient. The annual NTP temperature distribution pattern can also be viewed in terms of 7DADM values at specific locations in a thermal continuum (downstream field

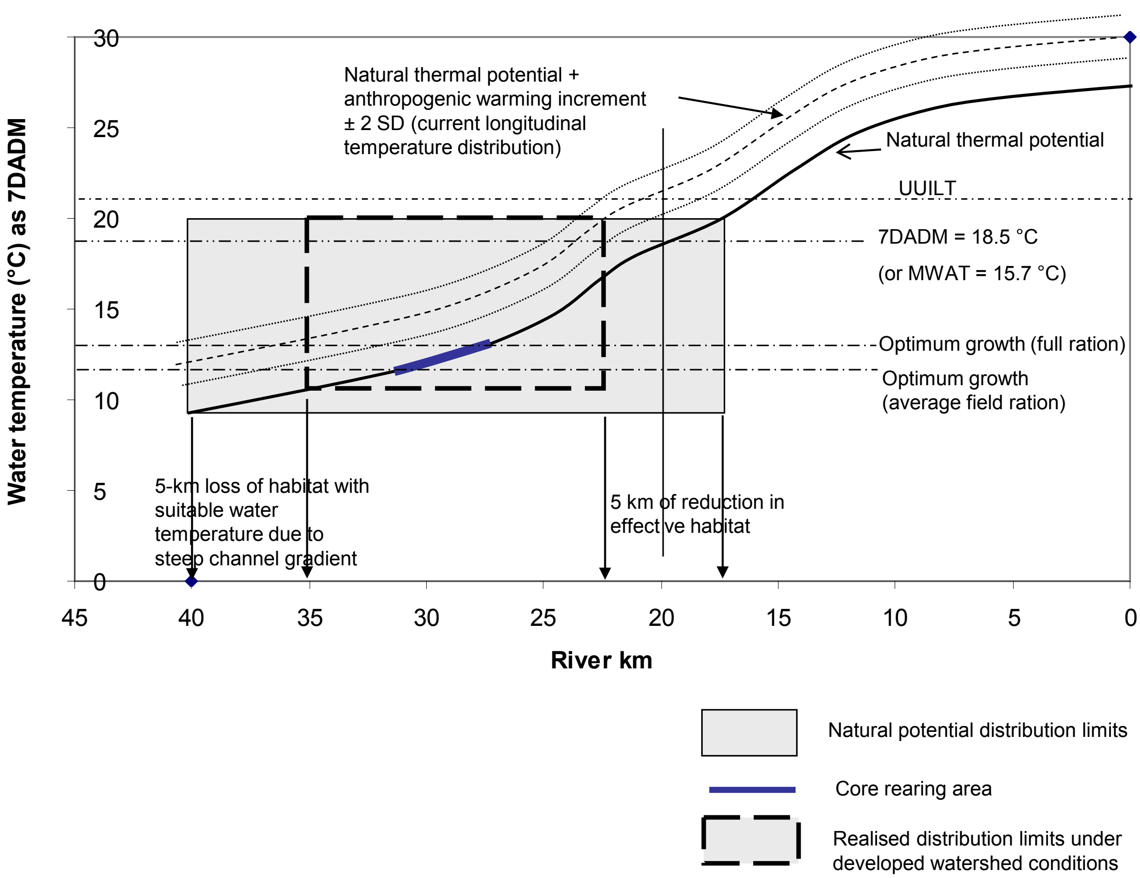

Figure 2 . Hypothetical longitudinal distribution of water temperatures (long-term average of annual maximum 7DADM) in a bull trout stream system under natural thermal potential (NTP) conditions and developed watershed conditions with an anthropogenic temperature increment.

distribution limit, Fig. 3a; optimum growth temperature, Fig. 3b). The long-term average July and August 7DADM values (Fig. 3a, b) are similar to the long-term maximum annual 7DADM values for representative points in the longitudinal mainstem path (Fig. 2). At the downstream distribution limit under developed conditions, the long-term average 7DADM is 20 °C ± 1.5 °C (or ± 2SD) (Fig. 4a) and this occurs at rkm 22.5. At the location in the mainstem where the NTP is equal to the field-based optimum growth rate temperature (7DADM of 12 °C, at rkm 32), the current condition (developed) is 14.5 °C ± 1.0 °C (or ± 2 SD) – a realised 7DADM for this location (Fig. 4b). The July–August temperature peak is fairly broad so that occurrence of maximum daily temperatures near the annual maximum 7DADM at any location are common for extensive periods during the growth season, which can limit growth for multiple days. This can be seen more clearly by examining the daily maximum and minimum temperatures during July and August. As 7DADM temperatures approach the UILT downstream, the cumulative impairment from multiple consecutive days of exposure begins to impose an increasing mortality load.

In the hypothetical example, if the bull trout biological MWAT (15.7 °C) is used as a physical MWAT to specify

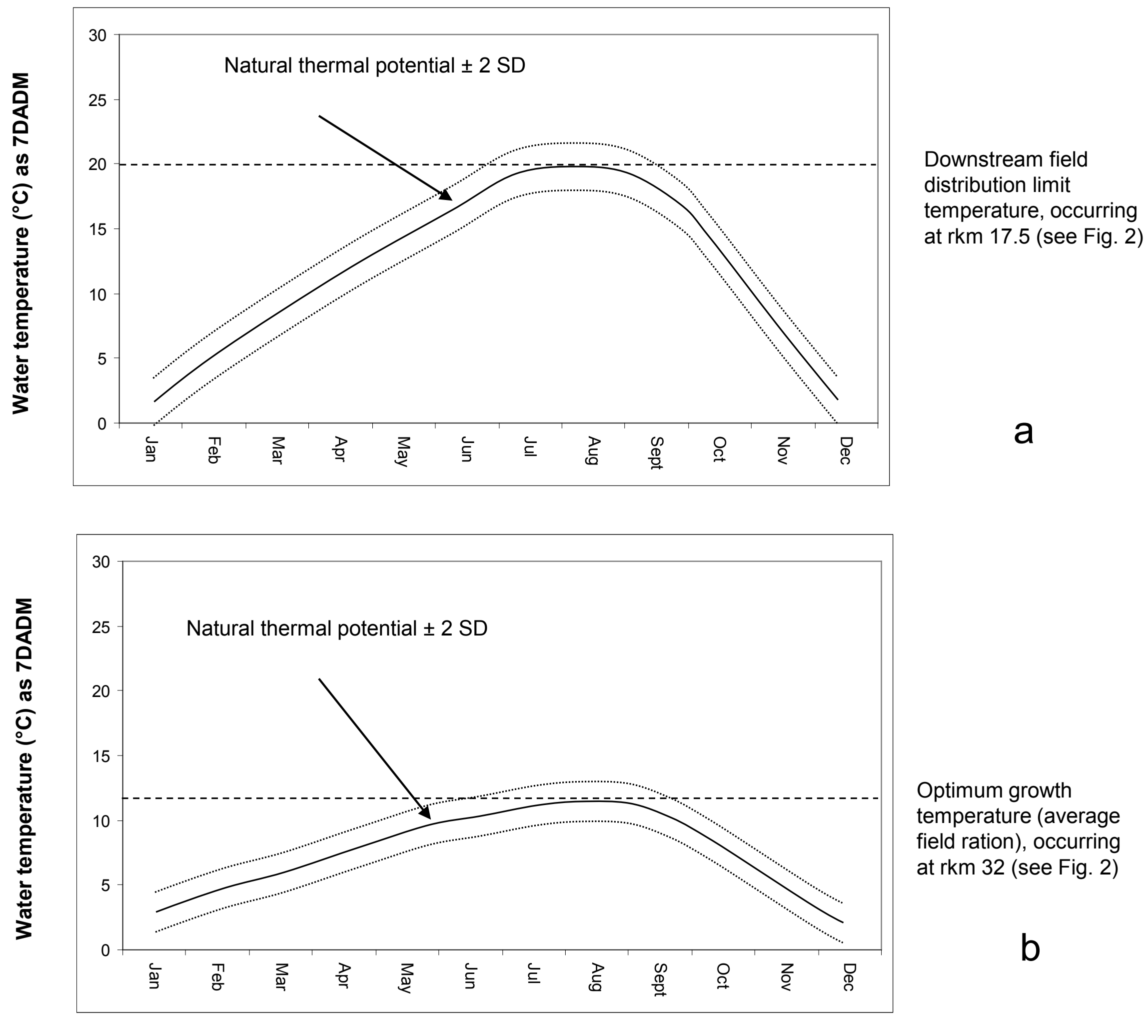

Figure 3. Hypothetical seasonal distribution of water temperatures (long-term average of monthly 7DADM) in a bull trout stream system under natural thermal potential (NTP) conditions at (a) the downstream field distribution limit (rkm 17.5) and (b) the site of optimum growth rate under average field rations (rkm 32).

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the maximum temperature permitted in the river continuum, the biological MWAT converts to a physical 7DADM of 18.5 °C. This value occurs in the developed watershed at rkm 24 whereas under NTP conditions this would have occurred at rkm 20 (Fig. 2). Under current conditions, the 20 °C downstream bull trout distribution limit occurs at rkm 22.5. This indicates that if the biological MWAT value is used as a temperature standard as proposed in EPA (1973) methodology, there is less than 1.5 km of usable habitat that would physically be incapable of achieving the MWAT value or better under current developed conditions (i.e. rkm 24-rkm 22.5). An MWAT target (or any biologically-based standard) that cannot physically be achieved is typically criticised as being indefensible and unrealistic. But a critical issue is whether a standard intended to protect fish should always be attainable throughout the entire current range or the entire historic range. In the full historic range in the hypothetical example, the biological MWAT cannot physically be achieved between rkm 17.5 and rkm 20 (see NTP longitudinal pattern). Does this mean that the only 'realistic' standard is one that results in a complete downstream end to the population distribution (i.e. 20 °C as a 7DADM)? One would have to determine what the

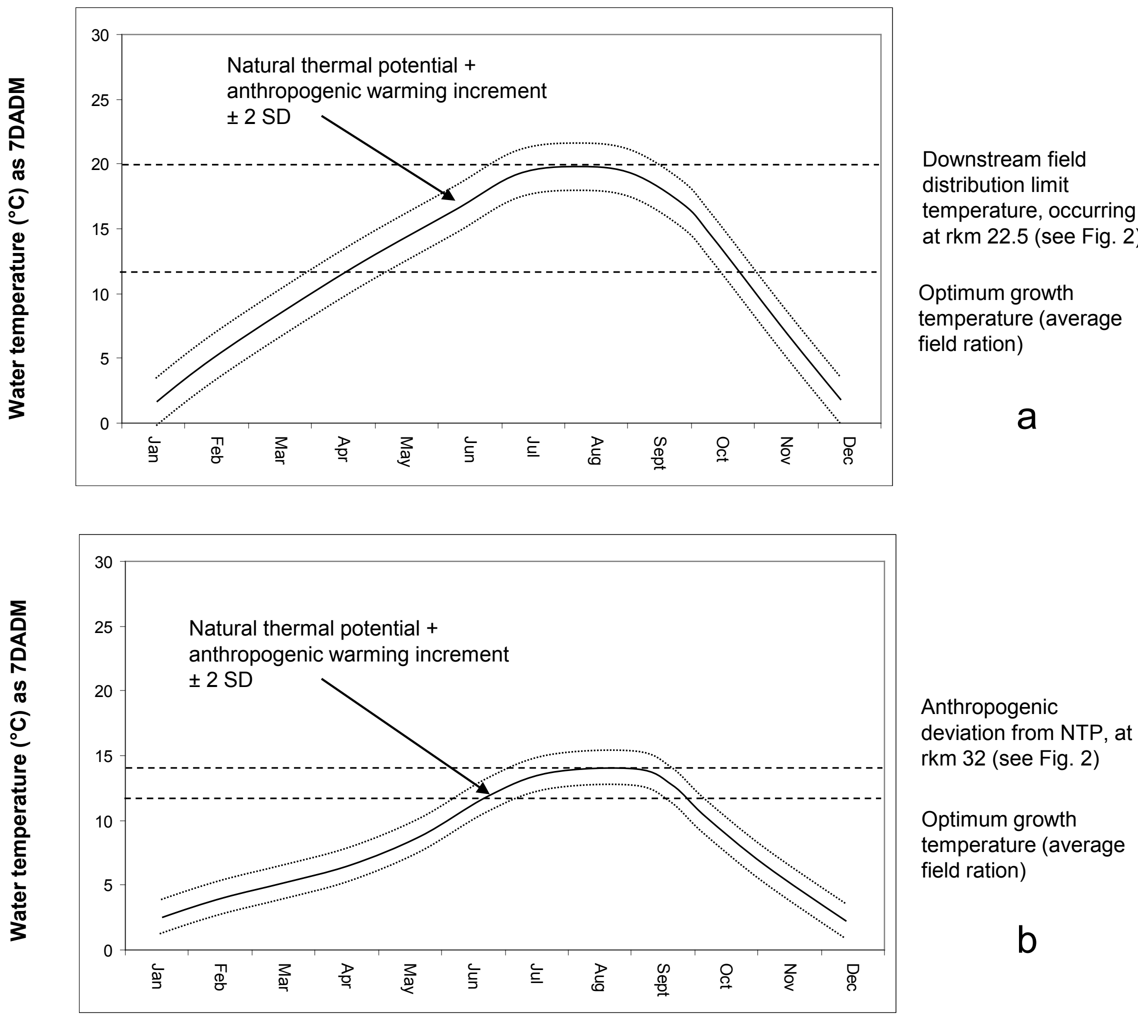

Figure 4. Hypothetical seasonal distribution of water temperatures (long-term average of monthly 7DADM) in a bull trout stream system under developed watershed conditions at (a) the current downstream field distribution limit (rkm 22.5) and (b) the site of where optimum growth rate occurs under average field rations and NTP conditions (rkm 32).

size of the watershed is to which this 20 °C standard applies. Does it apply at rkm 17.5 or rkm 22.5? If, for current conditions, we simply state that the standard is 20 °C and it applies at rkm 22.5, this means that the multi-year average 7DADM value that occurs at the lower distribution limit is the biological standard. This simply means that the current temperature trend is the status-quo standard for the future. If we make the biological MWAT (i.e. 18.5 °C as a 7DADM) the basin standard and apply it at rkm 24, it would merely reinforce the condition where the current degraded thermal regime for the basin is the target. If we require that this MWAT applies at rkm 22.5 (the current downstream end of the bull trout distribution), we would in effect adopt a standard that requires approximately a 1 °C reduction in the longitudinal trend (i.e. dropping the entire trend line by 1.5 °C). It is currently elevated by 2.5 °C. Only by applying it at rkm 20 would we be specifying a longitudinal pattern comparable to the NTP. It is clear that no single temperature standard applies effectively at all points in a river continuum. Rather, multiple check points related to the NTP are needed to control longitudinal temperature increases and create a set of targets for basin-scale restoration.

If a bull trout biological MWAT is specified for the basin and is required to be met at rkm 24, one could conclude that the current degraded regime is adopted. However, when management decisions arise concerning the allowable temperature increases permitted between rkm 24 and rkm 35, it is typically decided by reference to the basin standard. So, if the standard is the MWAT, cumulative increases up to this value might be permitted. This prospect, of a compounding set of land management actions over time on basin temperatures and the elevation of the longitudinal trendline in 7DADM values, prompted Oregon and Washington to adopt a 7DADM standard for various control points on river networks and for Oregon to adopt a maximum allowable increase that is specified as a maximum due to all combined upstream impacts at the point of greatest effect. This is a cumulative effect standard that requires tracking all point and non-point source impacts distributed in a basin. Washington has an allowable cumulative effect limit, but it is 2.8 °C in comparison to the 0.3 °C limit in Oregon. Other States have attempted to control the magnitude of temperature increases via simple standards for allowable increases, which in the cases of some States can be relatively small. But the ability to control cumulative effects with this method, which imposes no limit on the number of increases, is limited. Application of a single numeric limit such as MWAT for an entire drainage basin does not limit all points upstream within the distribution of a coldwater species to a standard less than MWAT. If MWAT is then exceeded at a fixed control point at the downstream distribution limit, it would be difficult for most States then to look headward in a river system for the points or zones of thermal increase that caused this exceedance downstream. Standards that emphasise allowable increase over natural could theoretically refer to NTP as natural. Standards for allowable increase over ambient are more prone, by definition, to consider the background standard simply as that temperature found upstream of a discharge under consideration.

Growth rates vary with temperature and describe approximately an inverted parabola. Growth rate vs. temperature curves can be very peaked or broad depending upon species (e.g. Bear et al., 2007) or juvenile life stage (e.g. Brett et al., 1969). It has been assumed that the process of calculating MWAT would be one that would identify a point on the 'shoulder' of the growth curve that represents the upper end of optimum. This conceptually would be a temperature above the centre of the optimum range but one that has nearly the same growth rate. This situation is depicted in Fig. 5. The centre of the optimum range for the bull trout example is at 13.2 °C, assuming feeding at full ration (Selong et al., 2001) (Fig. 5, point a). The UUILT for bull trout was measured by Selong et al. (2001) as 20.9 °C (Fig. 5). Consequently, the biological MWAT is calculated as 15.8 °C. If the growth response curve for bull trout has an optimum growth range that is fairly broad, the MWAT temperature would intersect this curve at point b (Fig. 5), where the growth rate is similar to that at the optimum temperature. If the growth response is more typical of fish reported in the literature, it intersects this curve at point c (Fig. 5), where the growth rate is 10 %

less than at the optimum temperature. The growth curve, however, only depicts response to constant temperature exposure. Growth at an MWAT of 15.8 °C (corresponding to a 7DADM of 18.5 °C) would not produce a growth rate similar to that expressed at point b (Fig. 5) because fish in fluctuating temperature environments acclimate to temperatures that are midway between the mean and the maximum temperature (Golden & Schreck, 1978), although Heath (1963) reported that sea-run cutthroat trout (*Oncorhynchus clarki clarki*) appeared to acclimate to the maximum temperature rather than the mean for temperatures cycled between 10 °C and 20 °C. In addition, growth rates at fluctuating temperatures with means above the optimum become increasingly impaired the greater the amplitude of variation around the mean (Jobling, 1997). The accuracy of a biological MWAT calculated from a growth curve, however, also depends first upon an accurate calculation of the centre of the optimum range. If one assumed that 15.8 °C was the centre of the optimum based on growth studies at constant temperatures where the only temperatures tested were 12.0 °C, 15.8 °C and 21.1 °C, inaccurate MWATs could be produced that are high and risky.

Using the Sullivan et al. (2000) risk-based growth analysis, a 10 % growth reduction would result from a constant test temperature of approximately 15.8 °C or a fluctuating temperature between 13.2 °C and 15.8 °C. If the stream is allowed to reach its MWAT temperature, the actual growth rates that would result would lie mid-way between points b and d or between points c and e. Most growth response curves in the literature are similar to the one with points a, c, and e with a narrow optimum growth zone (Brett & Shelbourn, 1975; Brett et al., 1982; Sullivan et al., 2000; Meeuwig et al., 2004), and hence significant declines in growth rate at the MWAT compared to the optimum. Both cool- and warmwater fish have been shown to have very steep declines in growth rates above their growth optima (for example, Kitchell et al., 1977 for yellow perch; Horning & Pearson, 1973 for smallmouth bass). It is a fallacy to assume that near optimum growth conditions occur at MWAT. This condition is expressed in the field by the close association between the physical expression of MWAT and the zone of extinction of a species.

Those opposed to recognising the value of optimum thermal habitats and extending coldwater zones spatially to the greatest extent feasible (by necessary land management practices to minimise non-point sources and regulation of point-source heat inputs) often maintain that an incipient lethal temperature would be a good standard as a maximum permissible value for each thermal guild (e.g. coldwater, coolwater, warmwater). Such a standard applied to all coldwater habitat would certainly be achievable in most cases, but where it is achieved, the species is likely to be eliminated from that point downstream. So, proposals for MWAT appear to some to be reasonable (in the sense of compromising) because they are based on average temperatures that are between optimum and UUILT temperatures. Even with a coldwater species' MWAT, there is only a small amount of habitat between the point where MWAT is expressed in the downstream temperature profile to where the coldwater species is eliminated, where it might be argued that MWAT cannot be achieved (Fig. 2). Unfortunately, a diel temperature

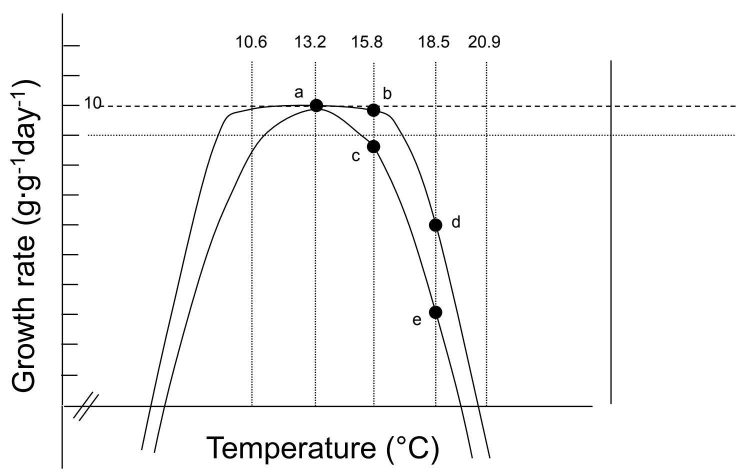

Figure 5. Two contrasting hypothetical bull trout growth rate patterns relative to constant water temperature exposure, illustrating the concept of breadth of the optimum growth response, a 10 % growth impairment level, and the problems associated with application of MWAT as a protective water temperature standard.

cycle having a mean equal to MWAT can have a maximum very close to the UUILT that for all intents would eliminate the species if applied to the full range of a species or to the range of the thermal guild. MWAT by itself would be little better than saying that UUILT is the maximum temperature permitted. For it not to eliminate species throughout the range of a thermal guild there would have to be rigorous application of narratives for land use practices that would protect and restore riparian zones, floodplains and water availability (Coutant, 1999, p. 87, Appendix A).

Application of standards such as MWAT, which are not based on biologically optimal temperatures, across a stream system without regard to natural longitudinal patterns of temperature increase, tends to allow upward ratcheting of temperatures basinwide. With MWAT being so aligned with upper thermal distribution limits, the risk is in significant declines in abundance of sensitive coldwater fish species. Numerous studies have demonstrated the control of maximum temperature on salmonid densities (Hendrickson & Knutilla, 1974; Li et al., 1994; Lessard & Hayes, 2002; Ebersole et al., 2003; Madriñán, 2008) and probability of occurrence (Gamett, 2002; Dunham et al., 2003a; Picard et al., 2003). These studies call into question presumptions by State agencies that a temperature standard can be adjusted upward to a level that is still tolerable but fully protective of the designated use. If we know that every increment of increase in the temperature standard beyond optimum causes a decline in population abundance or probability of occurrence, what higher temperature could provide 'full protection'? In addition, at temperatures approaching distributional limits throughout much of their historic range, fish experience suboptimal summer growth rates, which can lead to impaired reproductive fitness, alteration of migration timing for anadromous species, lower condition factor, impaired gamete viability, reduced competitive ability, and increased susceptibility to disease. Reduced species geographic range and thermal diversity lead to reduced genetic diversity. These mechanisms for lowering population abundance, productivity, genetic diversity, and geographic distribution reduce overall population viability (McElhany et al., 2000; Sloat et al., 2002). Consequently, temperature indices that place fish populations at upper thermal limits contribute to lowering population viability. In legal or regulatory terms, it must be considered whether significant shrinkage in range (causing a de facto loss of a designated beneficial use) and population abundance is consistent with the concept of full protection.

The effectiveness of forestry practice rules

Coldwater fish are highly dependent upon maintenance of riparian shade and, consequently, on the effectiveness of State forest practice rules. Unfortunately, the rules governing stream protection in the 50 States incorporate dramatic variations in required buffer widths on similar stream types (Murphy, 1995; Phillips et al., 1999; Broadmeadow & Nisbet, 2004; Lee et al., 2004; Williams et al., 2004). In addition to the variation among States in buffer widths, there is also a large variation in leave-tree requirements and level of zonation specified within buffers. In Oregon, a State with a long history of forest practice Act revision (Ellefson et al., 1997; ODF, 2003) as well as ESA protection of anadromous and resident salmonids, stream shading is still not fully protected. On large fish-bearing streams, buffer widths are only 30.5 m, with only a 6.1 m no-harvest zone. Tree removal is permitted outside this inner zone provided a minimum specified basal area is retained. However, on small fish-bearing streams (0.057 $m^3 \cdot s^{-1}$ flow) the minimum tree basal area coverage required is only 35 % of that of the large streams. On non-fish bearing small streams that flow into fish-bearing streams, the only leave-tree requirement is for no harvest of non-commercial timber and understory vegetation. In Idaho, the buffer for the major fish-bearing streams is only 22.9 m and only 75 % of current shade is required to be retained. For streams that have been previously harvested, the 75 % rule implies that shading is likely to continue to decline in successive riparian harvest cycles because there is no reference to site potential. For Idaho Class II streams (headwater fish-bearing streams whose primary influence is listed as maintaining water

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quality of Class I streams), there is a 9.1 m buffer that has no shade requirement and all trees larger than 20.3 cm dbh (diameter breast height) can be removed, provided a minimum number of trees between 7.6 and 20.3 cm dbh are retained. For Class II streams flowing into other Class II streams, buffers are only 1.5 m.

Riparian forest management on federal lands in the Pacific Northwest States (Idaho (ID), Oregon (OR), Washington (WA)), California and Alaska tends to provide nearly full protection of riparian shade functions (although lesser levels of protection of some other critical functions) on the major fish-bearing streams, but buffers on private lands subject to State forest practices provide far less protection (Murphy, 1995) on all stream types. Federally owned timberland accounts for 20 % of the nation's forests, while the remainder is nonindustrial private forest land (58 %), forest industry (14 %), State owned (6 %), and owned by counties and municipalities (2 %) (EPA, 2005). In its 2004 305(b) report, EPA (2009) reported that only 16 % of the nation's streams were assessed in a national water quality inventory. Of these streams, 44 % were found to be impaired from various sources. Loss of streamside habitat was a leading source of impairment. Of the 50 States, 13 have voluntary compliance, 9 mandatory compliance, and 12 both voluntary and mandatory compliance programmes. There are 16 States with no compliance monitoring (Kilgore et al., 2003). In a recent survey of senior State practice administrators, only 20 % of respondents judged voluntary guidelines to effectively regulate private forestry and 39 % viewed these to be ineffective (Ellefson et al., 1997). A random survey in Washington State of private forest practices under its new Forests and Fish programme (Lingley et al., 2009) showed that 37 % of the time forest harvest operations were noncompliant with forestry rules on fish-bearing streams. In addition to highly variable buffer requirements, allowances for removal of shade in riparian areas, and weaknesses in level of compliance and monitoring of forest practices, there is a weak feedback loop from water quality assessment to correcting forest practices.

Best practice in setting temperature standards

Technical strengths of the EPA-Region 10 temperature guidance

The EPA Regional Temperature Guidance Project (EPA, 2003a), conducted in EPA Region 10, serves as an excellent model for the 50 States of the US in development of protective temperature standards. Among the excellent ecologically-based provisions and statements of ecological concepts in this guidance are

1. EPA Region 10 guidance offers protection of existing cold waters as being essential to the process of protecting and restoring threatened fish populations.
2. It recommends protection of an entire thermal guild of fish by evaluating the full life cycle needs of all species in the guild.
3. It is explicit about full protection of the designated uses.
4. It recognises that the anthropogenic causes of stream temperature increases include riparian vegetation removal, irrigation withdrawal, channelisation and other channel modifications, separating the channel from the floodplain, removal of upland vegetation, dams and impoundments.
5. It highlights that elevated water temperatures are a major factor in the decline in coldwater species abundance.
6. The guidance requires expansion of the existing range, beyond those habitats occupied since 1975, if a greater range is needed for population viability. It recommends that stream habitat with presumed current use by a species, or potential use, is restored.
7. It recommends increasing the connectivity of a population by improving migratory pathway conditions.
8. It supports temperature requirements for all life stages.
9. It focuses on avoidance of sublethal or chronic thermal effects rather than acute effects as the primary means of controlling thermal impact.

10. In this guidance, the sublethal thermal impacts considered are comprehensive and the synergistic effects are formally evaluated.
11. Consideration is given to levels of food availability found under field conditions that affect optimum growth conditions.
12. It supports restoration of coldwater refugia in large rivers where they have been lost due to channel modification or hydroelectric impacts.
13. It demonstrates the importance of maintaining 7DADM temperatures below the critical limit in 9 years out of 10. This means that in most years, the limit should not be reached.
14. It recommends a 7DADM linked to optimum growth temperature so that acute thermal effects are not experienced.
15. It recognises that in a fluctuating temperature regime the mean temperature needs to be lower than the optimum in a constant temperature environment.
16. It supports separate thermal standards for fish with temperature requirements that are significantly colder and warmer than for the average coldwater fish species.
17. It subdivides the stream system into zones that provide for criteria for optimum spawning and rearing conditions for super-coldwater fish; optimum or core rearing and spawning conditions for coldwater fish; marginal rearing conditions for coldwater fish; and migratory corridors for coldwater fish.
18. It recommends that multiple check points within a stream system be set up at which the temperature limits should be met. These points represent the lower limit to the zones for optimal or core spawning/rearing, marginal spawning/rearing, and migratory habitat for coldwater fish. This acts as a means to control the rate of downstream warming. Standards for core spawning/rearing and for migration, and the dates by which these life functions must be met, coupled with the designated zones in the stream systems, provide geographically and temporally specific standards that provide anchor points on desirable thermal regimes, representative of the NTP.
19. It acknowledges that by controlling the 7DADM during summer, the prospect of meeting temperature criteria in other seasons is increased.
20. It acknowledges that in order to meet the required temperatures at the lower limits to a temperature zone, there must be controls on the rate of thermal increase within the zone starting from the headwaters.
21. It acknowledges that temperature control in a large watershed requires proper management at the watershed scale. The watershed is integrated longitudinally and the thermal condition at all points along this river continuum is a product of all ecological conditions and anthropogenic impacts (hydrology, vegetation, channel geomorphology etc) within the adjacent and upstream watersheds.
22. It recommends only a 0.25 °C cumulative increase in temperature (from all sources at the point of greatest impact) above full protective numeric criteria or natural background temperatures, to allow for monitoring error and negligible human impact.
23. It recommends a summer temperature at the warm end of the optimum range so that temperatures near the middle of the range would be the maximum achieved during most of the spring to autumn period. The upper end of optimum as a 7DADM was never considered as representing MWAT.

Strengths and weaknesses of Oregon's temperature standards

Oregon's temperature standards provide some good guidance for standards development elsewhere. Oregon's standards have predominantly adopted the major recommendations provided by the EPA Region 10 temperature guidance document. But, in addition, Oregon has pioneered certain other aspects of water quality standards:

1. Oregon's standards incorporate a cumulative effects provision that calculates the effect of all combined sources as demonstrated downstream at a point producing the greatest effect. This greatest effect should be ≤ 0.3 °C, a level that is assumed to be the

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maximum allowable effect that would have minimal biological impact.

2. Oregon uses a state-of-the-art water temperature model to conduct its TMDL analysis whereby allowable heat loading by point and non-point sources is calculated that will result in less than or equal to the permissible increase.

Oregon's standards have some weak points, however, that are often shared with other States:

1. Implementation of management actions needed to achieve temperature standards is a State responsibility expressed in the State Forest Practices Act and the Agricultural Practices Act. The State agencies (Forestry, Agriculture) implementing these Acts, devise BMPs (best management practices) that are typically 20 years behind the best available science, yet are trusted unequivocally to deliver conditions that will result in achieving standards if implemented consistently over the long-term. The combination of the lag time from implementation to monitoring results and the political process that slowly makes current practices conform to best available science, ensure that progress toward goals will result in slow recovery. Also, failure to achieve standards will be resisted as evidence of failure of current practices. Proponents merely claim that they have not been tried for a sufficiently long time to demonstrate their success.
2. Low flows with a 7Q10 recurrence interval (i.e. 7-day low flow with a 10-year recurrence interval) or lower are taken as conditions that exempt temperature exceedances. However, irrigation withdrawals can easily produce conditions that artificially trigger exemptions.
3. High air temperatures (> 90th percentile of an annual series of maximum annual 7DADM temperatures) are permitted as an exemption. Upward trends in climatic maxima make statistics of past meteorological conditions unsuitable for creating biologically safe exemptions.
4. Insufficient effort is directed to tracking of current levels of cumulative watershed impact in order to predict maximum thermal impact.
5. The BMPs provided as options in TMDLs are frequently voluntary (EPA, 1994; Boyd, 2000) and offer many easy but ineffective actions that reduce the already weak regulatory pressure.
6. Temperature modelling is typically conducted without incorporating full hydrological restoration, due to technical difficulties such as accounting for effects of lost wetlands and changes in watershed vegetation cover. Furthermore, the social conflicts involving water rights for agriculture versus minimum established flows for fish can inhibit State agency modellers from identifying the natural potential water temperatures that would be associated with historical flow. Consequently, modelled, restored temperature regimes are not representative of historical conditions.
7. No provisions for incorporating the anticipated effects of global warming as a measure of safety are made.
8. Intermittent or non-fish bearing streams typically have much lower levels of protection, which disregards the downstream impacts of thermal disturbances.

Recommendations for development of revised temperature standards in future triennial reviews

This review of water temperature standards at a national level suggests a number of actions that are needed for improving protection of the nation's coldwater fish species:

1. Recognise stream temperatures as having a natural potential thermal regime at all points along the river continuum that is also subsequently influenced by cumulative anthropogenic actions.
2. Given the spatial and temporal variations in temperature that have become increasingly recognised by FLIR (Forward Looking Infrared, or TIR) monitoring, establish control points along the river continuum that express desired quantitative targets along the NTP trajectory, allowing for a

minimum anthropogenic temperature increase that does not impair the use.

3. Use NTP as a guide for target temperatures along the river continuum.
4. Recognise that thermal restoration toward NTP as an endpoint provides the greatest balance in potentially occupied range and viability for all native species and provides the conditions where the thermal regime is ideally matched with channel gradient and other habitat factors suitable for all native species. As a corollary, when considered at a basin level, there is no such thing as being 'overprotective' when NTP and the associated riparian, channel, floodplain, and watershed protection required to achieve NTP are employed.
5. Derive the NTP from a combination of procedures, including physical temperature modelling (based on restoration of channel riparian cover, channel morphology, instream flows and floodplain condition and connectivity with the stream channel), inference from minimally disturbed streams of the same stream class, use of historical temperature records collected prior to significant disturbance, and use of fish records collected prior to disturbance (see ODEQ, 2004).
6. Abandon the use of the biological MWAT as specified by EPA (1973) as a criterion for full protection of growth, because it is linked to significant large reductions in growth.
7. Also abandon the concept that an adequate standard can create multiple levels of impairment (e.g. 10–20 %) on physiological function (e.g. growth, egg viability, disease resistance) that would not produce cumulative impairment to a species' population, restrict its spatial distribution, reduce viability, of impair genetic diversity.
8. Revise water temperature standards so as to provide full protection to the most sensitive species. If a State has salmonids with requirements for cold water that vary quantitatively (e.g. bull trout and Chinook salmon (*Oncorhynchus tshawytscha*), or cutthroat trout and rainbow trout), it is better to apply the optimum requirements for the various life stages to standards than to simply manage to a median condition. These standards would then apply to core areas on the basis of NTP. Appropriate land management upstream of core areas would then be required to ensure core habitat conditions were met downstream.
9. Abandon the use of RIS for thermal protection under point-source impacts. Instead, substitute the use of IBI (Index of Biological Integrity) or another full community composition measure and NTP, as inferred from a scientific investigative process such as outlined above.
10. Rely on optimum growth temperatures as the best guide to thermal conditions that are protective of the species rather than MWAT, UUILT, or avoidance temperatures.
11. Counteract the shifting environmental baseline paradigm by use of reference conditions, even for large river systems. There is no reason, for example, that the Penobscot River could not represent, as a rough guide, macroinvertebrate or fish community trends for the Connecticut River.
12. Eliminate the use of incremental comparisons to justify a conclusion of 'no effect'. For example, after allowing two centuries of thermal degradation on the Connecticut River, considering the biological impact of an additional proposed small temperature increment is the wrong test of protecting the designated beneficial use, unless a highly disturbed ecosystem (current condition) is the desired condition. Also, upstream–downstream comparisons in a system that is already highly disturbed will make significant changes increasingly difficult to demonstrate. Coldwater fishes are already rare and are found in numbers difficult to define statistically via sampling relative to more abundant species that are more heat tolerant. Under such circumstances, the disappearance of coldwater fishes may be overlooked.
13. Require actions to be implemented that are known from scientific studies to be effective in restoring thermal regimes. Most of these actions will have related benefits in improving other aquatic limiting factors, such as reducing fine sediment delivery,

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improving delivery of large woody debris, and increasing bank stability.

14. Recognise that the increasing air temperatures predicted from global warming effects will place greater demands on retaining and restoring riparian vegetative cover, restoring stream channels, and returning instream flows to natural hydrologic regimes.

Conclusions

The Clean Water Act is the key US law dealing with maintenance and restoration of water quality and aquatic health. Although the CWA has many strengths that have resulted in large gains in water quality in past decades, it has many built-in weaknesses that thwart continued progress. From its inception, implementation of the CWA focused primarily on point-source pollution (EPA, 1994). This emphasis might be assumed to provide protection of dependent fish communities faced with point-source discharges from facilities such as thermoelectric plants, were it not for the multitude of exceptions and conflicting provisions of the CWA in its various sections (noted above). Non-point source (NPS) pollution, contributed by diffuse sources (agriculture, forestry, rangeland grazing, and urban runoff), is the predominant cause of nonattainment of standards in the United States (Houck, 1999; Copeland, 2009). Despite their pre-eminent role in the intent of the CWA not being met (i.e. 'to restore and maintain the chemical, physical, and biological integrity of the Nation's waters' and achieving 'fishable, swimmable waters'), non-point sources of thermal pollution have remained largely unregulated except via an assortment of BMPs that may have a level of effectiveness inadequate to achieve standards, or TMDLs that may or may not be written or enforced by States. Lack of enforcement is a problem common in the US as well as in the EU (Rechtschaffen, 2004, 2007). Failure to address full protection in the US has arisen over the years due to a variety of agency administrative perspectives and legal interpretations as well as narrowing of the CWA by various courts, none of which hold true to the overriding goals of protecting biotic integrity. For example, EPA has consistently argued that pollution produced from dams was exempt from permit requirements under the CWA (Blumm & Warnock, 2003), which has resulted in thousands of US dams not being responsible for thermal pollution caused by their operation. EPA's preference has been upheld by various courts on grounds of agency deference. The US Forest Service has argued in court that livestock grazing or any NPS pollution on federal lands does not require a 401 certification (a certification provided by a State under Section 401 of the CWA that any applicant for a federal permit to discharge wastes into a waterbody will not violate State water quality standards). Again, EPA elected to support this view and the currently dominant court opinion makes it unfeasible to use the CWA to control NPS water pollution arising from federal land management (Blumm & Warnock, 2003).

The US Congress must also assume a greater role in clarifying the level of aquatic resource protection required under the CWA. This conclusion was reached in a recent US Supreme Court ruling in *Entergy Corp. v. Riverkeeper, Inc.* EPA had been operating under a policy of applying best available technology (BAT) to 'minimize adverse environmental impact'. However, in a 2001 amendment to EPA regulations with regard to existing power plants, it opted to employ cost-benefit analysis in deciding the extent to which mortality in cooling towers from impingement and entrainment would be limited when balancing aquatic resource protection and economic impact to industry (Harvard Law Review, 2009). The majority opinion in this case noted that minimising impact does not always imply the 'greatest possible reduction' in mortality. This case allows EPA to use discretion in determining the extent of impact allowed. However, it is questionable to what extent cumulative effects, external environmental costs, and general ecosystem impairment can be effectively assessed and prevented in such cost-benefit analyses (Mears, 2008).

EPA does maintain a role in requiring that the States implement TMDLs, but if they fail to produce a TMDL or even a sufficient TMDL to address meeting State water quality standards, EPA will promulgate a TMDL. The TMDL is designed to ensure that both point and non-point pollution sources are accountable for meeting water quality standards. However, NPS provisions in TMDLs, to the extent that a TMDL is ever developed, rely on application of BMPs, which are at the prerogative of individual States (Houck, 1999). Also, EPA does not enforce anti-degradation provisions as a means to 'maintain' water quality on high quality waters (i.e. Tier II) at risk from NPS pollution, because in the TMDL process used to control NPS pollution, control is not considered until standards fail to be met (Blumm & Warnock, 2003).

Clearly, a combination of a more engaged EPA plus clearer CWA language that directs efforts equally to point and non-point sources, remains true to the original 1972 intent to maintain and restore the nation's waters and biodiversity, and eliminates loopholes and conflicting directions, would assist in protecting the nation's coldwater fishes and aquatic diversity as a whole. Importantly, implementation of TMDLs that consider the dual thermal pollution sources (point source and NPS) is also predicated on the prior presence of temperature standards. To be effective and provide full protection, standards need to be biologically meaningful and optimal and also suitable to be implemented in a spatially appropriate fashion consistent with natural potential thermal regimes (Poole et al., 2004). Without standards or effective standards, monitoring, and 303(d) listings of streams not meeting standards, there is no TMDL process. The haphazard nature of 303(d) temperature listings and temperature TMDL programmes across the 50 States is testimony to the central lack of attention to temperature in many States as a key determinant of aquatic health. Lastly, EPA Gold Book guidance for water temperature is in great need of revision for temperature effects to fish and no longer serves as a reliable model for setting water temperature standards given the inadequacy of the biological MWAT index elaborated here.

The question of whether the US's coldwater fish species are being protected through application of the CWA is a complicated one involving consideration of the distribution of native and non-native species, natural potential of streams to sustain salmonids, land ownership and relevant management rules, inadequate laws, inconsistent EPA guidance, standards adopted as fully protective, exemptions in the State water quality standards, effectiveness of BMPs used in addressing a TMDL, the willingness of States to provide sufficient instream flows, the likelihood of development and enforcement of TMDLs, frequency of enforcement, and climate change. The climate change that has already occurred has resulted in widespread, upward water temperature trends (Kaushal et al., 2010). Even small water temperature increases can result in significant range reductions in coldwater fish species (Rahel et al., 1996). Also, most States have no obvious mechanism to prevent progressive thermal degradation, or to manage thermal regimes on a system-wide basis in relation to natural thermal potential, that would ensure a robust distribution and population health of historic native species. Considering all these factors, it is difficult to arrive at any other conclusion than that water temperature increases, usable habitat contraction, and range expansion by warmwater tolerant species present a clear threat to the future viability of salmonid populations in all States of the US supporting salmonids, placing increased emphasis on the need to devise and implement effective water quality standards that are technically sound and fully protective.

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Author Profile

I am a senior fisheries scientist for the Columbia River Inter-Tribal Fish Commission in Portland, Oregon with a background in stream ecology. I have worked for over 25 years for four Native American tribes with treaty fishing rights on the Columbia River. My work has focused on technical analysis of water quality, salmonid habitat, and land management in the interest of protecting and restoring stream and watershed conditions in the ancestral tribal lands.

Glossary

Commonly used acronyms and terms:

303(d)	A section of the Clean Water Act (CWA). It specifies that water bodies not meeting water quality standards after point sources have had pollution control technology applied, will be listed in a statutory report (commonly referred to as the '303(d) List'). Listed water bodies require development of a TMDL.
303(d) report/list	A report to EPA required under the CWA that lists all waterbodies not meeting water quality standards set individually by each State to meet all State-designated uses of the waterbodies. When a waterbody is placed on the 303(d) list for a water temperature impairment, the State must develop a TMDL that defines how thermal loads will be managed to fully protect the designated beneficial use(s).
305(b)	A section of the CWA requiring EPA and the States to submit a biennial report to Congress detailing water quality by State.
305(b) report	A report that is required to be submitted to EPA by each US State every two years stating the water quality condition of all navigable waters and the extent to which they provide protection and propagation of a balanced population of shellfish, fish, and wildlife, and allow recreational activities.
316(a)	A section of the CWA that regulates point-source thermal discharges into water bodies. A permittee (e.g. power plant owner) interested in gaining a thermal variance may conduct a 316(a) 'demonstration' to show that a 'balanced indigenous population' [interpreted as community] will be maintained. Harm to the BIP was interpreted by EPA (1977a) as impairment to growth, development, reproduction, survival and distribution.
316(b)	A section of the CWA requiring that power plant cooling water intake structures use best technology available (BTA) to minimise adverse environmental impact (AEI). Cooling water intake impacts to be considered in this section include organism impingement and entrainment.
AEI	Adverse Environmental Impact. AEI was interpreted by EPA (1977a) as absolute or percentage damage to fish impinged or entrained, or to endangered, commercially valuable or sport species, important aquatic species, and to the BIP.
Antidegradation	This is a policy under the CWA to prevent deterioration of good water quality.
BIC	Balanced indigenous community. BIC was interpreted by EPA (1977a) to be equivalent to BIP.
BIP	Balanced indigenous population. The BIP consists of desirable species of fish, shellfish and wildlife that are essential components of the foodweb, but may also include desired, introduced species.
BMP	Best Management Practices. BMPs are considered to be best procedures for controlling pollution sources and are employed in TMDL implementation plans to control thermal sources.
CFR	Code of Federal Regulations. This is the codification of rules published in the Federal Register by executive departments and agencies of the US Government, published under 50 titles representing subjects under Federal regulation.
CWA	Clean Water Act. This is the major US environmental law, passed in 1972, dealing with regulation of surface water pollution.
Designated beneficial use	A purpose or benefit to be derived from a water body and which is formally designated for the water body. Potential beneficial uses include watering of livestock irrigation, aquatic life, recreation involving contact with the water, municipal or domestic supply, industrial supply, propagation of wildlife, waters of extraordinary ecological or aesthetic value, and enhancement of water quality. States may tailor beneficial uses to local conditions, such as subdividing aquatic life uses as coldwater fish or warmwater fish uses. The CWA requires States to designate one or more beneficial uses for each waterbody. Water quality standards must then be devised so that designated beneficial uses are fully protected, especially the most sensitive use
EPA	US Environmental Protection Agency. EPA is an agency of the US Government charged with protecting human health and environmental quality. It interprets and enforces laws created by Congress.
IBI	Index of Biotic Integrity. This is a multimetric index of community composition.
Kendall-Tau b	This is a nonparametric statistical test of rank correlation between two variables (Kendall, 1970). This statistic has values of -1 to +1.
NPS	Non-point source pollution. Common sources of NPS pollution are the diffuse runoff from agricultural, urban and forestry lands.
NPDES	National Pollutant Discharge Elimination System. Section 402 of the CWA requires that industries obtain an NPDES permit with TBELs and/or WQBELs, and monitor and report pollution discharges.

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NTP	Natural Thermal Potential. NTP is an estimate of the thermal regime without prior anthropogenic modification.
OT	Optimal temperature. OT for a certain life stage is generally identified as that temperature that maximises growth rate and survival.
PS	Point source. This is a source of pollution typically emanating from a pipe, ditch, or animal feedlot.
RBP	Rapid Bioassessment Protocol. This is a set of protocols for characterising periphyton, macroinvertebrate and fish communities to assess reference and impaired stream conditions.
RIS	Representative Important Species. RIS is a set of species considered representative of a BIC. It is assumed that if they are protected, the entire community will be protected.
TBEL	Technology-based effluent limitations. TBELs are effluent limitations required by EPA for the discharge from a particular industrial plant, regardless of the quality of the receiving water.
TMDL	Total Maximum Daily Load. It is a calculation of the maximum pollutant load permissible that will still result in water quality standards being met. This is followed by an implementation plan to meet standards.
USC	US Code. The USC is the codification of the permanent laws of the US under 50 titles, published by the US House of Representatives.
UUILT	Ultimate Upper Incipient Lethal Temperature. UUILT is the highest UILT possible under conditions where organisms with prior temperature acclimation are then subjected to a test temperature for a period of either 1000 min or 24 h, producing 50 % survival.
WQBEL	Water quality-based effluent limitations. WQBELs are effluent limitations imposed on a point source and are based on the water quality standards for the receiving water body designed to support a designated beneficial use.
WQS	Water quality standard. WQSs include designated uses, water quality criteria (narrative or quantitative), and antidegradation provisions.

Acronyms used to describe water temperature statistics:

7DADM	7-day average of the daily maximum. 7DADM is the same as MWMT.
MWMT	Maximum weekly maximum temperature. MWMT is the highest 7-day running average of daily maximum temperatures during a year.
MDMT (and MDT)	Maximum daily maximum temperature.
MWAT (physical)	Maximum weekly average temperature. MWAT is the largest 7-day running average of daily average temperatures, generally computed from hourly temperatures.
MWAT (biological)	Maximum weekly average temperature. MWAT (biological) is computed by an equation relying on OT and UUILT EPA (1973). A physical MWAT under field conditions equal to the biological MWAT is assumed to be protective (EPA 1973).