LINKS BETWEEN WATER TEMPERATURES, ECOLOGICAL RESPONSES AND FLOW RATES: A FRAMEWORK FOR ESTABLISHING WATER TEMPERATURE GUIDELINES FOR THE ECOLOGICAL RESERVE

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by

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EXECUTIVE SUMMARY

Global ecosystems face unprecedented crises in habitat fragmentation, destruction and ultimately extinction (Groves, 2003), and of all the varying ecological systems rivers are the most neglected and endangered (Groves, 2003; Driver, et al., 2005; Roux et al., 2005). The greatest threat to these systems is the loss or degradation of natural habitat and processes (Driver et al., 2005), and water temperatures, after flow volumes, are a primary abiotic driver of species patterns within river systems. Stuckenberg (1969) highlighted the links between temperature, topography and faunal assemblages, while Rivers-Moore et al. (2004) highlights the major impacts of water temperatures on organisms, and illustrate how water temperatures are one of the primary environmental drivers structuring fish communities in the Sabie River, arguably the most icthyologically species-rich river in South Africa.

The National Water Act (Republic of South Africa, 1998) provides legal status to the quantity and quality of water required to maintaining the ecological functioning of river systems, through the declaration of the “ecological Reserve” (see Chapter 3, Part 3 of the National Water Act of 1998). To date, no methods have been developed for the water temperature component of the Reserve, although the importance of water temperatures in maintaining river systems is fully recognised (Poole and Berman, 2001; Johnson, 2003). Understanding temporal variability of temperature time series, regional variation, and how aquatic macroinvertebrates respond to thermal regimes both spatially and temporally, are central to determining ecological Reserves and in defining policies to manage river systems. This fundamental understanding must occur to inform policies since sound decisions rely on the best available scientific evidence (Brundtland, 1997). Ecological Reserve determination for rivers in South Africa presently does not include a water temperature component in spite of its importance in determining species distribution patterns. To achieve this requires an understanding of how lotic thermographs from South African rivers differ from northern hemisphere rivers.

The broad temperature classifications of potamon and rithron are suggested as being useful starting points to an initial water temperature contribution to the ecological Reserve. Notwithstanding the lack of available data and the inherent problems with simulating time series in highly variable systems, it is nevertheless worthwhile to characterize system variability since it is this aspect of a thermal regime that enables aquatic organisms with different tolerances to co-exist within a single reach of river. Based on the various techniques utilized within this study, the coefficient of variation is sensitive to extreme values which characterize flow rate time series but not water temperature time series to the same degree. As a simple first approach, the coefficient of variation is probably adequate in characterizing annual flow rates and water temperatures between rivers. Agglomerative techniques, such as the use of duration curves using successive days within class intervals, have potential for defining management targets such as Environmental Flow Requirements. What should emerge as a guiding principle is that a suite of different metrics should be used simultaneously to characterise water temperatures. In a country where river systems are highly variable, and often defined by extremes rather than averages, the importance of understanding variability and how this relates to the ecological Reserve cannot be overemphasized.

Hourly water temperatures from 20 sites in four river systems, representing a range of latitudes, altitudes and stream orders, were assessed using a range of metrics. These data were analysed using Principal Components Analyses and multiple linear regressions to understand what variables a water temperature model for use in ecoregions within South Africa should include. While temperature data are generally lacking in low and higher order, South African rivers data suggest that South African rivers are warmer than northern hemisphere rivers. Water temperatures could be grouped into cool, warm and intermediate types. Based on temperature time series analyses, this report argues that a suitable water temperature model for use in ecological Reserve determinations should be dynamic, include flow and air temperature variables, and modified by a heat exchange coefficient term. The inclusion of water temperature in the determination and management of river ecological reserves would allow for more holistic application of the National Water Act's ecological management provisions.

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1. INTRODUCTION

Global ecosystems face unprecedented crises in habitat destruction and extinction (Groves, 2003). Of all the different ecological systems within South Africa, river systems are the most neglected and endangered, a situation paralleled globally (Groves, 2003; Driver et al., 2004; Roux et al., 2005.). The biggest threat to these systems in South Africa is the loss or degradation of natural habitat and ecosystems (Driver et al., 2005). The conservation of streams and rivers has historically been given a low priority, largely because of the conceptual difficulties in conservation planning and management in these linear systems (O'Keeffe et al., 1987). Rivers are inherently difficult to study, but this lag in understanding has come at considerable cost. For example, Fausch et al. (2002) report an "alarming" rate of decline of fish in North American rivers.

In recognising the crisis faced in safeguarding this resource, current conservation planning aims to maintain not only the biological diversity within systems, but also the ecological and evolutionary processes which ensure the persistence of such systems (Groves, 2003). This is achieved through a better understanding of these processes, and in defining conservation goals and objectives, which are ultimately met through setting of targets. However, as Tear et al. (2005) point out "conservation practitioners are still typically at a loss when establishing a quantitative target" (p. 835). Certainly, "the complexity of conserving biological diversity cannot be overestimated" (Tear et al., 2005, p. 836). Conservation thinking is moving away from single-species management, to systems management (Driver et al., 2005), where there is a growing recognition of the non-equilibrium paradigm replacing that of systems in stable equilibrium through succession (Groves, 2003). Systematic conservation planning aims for representation, redundancy, and resilience of ecosystems (Margules and Pressey, 2000), where there is an inherent recognition of the need to preserve processes and system variability.

Water temperatures, after flow volumes, are a primary abiotic driver of species patterns within river systems, and thus ecosystem services. Stuckenberg (1969) highlighted the links between temperature, topography and faunal assemblages, notably for snakes and amphibians. Rivers-Moore et al. (2004), in a review, have highlighted the major impacts of water temperatures on organisms, and have also shown how water temperatures are one of the primary environmental drivers structuring fish communities in the Sabie River, arguably the most species-rich river in South Africa in terms of fish. Understanding temporal variability of temperature time series, how they vary regionally, and how aquatic macroinvertebrates respond to thermal regimes both spatially and temporally, are central to determining ecological Reserves, and in defining policies to protect river systems.

This basic understanding must occur to inform policies, since, as Brundtland (1997) remarks, "politics that disregard science and knowledge will not stand the test of time. Indeed, there is no other basis for sound political decisions than the best available scientific evidence. This is especially true in the fields of resource management and environmental protection".

The role of science is to inform policy (Groves, 2003), and to do this, "we need the best evidence to allow us to make the most informed decisions as to where we need to place most of our efforts" (Lomborg, 2003, p. 5). "If we allow environmental arguments to be backed merely by purported trends of two or three carefully selected years, we invariably open the floodgates to any and every argument. Thus, if we are to appraise substantial developments we must investigate long periods of time….Insisting on long-term trends protects us against false arguments from background noise and lone swallows." (Lomborg, 2002, p. 9). Tear et al. (2005, p. 838) also point out that "the job of scientists is to make clear exactly what numerical objective is required to reach an associated goal.

According to Ashton et al. (2005), integrated research can only occur after a foundation of pure fundamental research has been built. This will require an ecosystem perspective (Covich et al., 1999), and catchment-scale management (Allan et al., 1997). While it is recognised that often there is an artificial separation between government and ecological systems, where the policy basis for joint management of land and water is poorly understood (Allan et al., 1997), policy and land use decisions are typically applied at the lowest appropriate levels of government (Allan et al., 1997; Ashton et al., 2005). The management challenge is to take into account the highly variable resource base of our rivers. The governance of water in the hydrological cycle is distributed amongst several government agencies, as well as the soon-to-emerge Catchment Management Agencies. Good management of this resource will require co-operative governance, which is ideally aligned with the ecological regions.

A main objective of this report was to produce a literature overview which contributes to enhanced understanding of water temperature signature components critical to South African rivers, and how these may differ in type and degree from northern Hemisphere systems. Included in this report is a chapter on the theoretical effects of natural and modified thermal regimes on aquatic invertebrate communities, using the conceptual model of Vannote and Sweeney (1980) within a hierarchical approach.

This report constitutes two of three deliverables of a one-year consultancy to the Water Research Commission, entitled "**Links between water temperatures, ecological responses and flow rates: A framework for establishing water temperature guidelines for the ecological Reserve**", and specifically addresses four aims as follows:

1. Develop a conceptual model for identifying critical water temperature signature components and how these could be used to develop general temperature guidelines for the ecological Reserve.
2. Add to an understanding on possible differences between northern and southern Hemisphere thermal patterns in river systems.
3. Review of theoretical effects of natural and modified thermal regimes on aquatic invertebrate communities, and how this could be related to the ecological Reserve.
4. Refine water temperature simulation modelling approaches, including spatially dynamic models, and how such models could be linked to existing hydrological models.

The report is divided into four main sections, with aims one to three addressed in sections two to four respectively, and aim four covered in section five.

The third deliverable was the development of a Terms of Reference for a long-term water temperature programme, and is described in Dallas (2007).

2. CLASSIFICATION OF WATER TEMPERATURE VARIABILITY

Two abiotic signatures of any river system useful to river managers are water temperature and flow rate. Rivers with similar hydrological regimes are expected to express greater rather than random similarity in ecological organization (Poff, 1996), which also applies to rivers with similar temperature regimes. The importance of water temperatures to aquatic biota has been well documented (for example Claska and Gilbert, 1998; Eaton, 1996; Elliott, 1994; Sullivan et al., 2000). Changing the thermal regime of a river significantly alters a component of the environment for which river organisms are adapted (tolerances and life cycle cues) (Appleton, 1976; Ward, 1985). There is currently renewed interest worldwide, in understanding the thermal regime of rivers and streams, due to anticipated alterations to the natural thermal regimes of many rivers (Johnson, 2003) as a result of, *inter alia*, impoundments, land use change and climate change. More specifically, cumulative **daily maximum** water temperatures have been shown to have the greatest effect on the distribution of aquatic species (Armour, 1991; Caissie et al., 2001; Essig, 1998; Hines and Ambrose, 1998; Robison et al., 1999; Sullivan et al., 2000; Dunham et al., 2003). In addition to this, Vannote and Sweeney (1980) consider the most important aspect of a river's thermal regime to be its temporal predictability. According to Brunke et al. (2001), "any modelling and prediction of instream habitat requirements are incomplete when critical time periods or essential localities during the life-cycle of the target species are not recognised" (p. 674). Vannote and Sweeney (1980) proposed that variable seasonal river temperature patterns are the critical factor in maintaining temporal segregation in aquatic invertebrate communities, thus allowing for resource partitioning, and preventing competitive exclusions, while spatial differences in water temperatures allow for zonation of species.

2.1 Importance of variability

Aquatic systems are subject to great variability (Reynolds, 1998), and water temperatures are an integral component of this variability. There is a growing recognition of the importance of variability in maintaining river health (see for example Gunderson et al., 1995; Reynolds, 1998; Richter et al., 1996; Richter et al., 1997; Schindler, 1987), through its role in the constant revision of the thermodynamic base (Reynolds, 1998). Highly diverse systems have more pathways for energy flow (Reynolds, 1998) and greater resilience to disturbance. Less diverse systems tend to be less resilient (where resilience is defined as the rate at which a system returns to "normal" conditions) (Holling, 1973; Hashimoto et al., 1982), since they are unable to accommodate such variability. Generally, maximum biotic diversity is maintained in streams by a level of disturbance (or variability) which creates environmental heterogeneity, yet still allows the establishment of aquatic communities (Ward and Stanford, 1983).

The River Continuum Concept (RCC) of Vannote et al. (1980) has been an important heuristic tool in freshwater aquatic science for almost three decades. This concept includes predictions of how variability of abiotic components changes along the longitudinal axis of a river. There is a circumspect adoption of the RCC for southern

African River systems, with recognition of the importance of variability for maintaining ecological processes and biotic diversity. According to the RCC, rivers exist in a state of dynamic equilibrium, and because of this the roles of succession and island biogeography theory (species-area relationships) do not apply. Within river systems, the role of biotic diversity is less important in maintaining ecosystem stability in highly stable physical systems, such as in headwaters. Conversely, as variability increases with downstream distance, the role of biotic diversity becomes more critical in maintaining ecosystem stability. Within this paradigm, water temperatures contribute to instability (Vannote et al., 1980). Thus biotic diversity mirrors variability in the daily temperature pulse along the longitudinal axis of a river, peaking in the mid-reaches. Such a relationship was shown to apply, at least within the Sabie River for water temperatures and fish species (Rivers-Moore et al., 2004). Within the headwaters, aquatic diversity is low because only invertebrates with narrow temperature tolerances persist (Vannote et al., 1980).

Temperature is a key factor affecting the number and kinds of species in a stream (Vannote and Sweeney, 1980). Annual thermal variation allows closely related species to be sequenced temporally, and thus avoid competition, while diel temperature change increases the potential number of species which can coexist, as each one will be exposed to an optimum temperature during the day (Vannote and Sweeney, 1980). The degree of predictability in a stream's water temperatures will thus provide an indication of the degree of structure and functional predictability of invertebrate communities (Vannote and Sweeney, 1980).

Aquatic species typically respond in predictable ways to changing environmental conditions. Biological strategies depend on the reliability of a pattern (Colwell, 1974), or on how often the system fails (Hashimoto et al., 1982). In order to understand the pattern of such responses over time, it is necessary to quantify the predictability, or level of cyclical constancy, of a system, and to understand how this changes with downstream distance. Most important in characterizing temperature regimes is to establish reference levels, typically considered to be those that would be expected if the system were operating without significant human influence; they are not necessarily desired conditions (McCammon et al., 1998).

2.2 Classification of variability

Classification is "a basic procedure for imposing order on the diversity of the real world" (Gordon et al., 1994, p. 410). The challenges of classifying system variability in ways that are ecologically relevant are ongoing, and are reflected in the numerous approaches currently under debate in the available literature. Broad classification of rivers includes that of Illies (1961, cited in Gordon et al., 1994), who proposed a generalized method of stream classification useful to ecologists. In this classification, rivers and streams are divided into a cooler, upper section (rithron) and a warmer, lower region (potamon). The point on a stream separating rithron from potamon is defined as the location where the monthly mean temperature is 20°C. However, this transition also depends on altitudinal and latitudinal gradients. This division has been further refined and subdivided into "zones" based on distinctness of ecological structures. Importantly, there is typically high faunal diversity in the transition regions where these zones intersect (Gordon et al., 1994). Pennak (1971) produced a list of 13 critical features considered to be easily measured and adequate to characterise the ecological similarity of stream reaches, which included summer and winter water temperatures.

On a more local scale, approaches generally attempt to measure variability by either agglomerating or decomposing time series data. Agglomerative approaches make use of techniques such as duration curves, while reductionist approaches make use of indices that focus on state and threshold values using descriptive statistics, and attempt to understand the links between timing, duration and magnitudes of different system states. For example, Harris et al. (2000) proposed characterizing flow and water temperature time series by regimes. This approach uses basic descriptive statistics, such as mean, maximum, minimum and variances of the time series to define the "shape" of the regime over time for a particular river or group of rivers. System change is measured when the "shape" of the regime curves change.

Frequently used indicators of water resources system performance are mean and variance (Hashimoto et al., 1982). However, such measures describe the average level and average squared deviation from the mean of the parameters in question, and fail to describe system variability adequately. Hughes and Hannart (2003) recognised the need to characterise the hydrological variability within South Africa's rivers in order to assist in determining the ecological Reserve, as stipulated in the Water Act (DWAF 1998). A hydrological index for all 1946 quaternary catchments was developed, based on monthly coefficients of variation. While this approach advances issues of water quantity, research on water temperatures lags considerably and temperature time series are not available to undertake similar analyses. Typically, approaches consider a range of metrics considered to be of ecological importance, and these are centered on measures reflecting magnitude, frequency, duration, timing and rate of change (Poff et al., 1997). The timing of events provides important cues to species, while the rate of change reflects system "flashiness", and duration of an event is often related to physiological stress and the species-specific tolerance to these stresses.

Richter et al. (1996) developed the "Indicators of Hydrological Alteration", an approach which analyses flow time series for ecologically significant measures, based on 32 parameters which are divided into 5 groups relating to magnitude, timing, duration and frequency of ecologically significant events. Within this approach, there is recognition of the necessity of intra-annual variation for successful life-cycle completion. These analyses rely on comparisons of pre- and post-treatment of systems, although in reality, even with multiple tests for data homogeneity (or not), it is not always easy to determine drift and subtleties. In addition to this, time series are often incomplete, and require considerable data matching and patching and associated computing expertise (see, for example, Schulze and Maharaj, 2004, Lynch, 2004, for derivation of continuous, patched time series of air temperatures and rainfall respectively). The IHA approach should be used in conjunction with other metrics, and provides a useful technique for ecological restoration programmes.

Colwell's indices (Colwell, 1974) are appropriate for highlighting hidden periodicities within time series that are of biological significance. Colwell's indices (Colwell, 1974) are a useful technique for classifying rivers into different types, such as "predictable, seasonal". Using such an approach, continuous data are categorized; while information may be lost, there are gains in interpretability (Colwell, 1974). These indices classify temporal patterns into a measure of predictability, based on the degree of constancy and contingency. Gordon et al. (1994) define constancy as a measure of the degree to which a state stays the same (the system – a river – is the same for all seasons), while contingency is defined as a measure describing how closely the different states correspond to different time periods (the state of the system is different for each season; but the pattern is the same for all years). In other words, contingency may also be defined as a parameter which measures the degree to which time and state are dependant on each other, i.e. a measure of periodicity (Archer, 2000). Therefore, a high level of predictability is achieved due to either a high degree of constancy or contingency (Colwell, 1974). Constancy is highest when the state is constant, while contingency is at a minimum when state is independent of season (Colwell, 1974). The predictability of any periodic phenomenon is maximal when there is complete certainty with regard to state for a point in time, such as when a tree will be in flower (Colwell, 1974). According to Colwell (1974), contingency is the degree to which time determines state, and is smallest when all columns are homogenous, or when system state is independent of the effects of seasonality; contingency is highest when the state of the river is different for each season, even though the pattern is the same for all years. These indices were used by Vannote and Sweeney (1980) to quantify the predictability of a river system in the United States based on daily average river temperatures over a ten-year period. This particular system was found to exhibit a pattern of "predictable variability" with respect to water temperature. Poff (1996) used Colwell's indices as one approach to classifying flow regimes in undisturbed streams in the United States in an ecologically relevant way.

Archer (2000) describes Colwell's indices as being "perhaps the most focused [of] existing methods of describing aspects of temporal fluctuation and have been applied both to physical and biological phenomena". However, two drawbacks in analyses using these indices is that computed values are constrained by the way in which class intervals are defined to characterize the states of the phenomena occurring, as well as record length (Gan et al., 1991; Gordon et al., 1994; Archer, 2000). Colwell's indices suggest that the degree to which a system is classified as predictable and seasonal depends on how system states are defined.

2.3 Water temperature signatures

With the focus on river heterogeneity signatures gathering momentum in the aquatic conservation field of South Africa, it is important to bear in mind that while rivers may be classified as having excellent habitat structure based on their flow and geomorphology signatures, Harris and Silveira (1995) point out that many thermally polluted sites have excellent habitat structure.

In conservation planning, the "range of variability" or "natural range of variation" approach has been advocated as a useful tool in adaptive management, for setting flow targets (Richter et al., 1997; Groves, 2003). Within this, the role of disturbance and variability is recognised in maintaining diversity (Richter et al., 1997). To apply this requires that the time scale and geographical area considered be specified first (Groves, 2003). Analyses typically are based on frequency distributions of physical and biological conditions. In a study by Archer (2000), which considered flashiness, based on the frequency and duration of pulses, mean daily values were too coarse to reflect this, and 15-minute intervals were used, which is particularly appropriate in small catchments and streams, so that data logging interval could be scaled by stream order. Temperature is a continuous climatological variable, which is both temporally and spatially conservative. Consequently, record lengths do not need to be as long as for other variables (such as precipitation) to evaluate most statistics with confidence (Schulze and Maharaj, 2004). However, to make predictions with any degree of confidence requires time series, which are often lacking. Also, the time series should be at the resolution appropriate to the question(s) asked.

An additional compounding issue, which will require further study, is the macrohabitat variation of water temperatures. In a river study in central Ohio, which included considerations of water temperatures and dissolved oxygen, it was found that "the amount of oxygen was the most variable water condition, and it varied little from riffle to run at the same time on any given day." (Murvosh, 1971) An understanding of water temperatures becomes critical in understanding dissolved oxygen, because of the inverse relationship between these two variables. In the same study, water temperatures of a riffle were 1.1°C cooler than the two other habitats considered [run and "intermediate" between riffle and run]. These parameters are practically identical during much of the year." (Murvosh, 1971). However, hourly water temperatures collected in pool and riffle habitat in the Great Fish River (Eastern Cape) were not significantly different from each other (Rivers-Moore et al., 2007).

2.4 A South African approach & contribution to the ecological Reserve

South Africa is a signatory of the Convention on Biological Diversity (Driver et al., 2004), and with this comes a legal and ethical obligation to protect biodiversity, as defined earlier. South Africa's response, in line with other signatory nations, has been to begin developing a National Biodiversity Strategy and Action Plan (NBSAP), led by South African National Biodiversity Institute (SANBI). Important enabling legislation for management and conservation of biodiversity includes:

- National Water Act (Act No. 36 of 1998) (Republic of South Africa 1998a)
- National Environmental Management Act (107 of 1998) (Republic of South Africa 1998b) – requires environmental management plans. Established a framework for Biodiversity and Protected Areas Acts.
- Biodiversity Act (10 of 2004) (Republic of South Africa 2004) – allows Minister/MEC to list threatened and protected ecosystems. Four categories of threat status are listed: critically endangered; endangered; vulnerable; protected ecosystems. NSBA provides input on identification of threatened ecosystems.

- Protected Areas Act (57 of 2003) (Republic of South Africa, 2003) – allows for land to be declared as a protected area.

The National Water Act provides legal status to the water required to maintain ecological functioning of river systems, through the declaration of the "ecological Reserve", which is defined as relating "***to the water required to protect the aquatic ecosystems of the water resource. The Reserve refers to both the quantity and quality of the water in the resource, and will vary depending on the class of the resource. The Minister is required to determine the Reserve for all or part of any significant water resource. If a resource has not yet been classified, a preliminary determination of the Reserve may be made and later superseded by a new one. Once the Reserve is determined for a water resource it is binding in the same way as the class and the resource quality objectives"*** (Republic of South Africa, 1998). The determination process recognises determining the Reserve for all or part of a water resource. Components of the proposed Reserve will go through a public participation process. To date, no proposals have been developed for the water temperature component of the Reserve, although the importance of water temperatures in maintaining river systems is now recognised.

An initial step in determining the temperature component of the Reserve would be to develop suitable indices on characterising time series of temperatures. This should consider some of the points raised earlier. Most notably, frequency, duration and timing of thermal periods needs to be related to different biota, and related to life histories of different species. Indices of predictability, with careful definitions of classes, need to be applied to life history patterns and stability of community structures (discussed further in Section 4). This would at least set initial recommended thresholds of variability and seasonality. A critical step is defining these by geographical region (also discussed in Section 4). Additionally, the ecological Reserve should also consider the importance of temperature extremes, notably daily maximum water temperatures, as well as their temporal predictability. Given the relationship between water temperature and flow volumes, the ecological Reserve should consider the particular relationship between, and significance of, extreme low flows and water temperature, and flow/temperature-biotic response stressor relationships. This may assist in prioritizing future areas of research. An additional factor which may be useful to the ecological Reserve and water temperatures is the spatial representation of vulnerability to thermal alteration, particularly under anticipated scenarios of global climate change. The approach of Richter et al. (1997), which attempts to describe flow series based on their "natural" range of variability, is a useful one, particularly when applied spatially (Richter et al., 1998). It is critical that aquatic management aims to preserve as much system variability as possible in order to protect freshwater biodiversity, with river systems classified regionally based on the key attributes and ranges of variability of component time series (Arthington et al., 2006).

2.5 Conclusions

The broad temperature classifications of potamon and rithron are suggested as being useful starting points to an initial water temperature contribution to the ecological Reserve.

Notwithstanding the lack of available data and the inherent problems with simulating time series in highly variable systems, it is nevertheless worthwhile to characterize system variability, since it is this aspect of a thermal regime that enables aquatic organisms with different tolerances to co-exist within a single reach of river. Based on the various techniques utilized within this study, the coefficient of variation is sensitive to extreme values, which characterize flow rate time series but not water temperature time series to the same degree. As a simple first approach the coefficient of variation is probably adequate in characterizing annual flow rates and water temperatures between rivers, which Poff (1996) considers a straightforward, less arbitrary measure of flow variability that shows more stability with respect to length of time series than does Colwell's index. Conversely, Colwell's indices are not sensitive to extreme values, and are in turn appropriate for detecting seasonality. Agglomerative techniques, such as the use of duration curves using successive days within class intervals have potential for defining management targets such as Environmental Flow Requirements.

What should emerge as a guiding principle is that a suite of different metrics should be used simultaneously to characterise water temperatures. In a country where river systems are highly variable, and often defined by extremes rather than averages, the importance of understanding variability and how this relates to the ecological Reserve cannot be overemphasized. Additionally, fundamental research needs to occur, to link aquatic macroinvertebrate diversity and productivity to water temperature variability along longitudinal axes of rivers, to test the degree of applicability of the RCC to South African river systems. To what degree do South African rivers follow patterns of northern hemisphere rivers? The answer to this will effect the degree to which water temperature research in South Africa will need to be innovative, versus adaptive, and this is discussed further in Section 3.

3. NORTHERN VERSUS SOUTHERN HEMISPHERE THERMAL PATTERNS

3.1 Introduction

It is unknown as to what degree South Africa's rivers have their own distinct thermal characteristics. Ward (1985), in comparing the thermal characteristics of northern hemisphere versus southern hemisphere rivers, concluded that what makes southern hemisphere rivers distinct from northern hemisphere rivers is "a matter of degree rather than of kind", i.e. South African rivers may have parallels in the northern hemisphere, but a greater proportion of these will be more variable than in the northern hemisphere. Chiew et al. (1995) have demonstrated that southern African rivers, like Australian rivers, have extreme flow regimes, displaying twice the world average of flow variability. This variability is reflected in the thermo- and hydrographs from such river systems, and associated time series typically exhibit non-constant variance. Basson et al. (1994), in a study on the reliability of water resources yields within the Vaal River system of South Africa, recognized that system variability between months in non-humid southern hemisphere regions presented management and simulation challenges not present in humid northern hemisphere regions. However, in spite of these complexities, previous research on the characterization of ecologically important water temperatures (for example Eaton and Scheller, 1996; Essig, 1998; Poole and Berman, 2001) can be used as the basis for analyses for detecting system change in South Africa's rivers, and in highlighting future areas of research on river signatures.

Often, thermal patterns in headwaters are dominated by groundwater (Vannote and Sweeney, 1980). It is thus important to consider regional groundwater variations to more fully understand thermal patterns. For example, seasonal differences may be more marked in headwater stream temperatures versus downstream tributaries as a result of groundwater inputs (cooler in summer and warmer in winter). Other metrics used by Vannote and Sweeney (1980) were cumulative monthly degree days, as well as maximum diel range, as a function of stream order and latitude.

The aim of this chapter is to calculate metrics, which agglomerate time series, to compare trends reported from northern hemisphere rivers with those measured from selected river systems in South Africa. The degree of similarity between southern and northern hemisphere thermal patterns will determine the degree to which advances in understanding of thermal regimes in the northern hemisphere can be applied to South African river systems.

3.2 Methods

Temperature time series from 20 sites in 4 river systems (Table 1) were analysed to provide preliminary insights on thermal signatures in selected South African River systems. Hourly water temperatures for the Salt River (Western Cape) were available through current collaborative research (Rivers-Moore and de Moor, unpublished data), as an unregulated river system with a unique aquatic invertebrate fauna (de Moor, 2006, *pers. comm.*). Two year's of hourly water temperature time series for the Sabie-Sand

River system, an unregulated highly variable system, are available from Rivers-Moore (2004), while one year of hourly water temperatures were also recorded from one site on the Great Fish River. Finally, six months of hourly water temperatures for four sites on the uMgeni River, a regulated river, were obtained from Dickens et al. (2008). Daily temperature statistics (mean, maximum) were calculated from the hourly water temperature time series.

Table 1: Information pertaining to 20 water temperature time series in 4 river systems

River	Site[1]	Stream order	Record length (months)	Latitude (decimal degrees)	Altitude (m.a.m.s.l.)
Sabie	Upper Sabie (1)	2	32	25.14	1193
Sabie	Sabie (2)	3	32	25.06	870
Sabie	Hazyview (3)	4	32	25.04	523
Sabie	Middle Sabie (4)	5	32	24.99	287
Sabie	Middle Sabie (5)	5	32	24.98	357
Sabie	Lower Sabie (6)	5	32	24.99	242
Sabie	Lower Sabie (7)	5	32	25.10	157
Marite	Hazyview	3	32	25.02	443
Sand	Skukuza	4	32	31.63	237
Fish	Coniston	4	12	33.08	300
uMgeni	Above Albert Falls Dam (1)	3	6	29.46	705
uMgeni	Below Albert Falls Dam (2)	4	6	29.43	615
uMgeni	Weir above Nagle Dam (3)	5	6	29.57	408
uMgeni	Between Albert Falls and Nagle Dams (4)	5	6	29.47	595
Salt	Kurland Estate – Top (1)	3	12	33.92	380
Salt	Kurland Estate – Pool (2)	3	12	33.93	300
Salt	Kurland Estate – Weir (3)	3	12	33.93	280
Salt	Kurland Estate – Pump (4)	3	12	33.93	280
Salt	Mansfield farm (5)	3	12	33.95	440
Salt	de Vasselot (6)	3	12	33.98	280

[1]Upstream/ downstream position on river longitudinal axis indicated by numbers in brackets

Note: Longitude was not considered as Vannote and Sweeney (1980) demonstrated that the relationship between annual degree days and latitude did not change significantly with longitude

To begin to understand how water temperatures in South African rivers may be different to water temperatures in northern hemisphere rivers, the following agglomerative/ descriptive metrics were used, and related to stream order, after Vannote and Sweeney (1980):

- Maximum daily range;
- Cumulative degree days per month.

Stream orders were assigned according to Strahler's (1964) stream ordering system (Chow et al., 1988). This provided a broad basis, supported by selected literature on water temperatures from northern hemisphere river systems, with which to contrast southern hemisphere water temperatures.

Additionally, predictability of water temperatures in relation to stream order were calculated using Colwell's (1974) predictability indices, as described in Section 2.2. Time (columns) by state (rows) contingency tables were calculated using the cumulative number of days of each month within different temperature classes, based on frequency counts. Daily average water temperatures were reclassified into water temperature classes (Table 2), based on *n* successive standard deviations on either side of the mean for observed water temperatures collected from February 2001 to March 2003 in the upper Sabie catchment. This site was chosen since it exhibited the least variability of the nine sites surveyed in the Sabie River system, and was assumed to allow for the definition of conservative temperature classes. Colwell's indices of predictability (*p*), constancy (*c*) and contingency (*m*) values were calculated for each site, based on the contingency tables. The percent contribution made to predictability either by constancy or contingency was calculated by dividing the predictability value by either index. A basic requirement of any data for use in Colwell's indices, particularly involving phenomena with fixed lower bounds, is that the standard deviation and mean are uncorrelated (Colwell, 1974). For example, with data that have a fixed lower bound (0), such as hydrological data, there is often a high correlation between mean and standard deviation. While in practice water temperatures do not generally have a fixed lower bound, these data were tested for correlations between annual mean and standard deviation, since a high correlation between the mean and standard deviation necessitates a log transformation.

3.3 Results

Total annual degree days in the four river systems considered ranged from 5200°C to 8700°C, with a difference of 3500°C (Figure 1). In the 2nd to 5th stream orders considered, the winter months exhibited the lowest monthly degree day accumulations, with monthly accumulations ranging between 300 and 900°C. In general, monthly degree day accumulations also increased with stream order. Monthly temperature accumulations also showed an inverse relationship with altitude (Figure 2a), although the relationship was not as marked for latitudinal gradients (Figure 2b). Similarly with the thermal trends relating to stream order, maximum diurnal change also increased exponentially with stream order (Figure 3). However, this relationship was weak (R^2 = 0.23), with few data points for low order streams. It appears that maximum diel range increased in variability with stream order.

Table 2: Water temperature classes used to reclassify simulated daily maximum water temperatures, based on observed mean and standard deviation values for site WT9 on the Sabie River.

Class	Upper	Lower
1	> 34.04	
2	34.04	32.01
3	32.01	29.98
4	29.98	27.95
5	27.95	25.92
6	25.92	23.89
7	23.89	21.86
8	21.86	19.83
9	19.83	17.8
10	17.8	15.77
11	15.77	13.74
12	13.74	11.71
13	11.71	9.68
14	9.68	7.65
15	7.65	5.62
16	5.62	3.59
17	3.59	1.56
18	< 1.56	

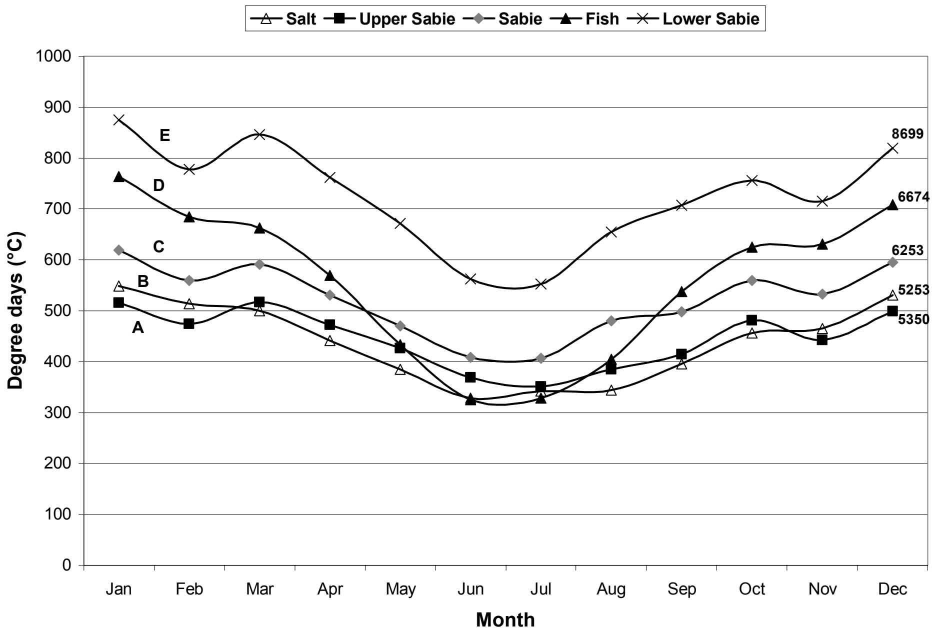

Figure 1: Distribution of monthly degree day accumulations for three South African rivers showing a range over four stream orders; A = 2nd order stream; B, C = 3rd order streams; D = 4th order stream; E = 5th order stream. Total annual degree days, as the sum of all month's degree days, are indicated on the right.

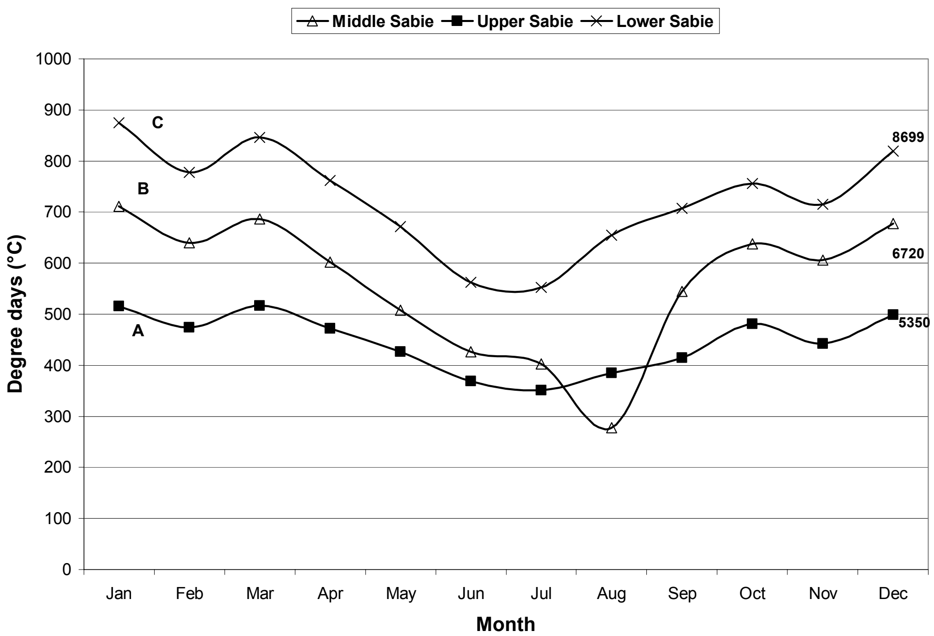

Figure 2a: Distribution of monthly degree day accumulations for the Sabie River over a range of altitudes; A = 1193 m, 2nd order stream; B = 523 m, 4th order stream; C = 157 m, 5th order stream. Total annual degree days, as the sum of all month's degree days, are indicated on the right

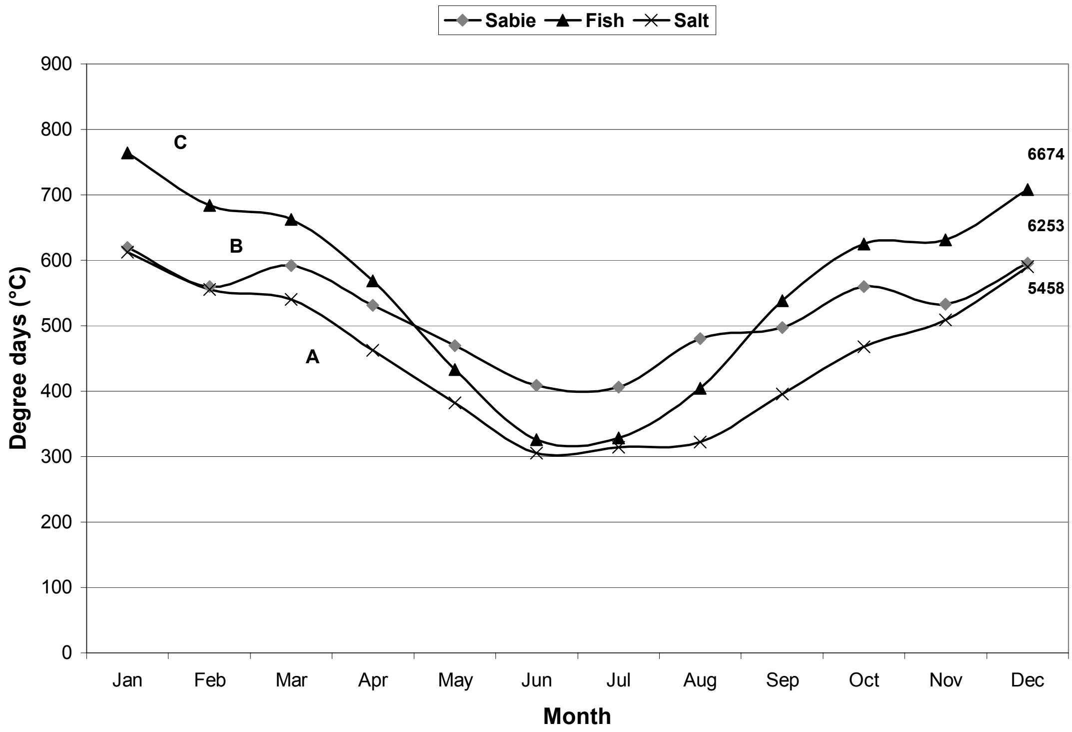

Figure 2b: Distribution of monthly degree day accumulations for three 3rd order South African rivers at different latitudes; A = 33.98° S; B = 25.06° S; C = 33.08° S. Total annual degree days, as the sum of all month's degree days, are indicated on the right.

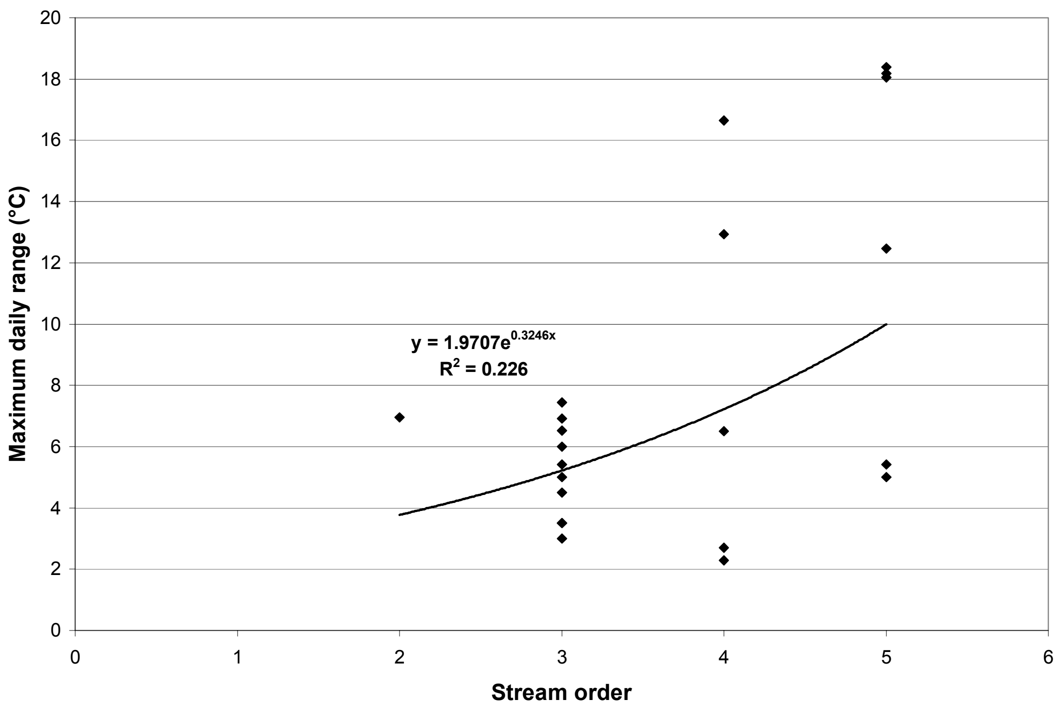

Figure 3: Maximum diurnal change in temperature as a function of stream order.

The correlation between the mean and standard deviation for the calculated water temperatures was non-significant ($p < 0.01$). No transformations of the simulated water temperature data were necessary, and untransformed temperature data were consequently suitable for subsequent classification using Colwell's indices. Predictability decreased with stream order (Figure 4), so that upper catchment rivers and tributaries had more predictable water temperature regimes higher order rivers. However, these data showed greater variability in the mid-order streams than in the higher and lower order streams. Water temperature predictability in the uMgeni River, albeit based on only six months of data, was higher than the other river systems considered.

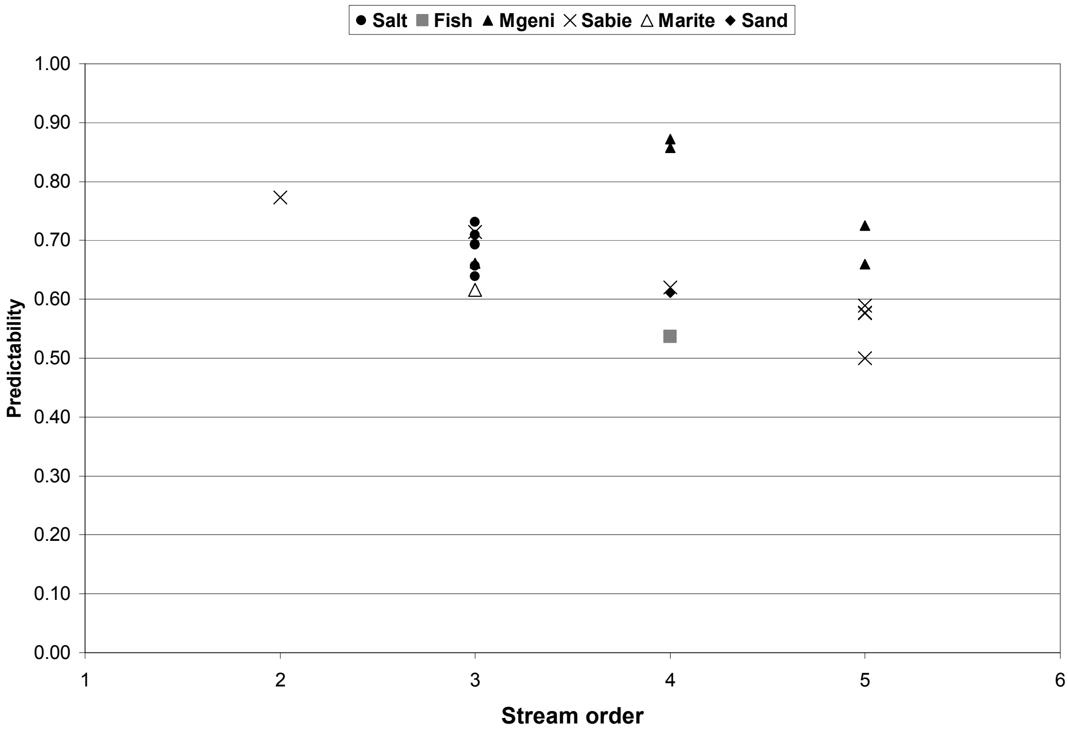

Figure 4: Colwell's (1974) predictability of water temperatures versus stream order

3.4 Discussion and Conclusions

In measuring water temperatures, a tradeoff exists between gathering high spatial resolution and high temporal resolution data, and requires developing complementary sampling approaches. In this preliminary analysis, high temporal resolution has been attained for a relatively short time period, while spatial resolution is fairly low. It is possible to record spatially continuous temperature data (for example using forward-looking infrared videography) (Torgersen et al., 2001), and complemented with continuous monitoring using data loggers (Torgerson, 2002, cited in Fausch et al., 2002). This is a consideration of time and budget, but should be borne in mind for future studies.

When compared to the figures of Vannote and Sweeney (1980), there are indications that there are differences in degree between northern and southern hemisphere river water temperatures. All rivers in this study were warmer than those considered by Vannote and Sweeney (1980), even though latitudes between the rivers considered in both studies were comparable. However, while the lower order streams in this study had lower amplitudes than the higher order streams, as found by Vannote and Sweeney (1980), they were generally cooler than the higher order streams, while in the study by Vannote and Sweeney (1980), the lower order streams generally remained warmer in winter due to groundwater inputs. One generalization which does seem to hold is that amplitude in degree days increases with stream order, as does maximum range in diel temperatures. A significant data gap in South African water temperatures is that very little is known on groundwater impacts on water temperatures; Vannote and Sweeney (1980) reported that streams with high groundwater inputs are less variable than streams with Hortonian flows, and that groundwater-fed streams have more stable temperatures throughout the year, i.e. cooler in summer and warmer in winter than higher order streams. Also, while these data differ annually by over 3000 degree days, with low order streams having much lower annual degree days than higher order streams, the figures of Vannote and Sweeney (1980) showed a difference of only 82 degree days between first and third order streams, with all values being similar (4222 – 4284). However, like Vannote and Sweeney's (1980) findings, diel variation increases with stream order. In this study, no data were obtained to test whether this variation again decreases after 5th order streams, as was reported by Vannote and Sweeney (1980). South Africa has relatively few "large", higher order rivers, with the Orange River being one notable exception.

All rivers in South Africa would be classified as temperate (23.5 – 66° latitude). One important difference between southern and northern hemisphere rivers is that most temperate northern hemisphere rivers drop to 0°C during winter unless they are groundwater-fed. This difference is largely due to the latitudinal differences in the distribution of land masses, with northern hemisphere rivers also flowing through much higher latitudes than South African rivers. Surface and subsurface ice is of very little importance in the ecology of southern hemisphere rivers (Ward, 1985). South Africa's highly variable rivers, with associated extreme flow fluctuations, typically have extreme temperature fluctuations, with marked year-to-year differences in thermal regimes (Ward, 1985). These data support this contention, with maximum diel range increasing, and predictability decreasing, with stream order. Lake (1982), in describing highly variable Australian river systems, hypothesized that unpredictable thermal regimes would have insects with flexible life histories, and flexible communities lacking highly synchronized species complexes. This decreasing predictability of water temperatures has important consequences for biota and their associated life histories (Section 2.1). Polyvoltine species have life histories already predisposed to unpredictable environments. A typical example is *Simulium chutteri* (Diptera: Simuliidae), which is an opportunistic colonising species. Univoltine species, on the other hand, rely on thermal cues for the transitions into different life history stages. In highly unpredictable systems, community structure is likely to be less clearly defined, as discussed earlier. The ratio of univoltine to polyvoltine species may be a useful index of system predictability.

This will require a deeper knowledge of life histories and species compositions, and linked to thermal data measured using suitable metrics of variability.

At this stage data are so limited that it is difficult to make generalizations regarding water temperatures for South African river systems. There are particular gaps in first-order streams, as well as in higher-order streams (5^{th} order and above). Additional data in targeted river systems, both regulated and unregulated (as far as possible) are required, and preferably for at least one full year, before further general patterns can be highlighted. From these data there are indications that water temperatures in regulated rivers are more predictable than in unregulated rivers, although these calculated are not based on a full annual temperature cycle. It would also be useful to compare South African water temperature data with time series from Australian rivers, as well as northern hemisphere rivers.

4. EFFECT OF MODIFIED THERMAL REGIMES ON AQUATIC INVERTEBRATES

The conceptual model of Vannote and Sweeney (1980) was used to evaluate the potential effects of modified thermal regimes on aquatic invertebrate communities. Such an approach has greatest practical applicability when a hierarchical approach is used, by examining the catchment-scale influence of land use on the relative contribution of water temperatures on aquatic invertebrate communities, with particular emphasis on appropriate bio-monitoring approaches. These are discussed in Sections 4.1 and 4.3 respectively.

4.1 A catchment-scale perspective

The heuristic framework first published thirty years ago, of Frissel et al. (1986) on the implications of scale in river systems, coupled to the River Continuum Concept of Vannote et al. (1980), are among the main pillars in current river management science. Coupled to this is a large and growing body of literature which emphasizes the importance of a catchment-scale perspective on river management (see, for example, Allan et al., 1997; Frimpong et al., 2005). Johnson and Gage (1997) highlight the role of GIS in the analysis of spatially referenced data, where remote sensing can provide information on the physical properties and temporal change in water temperatures. This has facilitated a deeper understanding of the growing recognition of the importance of hierarchy theory, patch dynamics and scale in understanding and classifying river systems.

Frissel et al. (1986) presented an organised view of spatial and temporal variation, as a spatially-nested hierarchical model. This provided a conceptual approach for classifying stream systems in the context of the catchment that surrounds them. In this approach, it is assumed that the structure and dynamics of the stream habitat is determined by the surrounding habitat (Hynes, 1975). Conceptually, then, "if…biological patterns in streams are largely adjusted to and controlled by physical patterns, the problem becomes one of understanding these physical patterns across time and space" (Frissel et al., 1986, p. 200). The central questions then become "How are streams organised in space?", and "How do streams change through time?" It is useful to place any classification of streams in a geographic, spatial hierarchy. Regional variation is caused by changes in geology, climate, soils, vegetation, and so on. Smaller-scale systems develop within the constraints set by the larger-scale systems of which they are a part. This then allows, theoretically, for describing system potential capacity (Warren et al., 1979), where the geomorphic features ultimately constrain how the system may develop. Concomitantly, management approaches should be at the correct scale of drivers. Microhabitats – with uniform depth, velocity, etc. – are useful for investigating behavioural ecology of fish and invertebrates. However, this remains largely as a useful academic perspective, with few examples of how this translates into true multidisciplinary management approaches to river systems.

The logistics of sampling all river systems within South Africa are clearly vast, and the current state of aquatic science assumes that abiotic variables are good surrogates for biotic patterns. Environmental parameters in the catchment area serve as an explanation for the variation in invertebrate distribution (Richards et al., 1996). Omernik's (1985, cited in Johnson and Gage, 1997) ecoregions have gained acceptance by aquatic scientists. Fish, macroinvertebrates, water quality and physical habitat have been shown to respond to regional patterns delineated by ecoregions (Whittier et al., 1988; Lyons, 1989, cited in Johnson and Gage, 1997). A good working definition of ecoregions is that they "integrate physical and/or biological data with the objective of identifying a homogenous unit based on a set of rules" (Higgins et al., 2005).

In a study by Higgins et al. (2005), an ecosystem approach for freshwater classification and conservation planning was used in which the region was divided into subcatchments, or ecological drainage units. Climate, catchment boundaries and landscape all exerted an influence on the broad patterns of aquatic ecosystem characteristics, including hydrological and thermal regimes. Frimpong et al. (2005) highlight the usefulness of evaluation of streams without field measurements, which will benefit stream managers. The length: slope factor is of particular importance because of its indication of sediment levels (Frimpong et al., 2005), since typically sediment yields increase with agricultural expansion (Allan et al., 1997), which in turn impacts on, *inter alia*, water temperatures and light penetration, which has a knock-on effect on primary productivity and invertebrate communities. In the study by Frimpong et al. (2005), catchment-scale habitat variables had good predictive power of stream health. Smaller catchment units, in comparison to larger ecoregions, improved the predictive power of such models. In related studies by Stein et al. (2002), it became clear that landscape-level assessments were needed for policy development, planning and management. The ecoregion approach is particularly suitable in data-sparse regions, since it is adaptive and initially only requires basic topographic and land use coverages. In their study of Australian rivers, as with South African rivers which are characterised by high flow variability and extreme events, very few large rivers remain undisturbed. The cumulative impacts of human activities downstream become important, which can be difficult to detect and may take years to occur. This is one of the unknowns within conservation planning (Groves, 2003), and has been termed "extinction debt" (Tilman et al., 1994).

In many cases, insights are gained through studies at a "riverscape" scale, and which would not have been obtained using a random sampling approach. Baxter and Hauer (2000) showed that location of redds of fall spawning bull charr (*Salvelinus confluentus*) were associated with low-gradient bounded alluvial valley segments, which in turn were zones of upwelling groundwater creating favourable thermal conditions for egg incubation during winter. In another example, Torgerson et al. (1999) showed that the distribution of spring-run Chinook salmon (*Oncorhynchus tshawytscha*) was related to large-scale thermal patterns of stream temperature and pool frequency. In a third example, Labbe and Fausch (2000) showed that within a system of pools which either dried or froze, populations of the Arkansas darter (*Etheostoma cragini*) were able to persist in a highly patchy environment through long-distance dispersal. It is increasingly recognised that riverscapes are strongly influenced by the landscape matrix through which they flow, in multiple dimensions (longitudinal; vertical = groundwater/surface water interactions, lateral = floodplain and temporal). Consequently, rivers need to be conceptualized as “spatially continuous longitudinal and lateral mosaics” where heterogeneity becomes an important study focus, rather than sampling points and gradients (Fausch et al., 2002). The River Continuum Concept remains a useful tool at a larger scale in understanding gradients, but these should be complement by also considering heterogeneity and patch dynamics.

Freshwater conservation planning in South Africa is already established on a national foundation, with a Level I Ecological Classification, based primarily on physiography, but also altitude and vegetation types, already established (Kleynhans et al., 2005). Air temperature metrics are also included in this classification. It is anticipated that ecological Reserve determinations will be based on at least a Level II ecoregion delineation, and it will be at this level of detail that water temperatures should initially be understood and classified.

4.2 Thermal modifications and aquatic biota

4.2.1 Catchment-scale impacts as modifiers of thermal regimes

Changes in land use are cited as having the biggest impacts on hydrology (Grubaugh and Wallace, 1995). Hydrological regimes, and by association water temperatures, are being increasingly altered by land and water uses (Richter et al., 1996). Agricultural practices often result in increased sediment loads, which impact on light penetration, albido effects and water temperatures. Allan et al. (1997) found that the best single predictor of local stream conditions was the extent of agricultural land at the subcatchment scale. This is because sediment levels increase dramatically with agricultural expansion, which has clear impacts on functional feeding groups.

A change in land use represents a change in runoff, which typically changes the biota which has evolved under a different flow regime. This will in turn impact on thermal patterns, where it has been demonstrated that flow volumes impact on thermal patterns (Rivers-Moore et al., 2004; Rivers-Moore et al., 2005). For example, a decrease in interannual variability is likely to have a positive impact on introduced species which have evolved under more stable thermal and flow regimes. South African rivers, together with Australian rivers, have been shown to exhibit high levels of variability. Low flow periods often result in increased water temperatures and reduced oxygen levels (Schlosser, 1991). In such situations, predation by birds and mammals increases (Lowe-McConnel, 1987, cited in Schlosser, 1991). This highlights the link between anthropogenic landscape disturbances and river health. Landscape activities tend to simplify structure. Schlosser (1991) also stresses the importance of heterogeneity as refugia. According to Schlosser (1991), we know considerably less about large lotic systems than streams because of the difficulties of sampling in large rivers. Land use activities in upper catchments tend to be more detrimental than further downstream because of the cumulative downstream effects of rivers.

Changes to thermal regimes are likely to impact on biotic integrity, which Grubaugh and Wallace (1995) define as taxonomic and functional structure. Agriculturally impacted streams have shown reductions in the less tolerant taxa (Ephemeroptera, Trichoptera and Plecoptera), and associated increases in richness and abundances of more tolerant taxa (Chironomidae and Simuliidae). To more fully understand the possible consequences of changes to invertebrate communities, reach-scale studies are necessary to characterise the taxonomic composition and functional structure (abundance, biomass and secondary production) (Grubaugh and Wallace, 1995).

4.2.2 Disruptions to the river continuum

Impoundments represent altered flow regimes, which in turn result in altered thermal regimes (Gordon et al., 1994). As Vannote and Sweeney (1980) have demonstrated, aquatic invertebrate communities reduce inter-specific competition through temporal partitioning based on differing thermal optima. Changes to thermal regimes are likely to result in changes to thermal triggers and hence in changes to seasonal aquatic invertebrate community patterns. For example, in a study by Murvosh (1971) it was concluded that the northern distribution limits of the water penny beetle *Psephenus herricki* (Coleoptera) are probably governed by lower water temperatures that would restrict feeding by the larvae and the completion of the life cycle. Not only will this probably affect river signatures (Schael and King, 2005), but it may also affect ecosystem processes, based on nutrient cycling.

According to Covich et al. (1999, p. 125), "the integrity of the freshwater supply depends on how various benthic species make their living and contribute to complex food webs". The Redundancy Hypothesis states that overlap in resource use among species is not unusual, especially in freshwater food webs. As long as each functional group is present, ecosystem processes will continue. Functional feeding groups have been shown to remain stable in spite of community changes (Schael and King, 2005), so the impacts may be mitigated. However, Covich et al. (1999) feel that redundancy in many freshwater benthic ecosystems is low. Research on biotic redundancy in South African river systems may give some ideas on resilience versus vulnerability. Ecosystems composed of a bare minimum of species are unlikely to continue to function merely by compensating (Covich et al., 1999). The spatial and temporal distributions of benthic invertebrates suggest that many species have different preferences for particular ranges such as temperature and velocity.

Regulated discharges result in more stable water temperatures, since dams stabilize hydrographs, and rivers downstream of impoundments typically don't go through the daily cycle of warming and cooling (Gordon et al., 1994). In a rare pre- and post-impoundment study downstream of van der Kloof Dam, Pitchford and Visser (1975) looked at looked at water temperatures on the Orange River, with a view to predicting the possible spread of *Schistosomiasis* vector snails. The temperature range was reduced from 19.6 degrees to 12.8 degrees, and seasonal effects were delayed by thermal inertia of the reservoir water mass: winter temperatures were elevated and summer temperatures decreased. In the same study, marked differences emerged, particularly with respect to reduced variability, maxima and reduced extremes (Visser, unpublished data in Ward 1985). Given the relationship between diel temperature range and stream order (Section 3.3), the relative position of dams within a catchment should be considered – impoundments on higher order streams are likely to impact water temperatures more s than in upper catchments. Additionally, large bodies of lentic water undergo thermal stratification, so that top-release versus bottom-release dams will have a profound impact on thermal signatures downstream of impoundments. In studies by Palmer and O'Keeffe (1990) on the downstream influences of an impoundment on macroinvertebrates in the Great Fish River, it was observed that reset distance was related to water temperature regimes, with reset distances being shorter under warmer temperature regimes. Thus bottom releases from impoundments in the upper catchment are likely to have a greater impact on aquatic macroinvertebrate fauna than similar impoundments in higher order rivers, because of cooler temperatures. Rivers-Moore et al. (2004) illustrated that within the Sabie River there was non-significant thermal mixing in pools of up to 1.5 m deep, so that in rivers with good flows suitable mixing will also reduce reset distances. It would be interesting to investigate relationships between flow volume, flow velocity and water temperatures on reset distances and thermal stratification in rivers and how these change seasonally, and link this to changes in invertebrate communities.

The serial discontinuity concept of Ward and Stanford (1983) provides a theoretical framework for testing hypotheses on the impacts of impoundments on river health. Davies et al. (1989) looked at the effects of impoundments on water quality and quantity on the Palmiet and Buffalo Rivers, and also on reset distances, and found the following:

- Decrease in spot and annual temperature ranges downstream of large dams. This is more variable for smaller dams;
- Dam position determines impacts;
- Large impoundments cause greater impacts than small impoundments;
- Bottom-releases reduce temperature ranges;
- No effects common to all dams.

Inter-basin transfer schemes also impact on water temperatures, primarily through reductions in ecosystem variability and increases in flow volumes: Snaddon and Davies (1998, 2000) list a number of ecosystem responses in recipient systems to impacts including decreased flow variability, unseasonal increase in discharge, and altered aquatic macroinvertebrate community composition and hydraulic biotopes. Davies et al. (1989) illustrated 15 possible forms of IBT, but if pulsed versus constant or seasonal versus aseasonal deliveries are added, there are up to 60 permutations. Disruptions to the continuum are inevitable – for example, a cold headwater transfer to a warmer mid-reach will have a temperature recovery component, or a multiplying effect.

4.3 Biomonitoring approaches

A river reach's thermal regime is optimal for a particular species when body weight and fecundity are maximised (i.e. inversely related to temperature). Variation in body size and fecundity are a phenotypic response to optimal versus suboptimal water temperatures (Vannote and Sweeney, 1980). Thus the greatest subpopulation biomass will occur at the location of the thermal equilibrium location.

Non-optimum habitats will be characterised by low population density and decreased fecundity, which translates into lowered competitive ability. This in turn increases the chances of local extinctions, which is an important consideration in conservation planning (Vannote and Sweeney, 1980).

Also, adult body size is inversely related to ambient stream temperatures during larval growth. This is critical to understanding size variation and the distribution of aquatic insects, because river temperatures vary spatially and temporally (Vannote and Sweeney, 1980). Large size differences typically occur between winter and summer cohorts of polyvoltine species, so that adults in rivers with thermal regimes warmer or cooler than optimal tend to be smaller and less fecund than adults in a thermally optimum environment. For example, de Moor (1982, 1994) has illustrated this for the pest black fly *Simulium chutteri*, where winter larvae are typically much larger than summer larvae, and emergent spring adults are also considerably larger than at other times of the year. One of the direct consequences of this is that larger larvae are more fecund as adults than smaller larvae adult females. A practical application of this is that spring emergent adult females of *S. chutteri* are highly fecund, and the best timing for control periods is during winter before these adults emerge. In experiments between body size and temperature, Vannote and Sweeney (1980) showed how adult body size is reduced when larvae were reared at both high and low water temperatures, but with food levels kept constant. From such experiments the general principle emerged that intermediate diel fluctuations were best, and extremes were not preferable. Furthermore, seasonal temperature patterns are a critical factor in maintaining temporal segregation of invertebrate communities, so that related species temporally segregated all perform the same functions at different times. Thus resources are partitioned through seasonal water temperature variation, and temperature patterns influence both the stability and number of species in a given reach (Vannote and Sweeney, 1980). This spatial, temporal and trophic segregation and specialization of invertebrates ensures the efficient annual use of stream resources.

Such sensitivity to thermal regimes makes aquatic macroinvertebrates useful in biomonitoring programmes. Certain invertebrates respond strongly to environmental gradients, which makes them potentially good indicators of hydrological degradation. For example, benthic invertebrates are useful biological indicators of hydro-morphological river quality, integrating and reflecting stream degradation and riparian land use (Lautenschläger and Kiel, 2005). Blackflies are particularly useful as indicators, where studies have demonstrated that blackfly composition is controlled by environmental variables sensitive to anthropogenic pressures (Feld et al., 2002; Zhang et al., 1998). For example, Lautenschläger and Kiel (2005) argue that since the larvae and pupae of several blackflies have specific flow requirements, they should be good indicators of morphological degradation. In their research, they found a 78% species-environment correlation using redundancy analysis (RDA) within a canonical correspondence analysis (CANOCO 4.5) for blackfly fauna. Hydraulic (depth, relation of pools and riffles, and diversity of current velocity) and hydrochemical (conductivity and oxygen level (related to water temperature)) explained 45 and 50% respectively of the blackfly distribution in a study of 57 streams in the European Central Highlands at a reach scale. At the catchment scale, 69% of the variation was explained by stream type information (surrounding land use, bank structures, cross profile, channel bed structures, longitudinal profile, stream course development). Generally, large-scale environmental factors (stream type and land use) govern large Simuliid distribution patterns, while these are overlaid by the "direct dependency of the Simuliids on water and habitat quality". Similarly, Rivers-Moore et al. (2007) found good species-environment correlations for *Simulium chutteri* larvae in the Great Fish River, as well as for 11 macroinvertebrate taxa (Baetidae, Leptophlebiidae, Gomphidae and three additional species of Simuliidae). In all cases, the major environmental gradient was a width/velocity one, at the macrohabitat scale. However, certain species groups do not show strong responses to environmental gradients, such as metacommunities of Chironomidae (Heino, 2005). The species groups which show significant responses to environmental gradients could potentially act as a useful, manageable subset of taxa which indicate the desired ecological condition. For example, Lawes et al. (2005) used canonical correspondence analyses to identify a group of unrelated macroinvertebrate species which serve as a suite of biological indicators of the condition of Afromontane forest.

As a general rule, there is a worldwide resemblance of component insect fauna (Vannote and Sweeney, 1980), which potentially means that metrics developed elsewhere should be broadly applicable within South Africa. However, Schael and King (2005) found that while these groups within macroinvertebrate communities remain relatively constant between rives and along river axes, each river system has its own unique macroinvertebrate "signature" due to unique combinations of species. Consequently each system should also be recognised as unique. Water temperature, catchment characteristics and past geology all contributed to an unknown degree to explaining these patterns (Schael and King, 2005). Water temperature is an important parameter which may be a contributing variable in differentiating tributaries both within and among drainage basins (Vannote and Sweeney, 1980). Tributary junctions represent natural discontinuities in the river continuum, with discrete changes in stream macroinvertebrate communities (Bruns et al., 1984). Furthermore, the composition of aquatic invertebrate communities changes seasonally with a reach, from reach to reach, and along the longitudinal axis of a river. Thus water temperature patterns lead to an array of invertebrate species and relative abundances within a reach.

Numerous indices have been developed to assess river health, and measure the impacts of humans on such systems, with aquatic insects being among the most widely used organisms (Bonada et al., 2006). It is important to bear in mind that all of these indices measure relative differences in environmental quality, and have no absolute meaning (Harris and Silveira, 1995). Furthermore, such indices assume a linear response to degradation, and while one of the important attributes of a biotic index is indeed a linear response, Allan (2004) illustrates an additional three hypothetical non-linear responses. Certain higher-level indices (benthic secondary production and leaf-litter decay) are useful in measuring integrity of ecosystem processes, whose intactness is a necessary part of biodiversity conservation through persistence (Margules and Pressey, 2000). However, secondary production and decay rates are a function of various natural properties, including water temperatures (Bonada et al., 2006). Such indices may be coupled with invertebrate biomass and functional feeding groups, which have been shown to be one of the components of the RCC which apply to southern African rivers (Palmer et al., 1991). While these indices remain a useful approach to measuring river health, their utility value remains hampered without a more complete understanding of river temperatures.

Groves (2003) lists a number of weightings of species to reflect environmental gradients, which are popular with ecologists and well entrenched in the ecological and conservation literatures. These include the use of keystone species (i.e. those species upon which whole ecosystems depend), indicator species (which are assumed to reflect ecosystem stress and process) and umbrella species (where it is assumed that through recognising their habitat requirements and conserving them, the system will and all associated biota will be conserved). While all of these measures have value, the use of any one of these should be after careful consideration based on whether such species choices truly act as barometers of the system (Groves, 2003).

4.4 Ecological Reserve

South Africa is a conservation planning "hotspot" (Knight et al. (2006). Conservation plans should provide a "scientifically sound, and therefore defensible, basis for land-use decision making (Knight et al., 2006, p. 5). Setting conservation goals becomes necessary in order to achieve representation and persistence. To this end, relevant spatially explicit data at the appropriate scales is necessary to identify regions of importance. According to Knight et al. (2006), "the lack of spatially explicit data on environmental processes is a …hindrance" (p. 6). Spatial layers showing transformation and predicted future pressures are relatively expensive (Knight et al., 2006). However, under conditions of anticipated climate change, this becomes necessary. For example, a study on the potential impacts of increased water temperatures in the United States, Mohseni et al. (1999) showed significant impacts on fish distributions, particularly stenotopic cold-water species. Concomitantly, aquatic macroinvertebrate disease vectors (such as *Anopheles* mosquitoes) may enlarge their ranges under conditions of warmer water temperatures, with associated societal costs.

Altitude is a variable of overriding significance in determining water temperatures (Ward, 1985). In an aquatic ecoregion classification for KwaZulu-Natal, Graham (2006) showed that altitude and temperature were overriding variables in defining biotic zones. Roux et al. (2002) developed a conservation plan for the rivers in Addo Elephant National Park, using landscape attributes as surrogates for biodiversity patterns. It is assumed that stream biota are protected if habitat heterogeneity and pattern are conserved (Roux et al., 2002). River signatures were identified using hydrology, ecoregions and geomorphology. However, the resolution of these river signatures could be improved if water temperature data were able to be incorporated into subsequent conservation planning processes. Thus existing ecoregional maps would most likely be suitable to water temperature zones too. According to Harrison (1965), "using temperature as a major factor for the primary division of rivers into rithron and potamon segments is generally applicable to the running waters of southern Africa."

Conservation of diversity is often leveraged using targets – % area required to conserve x % of species, based on species-area curves. An equivalent approach in river systems is to use species-discharge curves as a conservation planning tool (Xenopoulos and Lodge, 2006). Flow rates are reduced by water abstractions, as well as anticipated climate change, which in turn will impact on water temperatures. Species-discharge model's predictive power will be increased by the inclusion of other factors in addition to discharge, including water temperatures. As an illustrative example, reduced flows lead to increased water temperatures, and increased eutrophication, which in turn facilitates establishment of parasites and fosters infections (Steedman, 1991). Thus determining ecological temperature requirements, linked to discharge, becomes a critical part of the ecological Reserve determination process, since maintaining these becomes critical in reducing the spread of diseases and parasites. Wilson et al. (2005) studied the effects of water temperature, conductivity and turbidity on the efficacy of Vectobac® on *Simulium damnosum*. Water temperature was found to be the most significant of the three parameters, possibly due to its indirect effects on metabolism which leads to increased ingestion rates of toxins by invertebrates. For larvicidal control programmes, this implies an optimal temperature range for toxin uptake.

4.5 Conclusions

In order to take research on aquatic macroinvertebrate communities into the realms of understanding processes, it is recommended that different functional feeding groups be sampled for all seasons at different localities. As mentioned in Section 4.3, one approach is to compare ratios of less tolerant taxa (Ephemeroptera, Plecoptera and Trichoptera) to more tolerant taxa (Chironomidae and Simullidae). Pertinent questions are how predictable, and how structured are South African aquatic communities? What is the theoretical level of redundancy in aquatic macroinvertebrate communities? To answer the former question, it is necessary to collect hourly stream temperatures and dry weights of different invertebrate species, in order to maximum diel range of temperatures, and thermal optima for different species. Answers to the latter question may provide further insights into the nature of macroinvertebrate river signatures, but may also open the danger of how far a river may be degraded and still function – i.e. clearing down to the target.

Research should be conducted at appropriate scales, since the importance of different physical and ecological processes will be revealed at different spatio-temporal scales. Also, rare or unique features, in space or time (waterfalls at critical nodes) are often of great significance in linear systems such as rivers (Fausch et al., 2002). Effective conservation requires research and management questions at scales that match the life histories of species involved. This will require a shift in thinking and in sampling approaches (Fausch et al., 2002). Research has often focussed on small spatial and short temporal scales, when conservation problems are typically at much greater scales. To this end, it is advocated that river systems be studied at an intermediate scale – equivalent to terrestrial landscape ecology – the "riverscape" scale.

As a useful guiding principle, multifactor approaches are necessary to understand causes of individual dispersal; an understanding of this may assist in predicting responses at higher organisation scales (Holomuzki et al., 1999). What remains to be researched is the careful selection of different groups of taxa, based on their sensitivity to temperature, and their reflection of taxonomic diversity versus ecological process.

5. WATER TEMPERATURE MODEL – CONCEPTUAL DEVELOPMENT

5.1 Introduction

Water temperature is a measure of the energy fluxes in a river, and reflects the energy exchanges with the catchment. Energy is the currency of any ecological system, including all freshwater systems (Reynolds, 1998), and the continual renewal and release of energy down a river system aids in balancing the energy budget. Webb (1996) breaks the energy budget of a river into six main components (Equation 1):

$$Q_n = \pm Q_r \pm Q_e \pm Q_h \pm Q_{hb} \pm Q_{fc} \pm Q_a \qquad [1]$$

where Q_n = total net heat exchange; Q_r = heat flux due to net radiation; Q_e = heat flux due to evaporation and condensation; Q_h = heat flux due to sensible transfer between air and water; Q_{hb} = heat flux due to bed conduction; Q_{fc} = heat flux due to friction; Q_a = heat flux due to advective transfer in precipitation, groundwater, tributary inflows, streamflow and effluent discharges.

The thermal inputs of a river (solar radiation and surface friction) are in dynamic equilibrium with thermal losses through heat transfer processes such as evaporation (Bartholow, 1989). A consequence of this dynamism is that water temperature varies along the longitudinal axis of a river, on a seasonal and daily basis (Webb and Walling, 1985; Allan, 1995), with diel fluctuations superimposed on seasonal and annual cycles (Webb and Walling, 1985). The magnitude and speed of these changes over space and time are a function of many variables, which operate at different spatial and temporal scales.

Water temperatures are thus primarily a function of thermal inputs minus thermal losses (i.e. a heat exchange coefficient), which are termed “drivers” of water temperatures. Constant heat inputs through solar radiation and surface friction and heat losses through evaporation are in turn moderated through “buffers” which include shading, channel form, latitude, altitude and flow (groundwater and surface water) (Ward, 1985). These variables all interact at different scales to produce water temperature signatures with distinct sinusoidal patterns at different spatial and temporal scales. For the ecological reserve, micro-scale water temperatures (sub-daily) are unlikely to be of importance. Rather, water temperatures at the meso-scale (10 m^2 – 10 km^2; days) become of greater importance. At this scale, water temperatures are a function of the flow regime, as well as the water velocity (residency time). Owing to increases in water volume with downstream distance, the significance of the thermal lag increases with downstream distance (Rivers-Moore et al., 2004). Stefan and Preud’homme (1993) report time lags of up to 2-3 days. However, the effects of these time lags are very much scale-dependant, and still apply at a daily time step. For example, daily maximum water temperatures in the Great Fish River exceeded daily maximum air temperatures on days succeeding a day where daily maximum air temperature was relatively high (> 35°C) (Rivers-Moore, 2007, *pers. obs.*) Also at this scale, the buffering effect of the hyporheic zone is regarded as the most important buffer (Poole and Berman, 2001). Water temperatures cannot be fully understood without due consideration of the hydrograph, although Smith (1972) regards the **volume** of water as the most important factor. At the macro-scale (>10 km^2; years), latitude and altitude become of increasing importance. It is at this scale that the usefulness of an ecoregional classification becomes clear, where each ecoregion is associated with different parameters of relevance to water temperature simulations. An alternative view of this is to associate thermally homogenous river segments (thermal reaches) with ecoregions.

Water temperature models are important in simulating water temperatures, particularly in cases where observed data are scarce, where surrogate driver data are available, and in situations where decision-makers are concerned with environmental scenario analyses (Rivers-Moore and Lorentz, 2004). As discussed in Deliverable 1, a consideration of water temperatures should form part of the ecological Reserve determination. However, because of the general lack of water temperature data, simulation modelling of water temperatures, with associated scenario analyses, is the most suitable approach to incorporating water temperatures into ecological Reserve determination studies, using the recommended ecoregion approach. Of greatest importance biologically are daily maximum water temperatures. However, also of biological significance are daily range, mean and standard deviation, which are all related to timing, magnitude and duration of thermal events. However, these latter categories rely on sub-daily (hourly) water temperature models (Rivers-Moore and Lorentz, 2004). Thus the approach adopted is very much a function of the temporal and spatial scale chosen.

5.2 Methods

Water temperature data described in Section 3 were used for analyses in the water temperature modelling exercises in this report. Hourly water temperature data for 20 sites in four river systems (Salt, uMgeni, Great Fish and Sabie-Sand Rivers) were used. Water temperature sites covered a range of stream orders and river latitudes and altitudes. The geographical location of data sites is shown in Figure 5.

Figure 5: River systems for which water temperature data were obtained for the water temperature model development

5.2.1 Assessment of model variables

Using the water temperature data from Section 3, multiple linear regression modelling was undertaken to determine which variables contributed most to water temperature signatures. Four analyses were conducted using standard, stepwise forward and stepwise backward techniques (StatSoft, 2003) to determine the relationship between maximum daily water temperatures and other measured parameters, namely air temperature, flow and relative humidity. As the dataset from the Sabie River was the largest, half of the data were used for developing the model (in addition to data from the Great Fish, Salt and uMgeni Rivers), and the other half of the Sabie River data for assessment of the four models. Therefore, the models were based on Sabie River water temperature data collected between February 2001-September 2002 and all the data for the other rivers for which air temperatures were also available. The assessment of goodness of fit for the four models developed was done on Sabie River water temperature data collected between October 2002 and October 2003.

Model accuracy was calculated as the mean (± standard deviation) percentage difference between observed and simulated maximum daily water temperature. Due to the potential albedo effect (fraction of incident radiation reflected by a surface) on water temperatures, it was also considered necessary to include an analysis on the possible relationship between water temperatures and total dissolve solids (TDS) as a surrogate for turbidity. TDS data, collected between 1997 and 1999 as part of the River Health Programme (DWAF, 2002a), were available for four sites along the Sabie River which corresponded with four of the water temperature monitoring sites used by Rivers-Moore et al. (2004) between 2000 and 2003 (upstream and lowveld sites). It was not possible to determine directly whether a correlation existed between TDS and water temperatures, since collection dates for both variables did not overlap. The only approach at this stage was to compare regression slopes between mean daily air and maximum daily water temperatures at sites where high TDS had been recorded and sites where low TDS values had been recorded, as an indirect measure.

In addition, the 20 sites in the four river systems (11 470 water temperature records) were characterised by the 16 variables listed below:

- Annual mean temperature
- Annual standard deviation
- Annual coefficient of variability
- Predictability (Colwell, 1974)
- Absolute minimum
- Absolute maximum
- Mean daily minimum
- Mean daily maximum
- Average diel range
- Maximum diel range
- Mean spring temperature
- Mean summer temperature
- Mean autumn temperature
- Mean winter temperature
- Julian date of annual minimum
- Julian date of annual maximum

Prior to undertaking a principal components analysis (PCA), correlations between variables were tested for in order to eliminate redundant variables (StatSoft, 2003). A PCA was performed using PC-ORD (McCune and Mefford, 1999) to determine which variables had the greatest influence on structuring site clusters.

In this analysis, a correlation matrix was used, since there was little "*a priori* basis for deciding if one variable is more ecologically important than the other" (McGarigal et al., 2000). Site groupings were also examined using cluster analysis techniques to assist in the interpretation of PCA scatterplots (Euclidean distance measure; un-weighted pair-group averages) (McCune and Mefford, 1999).

5.2.2 Conceptual model development

The review of Rivers-Moore et al. (2004) on physical versus statistical modelling approaches applicable to water temperatures was extended and evaluated. This review was undertaken with reference to target groups (such as ecologists), which would dictate scale and type of models adopted.

The potential for developing a physically-based statistical model incorporating a heat exchange coefficient within a dynamic water temperature model, as proposed by Rivers-Moore and Lorentz (2004), was investigated. In this model, the conceptual approach was that solar radiation (using air temperatures as a surrogate) acted as a heat pulse applied to water, using a convolution integral. Additionally, the possibility of using low flow effects, and additional site-specific thermal pollution effects (for example, bottom releases from dams, cooling tower blow downs) to enhance the utility value of such a model was investigated. Within this model, a heat exchange coefficient was recommended to dampen the response of water temperatures to air temperatures, which is related to increases in water volume with downstream distance.

5.3 Results

5.3.1 Assessment of model variables

All the four multiple linear regression model sets assessed, based on standard, stepwise forward and stepwise backward methods, provide similar results. The air temperature parameters (mean and minimum daily air temperatures) were significant, while the flow and relative humidity terms were not significant and were therefore excluded from the models (Table 3). Model verification showed that water predications were, on average, better the larger the sample size.

TDS values in the lowveld reaches of the Sabie River were almost four times as high as those in the upper reaches of the Sabie River (> 93 mg/ℓ versus 25-26 mg/ℓ). The regression slope at the lowveld site was 2.4 times steeper than the upstream site (1.063 vs. 0.445), suggesting that different factors were affecting the relationship between air temperatures and water temperatures at the different sites. These data do not show causation, although inclusion of a turbidity term may account for more variability in the water temperature data.

Correlation analyses showed that mean minimum and mean maximum water temperatures were highly correlated ($R^2 = 0.98$) with annual mean water temperatures, while mean maximum water temperature was also highly correlated ($R^2 > 0.96$) with mean summer and mean autumn water temperatures. Consequently, mean maximum and annual mean water temperatures were excluded from the principal components analyses. Additionally, since annual standard deviation was used to derive the coefficient of variability, this variable was also excluded from further analyses. The correlation matrix of the remaining 12 variables is shown in Table 4. Mean spring water temperatures were also excluded from the analyses, since values could not be calculated for all sites.

The first two principal component axes accounted for almost 71% of the variability between sites. Eigenvectors are presented in Table 5. From these analyses, the variables absolute maximum, diel range, mean summer temperature and mean autumn temperature contributed most to the site weightings in Axis 1, and exhibited a negative gradient with site groups. For Axis 2, absolute and mean minimum water temperatures were highly positively correlated with site groupings, while maximum temperature range and Julian date of annual maximum and minimum were highly negatively correlated with site groupings.

A scatterplot of the principal components analysis is shown in Figure 6, which showed three clear site groupings. Used in conjunction with the cluster analysis (Figure 7), sites could generally be grouped into a warmer, downstream (lowveld) assemblage (S7, S6, S5, SS), a cooler upstream group (S1 and Salt River (Sa) sites), and an “intermediate” middle reach grouping.

Table 3: Results of multiple linear regression models based on correlations between maximum daily water temperatures, and flow, relative humidity, and mean, maximum and minimum daily air temperatures

Model	Sample size (n)	Technique	Terms	Model	R^2	Adj. R^2	Simulation accuracy (% mean±sd)
1	2650	Standard regression	AT_{avg}, AT_{min}, AT_{max}	$WT_{max} = 1.655 + 1.089 * AT_{avg} - 0.097 * AT_{min}$	0.81	0.81	-0.02±0.13
		Stepwise forward		$WT_{max} = 1.641 + 1.058 * AT_{avg} - 0.081 * AT_{min}$	0.81	0.81	
		Stepwise backward		$WT_{max} = 2.248 + 0.773 * AT_{avg} + 0.150 * AT_{min}$	0.84	0.84	
2	778	Standard regression	AT_{avg}, AT_{min}, AT_{max}, Flow	$WT_{max} = 2.170 + 1.052 * AT_{avg} + 0.116 * AT_{min}$	0.86	0.86	0.11±0.16
		Stepwise forward		$WT_{max} = 2.166 + 1.053 * AT_{avg} + 0.117 * AT_{min}$	0.86	0.86	
		Stepwise backward		$WT_{max} = 2.119 + 0.891 * AT_{avg} + 0.196 * AT_{min}$	0.86	0.86	
3	2452	Standard regression	AT_{avg}, AT_{min}, AT_{max}, RH	$WT_{max} = 2.220 + 1.012 * AT_{avg}$	0.81	0.81	-0.03±0.13
		Stepwise forward		$WT_{max} = 1.679 + 1.078 * AT_{avg} - 0.119 * AT_{min}$	0.81	0.81	
		Stepwise backward		$WT_{max} = 1.679 + 1.078 * AT_{avg} - 0.119 * AT_{min}$	0.81	0.81	
4	594	Standard regression	AT_{avg}, AT_{min}, AT_{max}, Flow, RH	$WT_{max} = 1.890 + 0.795 * AT_{avg} + 0.206 * AT_{min}$	0.86	0.86	-0.06±0.14
		Stepwise forward		$WT_{max} = 2.184 + 0.788 * AT_{avg} + 0.215 * AT_{min}$	0.86	0.86	
		Stepwise backward		$WT_{max} = 2.199 + 0.920 * AT_{avg} + 0.152 * AT_{min}$	0.86	0.86	

Sites were also characterised by their level of predictability, even though this variable contributed less (0.294) to the weightings in Axis 1 than those weightings already mentioned, because of the importance of predictability to biological patterns (Colwell, 1974; Vannote and Sweeney, 1980). The warmer assemblage exhibited high summer and autumn water temperatures, high temperature extremes (high absolute maximum and diel range) and lower predictability. Conversely, the cooler upstream group was characterized by more predictable, cooler and less extreme water temperatures. The intermediate group showed higher absolute and mean minima, and an associated lower maximum range. This was also manifested in earlier onset of cool winter minima (Lower Julian annual minimum), which was approximately in June for the uMgeni and Fish River sites, in August for the Salt River Sites, and July/August for the Sabie River sites. The cluster analysis showed the sites Sa1 (Salt River upstream site) and S4 (Sabie River) to be distinct from the remaining sites at the first level of separation (Figure 7). The larger group could be divided further into generally downstream (warmer) versus upstream (cooler) sites at a second level of separation. The warmer group could be further split based on lower versus lower reach sites, while the cooler group split into a pool site versus the remaining non-pool sites.

Table 4: Correlation coefficients for 12 water temperature statistics for 20 sites in four river systems in South Africa

	Co-efficient of vari-ability	Predict-ability	Absolute minimum	Absolute maximum	Mean minimum	Diel range	Maximum diel range	Mean summer temp.	Mean autumn temp.	Mean winter temp.	Julian date of annual maximum	Julian date of annual minimum
Coefficient of variability	1.00	-0.43	-0.30	0.23	0.11	0.04	0.06	0.33	0.04	-0.31	-0.09	-0.32
Predictability	-0.43	1.00	0.48	-0.66	-0.34	-0.68	-0.52	-0.59	-0.47	-0.47	0.02	-0.09
Absolute minimum	-0.30	0.48	1.00	-0.19	0.40	-0.40	-0.61	0.09	0.20	0.11	-0.10	-0.38
Absolute maximum	0.23	-0.66	-0.19	1.00	0.72	0.85	0.73	0.88	0.85	0.76	0.20	0.03
Mean minimum	0.11	-0.34	0.40	0.72	1.00	0.47	0.16	0.92	0.95	0.76	-0.17	-0.33
Diel range	0.04	-0.68	-0.40	0.85	0.47	1.00	0.76	0.69	0.69	0.71	0.14	0.24
Maximum diel range	0.06	-0.52	-0.61	0.73	0.16	0.76	1.00	0.40	0.38	0.47	0.43	0.46
Mean summer temp.	0.33	-0.59	0.09	0.88	0.92	0.69	0.40	1.00	0.95	0.73	-0.09	-0.25
Mean autumn temp.	0.04	-0.47	0.20	0.85	0.95	0.69	0.38	0.95	1.00	0.88	-0.11	-0.18
Mean winter temp.	-0.31	-0.47	0.11	0.76	0.76	0.71	0.47	0.73	0.88	1.00	-0.02	0.06
Julian date of annual maximum	-0.09	0.02	-0.10	0.20	-0.17	0.14	0.43	-0.09	-0.11	-0.02	1.00	0.62
Julian date of annual minimum	-0.32	-0.09	-0.38	0.03	-0.33	0.24	0.46	-0.25	-0.18	0.06	0.62	1.00

Table 5: Eigenvectors for principal components analysis of water temperatures (20 sites, 12 variables). Shaded cells represent variables which are most highly correlated with site clustering

	PC Axis 1	PC Axis 2
Cum. % of variance	47.82	70.81
Variable		
Coefficient of variability	-0.071	0.000
Predictability	0.294	0.167
Absolute minimum	0.056	0.472
Absolute maximum	-0.404	-0.062
Mean minimum	-0.334	0.328
Diel range	-0.363	-0.194
Maximum diel range	-0.273	-0.397
Mean summer temperature	-0.388	0.177
Mean autumn temperature	-0.385	0.205
Mean winter temperature	-0.354	0.086
Julian date of annual minimum	-0.021	-0.367
Julian date of annual maximum	-0.001	-0.478

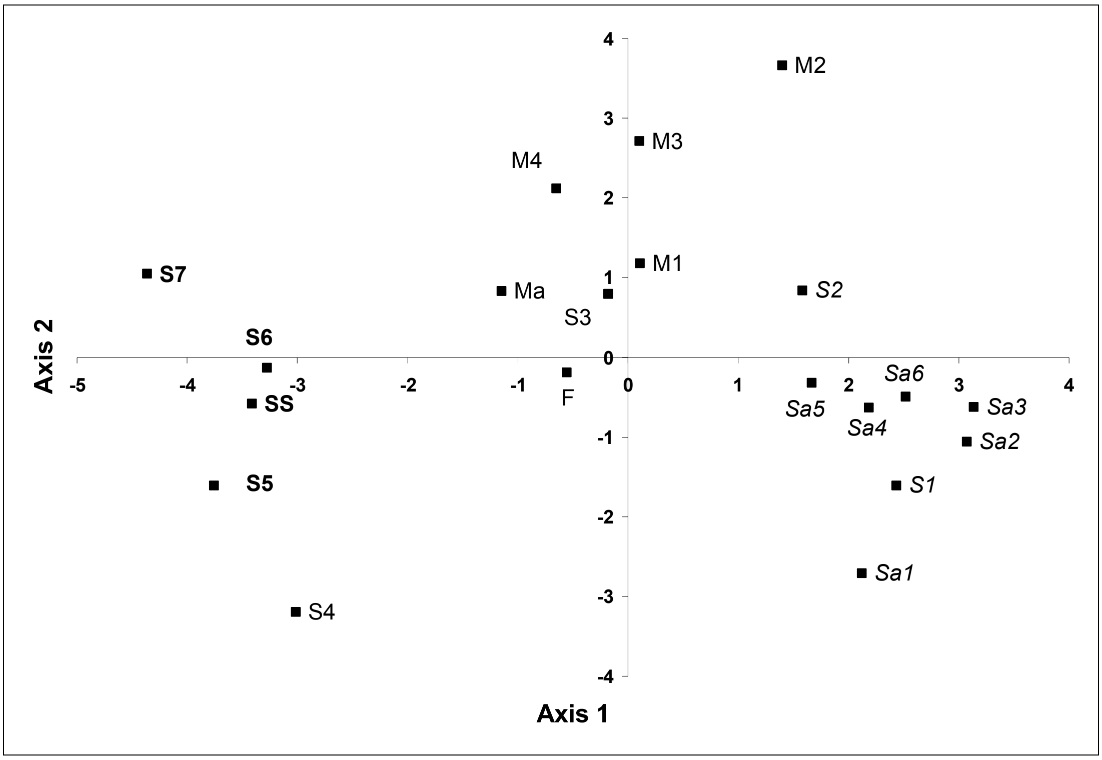

Figure 6: Principal components analysis of twenty sites where water temperatures were recorded in four river systems in South Africa, based on 12 water temperature statistics. Site labels are listed in upstream to downstream order by river as follows: **Salt River** = Sa1, Sa2, Sa3, Sa4; **Great Fish River** = F; **uMgeni River** = M1, M2, M3, M4; **Sabie River** = S1, S2, S3, S4, S5, S6, S7; **Marite River** (Sabie-Sand River system) = Ma; **Sand River** (Sabie-Sand River system) = SS. Upstream sites are indicated in *italics*, while downstream sites are indicated in **bold**.

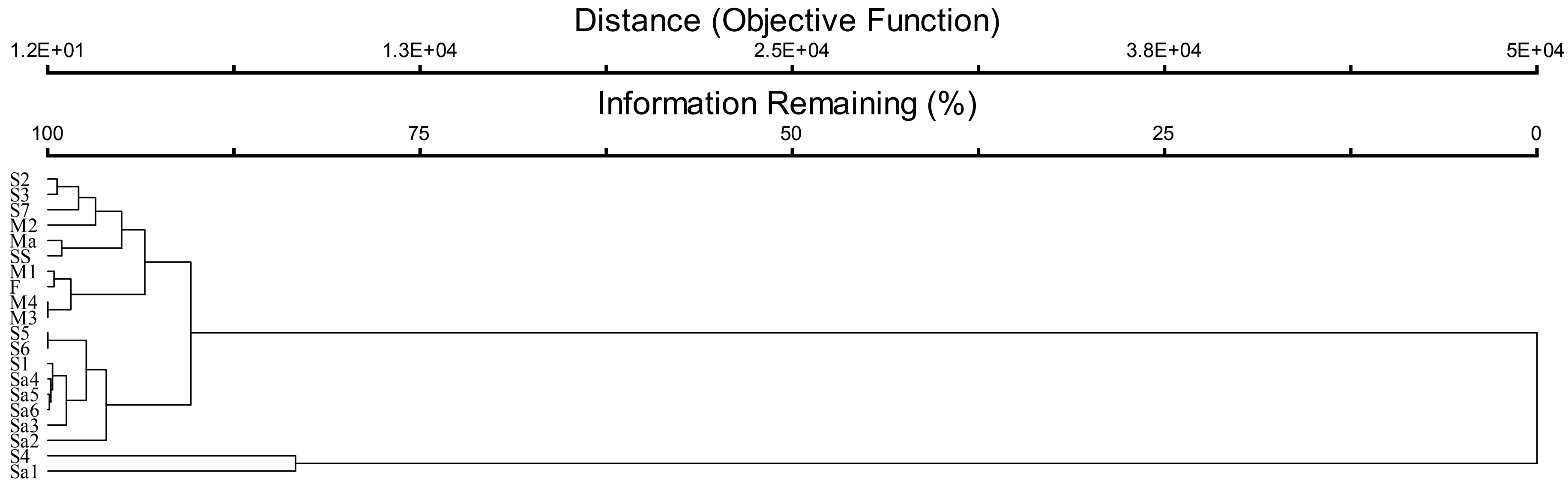

Figure 7: Cluster analysis of twenty sites where water temperatures were recorded in four river systems in South Africa, based on 12 water temperature statistics

5.3.2 Conceptual model development

Appraisal of existing approaches

A number of points pertinent to developing a water temperature model emerged from the principal components analysis. Site groupings again suggest that an ecoregion approach to predicting water temperatures would be appropriate. Data also suggest that temperature extremes (mean and absolute maxima and minima; daily range) are important in site-specific water temperature signatures, and that any water temperature model would need to take cognisance of this. Finally, the date of winter minima is more variable (June to August) than the onset of summer maxima (February), suggesting that winter temperatures are important in determining local site characteristics, and determining species community patterns and turnover in time and space (Vannote and Sweeney, 1980). Thus an ecologically suitable water temperature model should focus on not only daily maximum water temperatures, but also daily minimum water temperatures, which could provide insights into understanding the magnitude and direction of energy fluxes influencing water temperatures (Johnson, 2003).

Various modelling approaches were considered in this report. Correlations between air temperatures and water temperatures were shown to be the most significant predictor of water temperatures, and have been shown to be significant up to, on average, 70 km from the river (Steffan and Preud'homme, 1993). Multiple regression models suggest that the effects of relative humidity, rainfall and flow are small compared to air temperatures (current analyses, but see also Webb and Nobilis, 1997; Rivers-Moore et al., 2005). It would appear that no single generic statistical model is possible, since relationships typically vary between catchments (Webb and Nobilis, 1997; Rivers-Moore et al., 2004). Such a complex relationship was also demonstrated for the Sabie River (Rivers-Moore et al., 2004; Rivers-Moore et al*.,* 2005), where each of nine site's water temperatures could best be simulated by a unique multiple regression model, and that site-specific simulation accuracy was reduced when a single generic model was used. The data in this study also suggest that each river site tends to be unique, such that were a statistical modelling approach adopted, a unique model would need to be developed per site, which is logistically not feasible. The highest correlations were achieved using multiple linear regression models which **did not** incorporate a flow-dependant term. However, the incorporation of a flow term, while perhaps reducing model accuracy, greatly enhances the utility value of such a model (Rivers-Moore et al., 2005), because of the increased model utility to aquatic ecologists and engineers in predicting the consequences of different flow modification scenarios on water temperatures. The results of this study showed, in addition to the overriding influence of air temperatures as a surrogate for solar radiation, that model accuracy increased with increasing sample size. Ongoing development of a water temperature model should thus proceed in conjunction with ongoing collection of sub-daily water temperature data.

Risley et al. (2003) used an artificial neural network (ANN) model, based on a sigmoidal /harmonic function (*tan*). This approach is common where processes are difficult to simulate using mechanistic modelling. The model "learns" to predict water temperatures more accurately by adjusting model weights using a nonlinear multivariate optimization algorithm to reduce the root mean square error. However, this approach relies on site-specific input time series (observed data) of water temperature to train the model, and also requires multiple model sets, depending on how many clusters of sites are present. This suggests again that the ecoregional approach is a correct one, also borne out by the PCA analysis, but that an ANN modelling approach would not be appropriate where data are scarce, and where a generic approach is desired.

A promising alternative would be a process-based approach, provided this could be kept relatively simple, since process-based models are typically expensive to develop (Webb, 1996), and data-intensive (Jeppeson and Iversen, 1987; Stefan and Preud'homme, 1993; Smith, 1972), while simple pragmatic models are better suited to ecological studies. Johnson (2003) report that site-specific river temperatures are controlled by their immediate surroundings, as well as by upstream conditions. This makes a good argument for using a process-based dynamic water temperature model which combines incoming upstream water temperatures with temperatures at the site. In this approach, the basic principle is to simulate heat gains and losses, where the rate of change over a particular river length ($\delta T°/\delta x$) is calculated as the difference between gains and losses, and downstream water temperatures are the sum of input water temperatures and heat gains. This approach provides a more generic solution, where a single model could have multiple site applications.

One approach which shows promise, and which could be adapted to South African rivers, is the Stream Network Temperature Model (SNTEMP), and a simplified version, the Stream Segment Temperature Model (SSTEMP) (Bartholow, 2002). These models were developed with the aim of providing a tool to aquatic biologists and engineers to predict the consequences of stream manipulation on water temperatures. Both models are based on a mechanistic one-dimensional heat transport model, which is able to predict daily mean, minimum and maximum water temperatures. In this modelling system, the river is divided into homogenous segments, and water temperatures are simulated based on heat gained or lost from a parcel of water as it passes through a stream segment, based on inputs which include radiation, surface and groundwater flows (Bartholow, 1989). One drawback of this system is that even the simpler model, SSTEMP, requires at least 22 input variables, based on hydrology, river geometry, shading and meteorology (including solar radiation inputs). An example of the model interface for SSTEMP is presented in Figure 8.

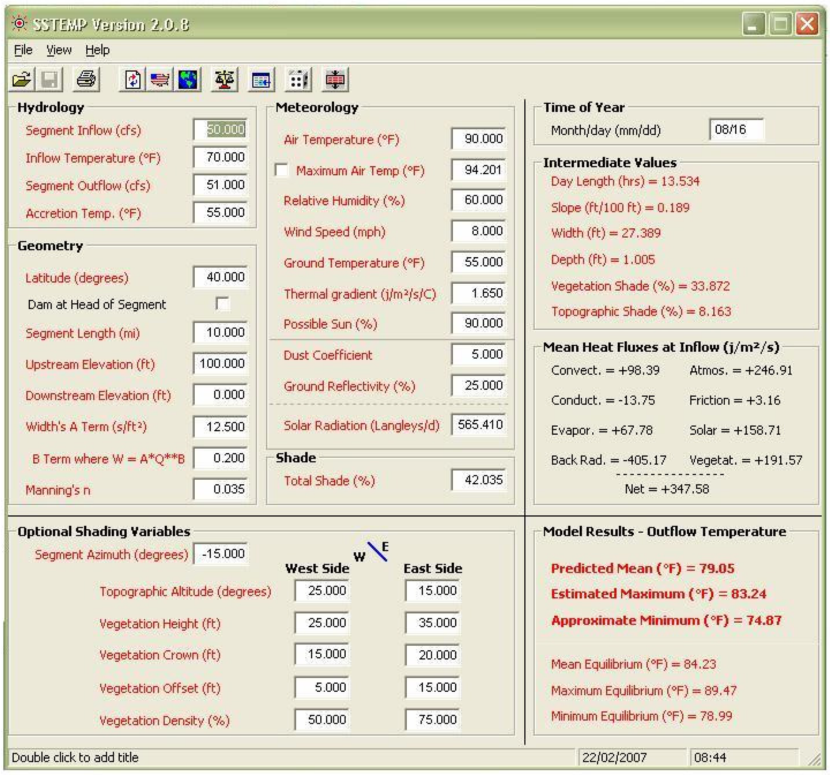

Figure 8: The SSTEMP interface, showing input variables required to calculate mean, maximum and minimum daily water temperatures

Walters et al. (2000) simulated mean monthly water temperature for the Colorado River, as part of a larger modelling exercise to understand the effects of flow regulation on this river system. Water temperature change along the downstream axis of the Colorado River was simulated by calculating gains (drivers) and losses (evaporation) in heat to a "parcel" of water (Lagrange modelling) as it moves downstream, with the river divided into different reaches. According to Walters et al. (2002) simulating gains and losses of materials (such as temperature) is one of the most difficult computational problems, and the Lagrange approach uses non-trivial mathematics. In this model, water entering a reach has a temperature as a function of upstream water. Downstream water temperatures (C_{i+L}) were simulated by multiplying the upstream water temperature entering the reach (C_i) by a coefficient that is a function of the residency time of the water within a reach (which in turn is a function of reach length and water velocity), and a heat exchange coefficient (Equation 2). Thus, downstream temperatures are a function of water temperatures entering the reach, which are modified by reach characteristics. Water temperatures entering the reach are a flow-weighted average of the sum total of water entering the reach (mainstems and tributaries). These are in turn modified by the equilibrium temperature (which was simply a lookup table in Walters et al., 2000 with monthly values to incorporate seasonality). The equilibrium temperature was defined as "the equilibrium concentration or temperature that would occur if the sample water parcel moved forever over an infinitely long reach at fixed gain and loss rates and no change in cross-sectional area" (Walters et al., 2000). The extent of the departure of the equilibrium concentration or temperature from the upstream boundary condition (C_i-C_{eq}) is modified by an exponentially decreasing function (0 to 1 scale), which incorporates reach characteristics, while a heat exchange coefficient allows tuning of the model (Walters et al., 2000). One possible approach to simulating the extent of departure from equilibrium temperature is to use (multiple) linear regression models which apply to different ecoregions to estimate regional departures from equilibrium temperatures. These could then be applied to the proposed dynamic water temperature model using lookup tables and indexed using ecoregions.

$$C_{i+L} = C_{eq} + (C_i - C_{eq})\exp(-vW_iL_i / Q_i) \qquad [2]$$

where C_{i+L} is downstream temperature; C_{eq} is equilibrium temperature; C_i is the flow-weighted input temperature from upstream; and *exp( )* is a modifier term with *v* being a heat exchange coefficient; W_i is wetted width; L_i is reach length and Q_i is mean monthly water discharge.

Dynamism in the model is added by assuming a linear change in water temperatures within a reach. Thus, "dynamic change along each Lagrange sample track is assumed to follow relatively simple linear dynamics (*dx/dt* = *a -bx,* where *a* is the input rate, *b* is the loss rate, and *x* is reset at each at the time when the sample parcel passes each tributary input point) for which we can obtain an analytical solution for downstream [temperature] changes" (Walters et al., 2000).

The complexities of applying dynamism to water temperature modelling were, however, illustrated by Rivers-Moore (2003), based on an application of the model proposed by Rivers-Moore and Lorentz (2004).

Conceptual model framework

Results from Section 5.3 showed that distinct thermal differences existed between upper reaches versus lower reaches, which was also borne out by the principal components analysis. Most notably, cumulative monthly water temperatures in the upper reaches of the rivers considered (Salt River, upper Sabie River) showed less seasonality (flatter curves) than cumulative monthly water temperatures in lower river reaches (Fish River, lower Sabie River) (Figure 9). It was assumed that greater groundwater inputs in the upper reaches resulted in more stable thermal regimes with less seasonal effects. Conversely, water temperatures in the lower reaches are relatively more influenced by thermal radiation, and show more marked seasonality than water temperatures in upper river reaches.

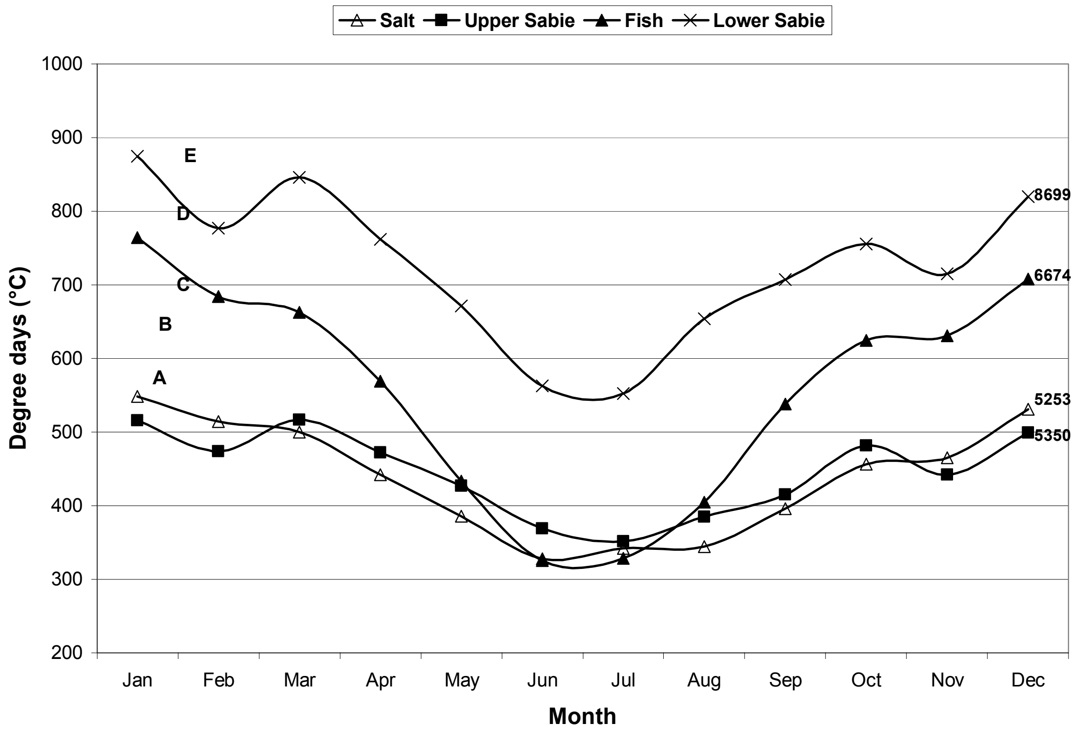

Figure 9: Cumulative monthly degree days for two upstream sites (Salt, Upper Sabie) and two downstream sites (Fish, Lower Sabie)

Such results illustrated that the following components were important to a water temperature model:

- Solar radiation (air temperatures as surrogate)
- Spatially dynamic model flexible enough to incorporate upstream-downstream effects (see Rivers-Moore and Lorentz, 2004)
- Evaluation of temperature signals attached to different flow events; disaggregation of (monthly) flows into groundwater *vs.* instream (surface and interflow) flows.

The "ideal" water temperature model should consider the following:

- Use data which are readily and widely available (air temperatures)
- Integrate with other models (and use outputs from these e.g. Pitman Model)
- Be useful in scenario analyses (e.g. incorporate flow and turbidity terms, such as for dam releases; climate change)
- Scale should be suitable for ecological Reserve application, and general ecological use; a daily time-step model would be the most useful for these purposes
- Build on existing research on water temperature models in South Africa
- Generic i.e. applicable at a range of spatial scales and applicable throughout South Africa
- Simple, with as few terms as possible
- Dynamic (i.e. allow for simulating change in water temperature with change in downstream distance).

Outputs from a suitable model, such as the Pitman Model, which generates three flow parameters (groundwater, surface water and interflow) based on rainfall inputs, would provide input data into a water temperature model. A generalized water temperature model should be of the form of Equation 3, in which flow-weighted heat gains and losses are modified through an exponential modifier term, similar to that used by Walters et al. (2002).

$$\frac{\delta T^{\circ}}{\delta x} = f\left(AT, flow1, flow2, flow3\right)\exp\left(c, Q, Tb, Sd\right) \qquad [3]$$

where change in water temperature (δT°) with change in downstream distance (δx) is a function of air temperature (*AT*); groundwater, surface water and interflow components of a hydrograph respectively (*flow1*, *flow2* and *flow3*); *c* is a heat exchange coefficient dependant on ecoregion geomorphology - the pool/riffle ratio (profile classification) and flow depth (residency time of water – hydraulics of width:depth ratio, as derived from a hydrological model e.g. Pitman model), and modified by flow volume (*Q*), turbidity (*Tb*) and riparian shading (*Sd*).

The basic concepts of the proposed approach are derived from a similar experience of interfacing the monthly Pitman model (with revised ground water routines – Hughes 2004), as well as the daily variable time interval (VTI) model (Hughes and Sami, 1994) with some simple water quality mass balance equations to simulate time series of total dissolved solids. The approach has been developed for ephemeral rivers where evaporation from pool storage is thought to play a major role in both the water and chemical mass balances. Both models operate in a semi-distributed mode, where each sub-catchment within the whole basin has separate input and parameter data and the model simulates the links between the sub-catchments.

To keep things simple during the development of the approach, the hydrological models are run and then the component outputs (runoff and storages) are used as input to the water quality model. Initial results of testing the model on the Seekoei River under the WRC Ephemeral Rivers Hydrology project (K8/679) are relatively encouraging.
The basic principles of the water quality model are presented in Tables 6a-b

Table 6a: Water quality model principles

Water quantity com - ponent output by the hydrological model	**Water quality parameters required**	**Quality modeling process**
Surface runoff	TDS signature of the three components (conc.)	Simple volume * concentration calculation generating load input to the pool storage.
Interflow		
Ground water discharge		
Pool storage	Pool dimensions/geometry Bank exchange parameter	Pool mass balance calculations including increase in concentration due to evaporation and simple TDS exchange with pool bed and banks during wetting and drying.
Downstream outflow	None	Pool overflow generates load inputs into next downstream sub-catchment.

Table 6b: Equivalent water temperature proposed principles

Water quantity component output by the hydrological model	Water temperature parameters required	Quality modeling process
Surface runoff Interflow Ground water discharge	Temp signature (seasonally varying) of the three components. The surface and interflow runoff signatures may be linked to an input time series of air temperature.	Volume/temperature mixing calculation for the sub-catchment as a whole.
Pool storage	Channel dimensions & geometry (e.g. volume v surface area relationship). Parameters used to define the heat exchange between the channel water and the atmosphere within the channel reach of the sub-catchment. May be related to inter alia solar radiation, air temperature, water turbidity, riparian veg. shading, etc.	Channel temperature dynamics. The form of the model algorithms will depend on the available input data but will be essentially a temperature mixing model that may account for diurnal effects. The model will have to account for the length of the channel reach and the increases or decreases in temp. that occur within the reach.
Downstream outflow	None	Output temperature and water volume will form an input to the temperature mixing model at the upstream end of the next sub-catchment.

6. SUMMARY AND CONCLUSIONS

What is known about water temperatures in South African rivers is considerably less than what is unknown. In this situation of uncertainty, where data and knowledge are scarce, the precautionary principle should be applied (Ashton, 2005; Groves, 2003), in which the emphasis changes from the burden of proof from showing how a change would be harmful, to that it would not be harmful.

Three methods of "measuring" river health are to establish baseline conditions, measure departure from the baseline, and implement management action through recognising when thresholds have been exceeded (Ladson et al., 2006). Within this hierarchy, the definition of baseline condition is critical, since there are a wide range of these depending on the starting point – historical (pre-human; pre-Colonial), least disturbed, best attainable – which all reflect differing degrees of biological integrity (Stoddard et al., 2006). Metrics using aquatic macroinvertebrates can be useful in determining these levels. However, it is becoming increasingly clear in the literature that the way such data are collected and used determines the value of any stream measurements. Specifically, there is criticism for the use of the "representative reach", which is a subjectively chosen stretch of river which does not afford the collection of means and standard deviations, and the lack of being able to assign confidence limits to such data (Ladson et al., 2006; Smith et al., 2005). There is alternatively support for randomly selected sites, with the number of sites increasing depending on which measure of river health is required (Ladson et al., 2006), and the development of techniques to assign confidence thresholds to these metrics (Smith et al., 2005). However, random sampling may not be appropriate when particular ecological problems require answers: the examples on intermediate-scale sampling have also illustrated this point. These emerging approaches and the associated mixed debate is encouraging given that there is an established recognition of the importance of variability within river systems, as well as the need to establish "natural" ranges of variability. Given that South African river systems have been shown to be highly variable and statistically to be described by extremes, these emerging approaches are particularly pertinent to any application and development of water temperatures to the ecological Reserve.

In the absence of time series, scenario analyses assume greater importance, and this requires good predictive models. Even simple models, such as the expert system for assessing the conservation status of rivers developed by O'Keeffe et al. (1987) are powerful tools in evaluating scenarios.

Water temperatures are a climate-dependant variable - anticipated global warming scenarios may change the shape of ecoregions in the future. Subtle ecosystem relationships may unravel through impacts of altered temperatures on the timing of insect lifecycle stages (Saxon, 2003). The importance of temperature research will be to identify critical temperature thresholds (degree days) and relate this to ecological functioning, which at this stage may be best attained using non-parametric statistics (percentiles) within a "range of variability" approach, and associated confidence levels attached to these.

Confirming the findings in this study, air temperature has been shown to be the driver variable which water temperatures are most sensitive to (Bartholow, 1989). However, the effects of air temperatures on water temperatures are buffered by a combination of flow and residency times (width:depth ratios) (Bartholow, 1989). Preliminary data suggest that turbidity may be a factor which should be incorporated into a water temperature model to simulate water temperatures more accurately, since turbidity is known to modify heating rates of water. One option considered was to simulate daily water temperatures for 1997-1999 using maximum and minimum air temperatures obtained from Schulze and Maharaj (2003). However, this would not have reflected effects of turbidity on water temperatures. Such a relationship could be determined in the future by comparing differences in residuals between simulated and observed water temperatures at sites with different turbidity loads.

Additionally, a suitable water temperature model should also be flexible enough to incorporate the relative buffering effects of different components of a hydrograph (groundwater and surface water) on water temperatures. Preliminary analyses also suggest the importance of both daily minima and maxima in determining site-specific water temperature signatures, so that an ecologically useful water temperature model should ideally be able to simulate daily mean, minimum and maximum water temperatures.

It is recommended that for specific river applications, the river be divided into thermal reaches. **In-stream, in-reach water temperatures** will be simulated based on drivers and modified by buffers: driven by air temperatures, and buffered by the effects of turbidity (deep muddy *vs*. shallow muddy; dam releases), riparian shading, residency time (pool:riffle ratios, as derived from lookup tables based expert opinion or site surveys and river profiles), and hydraulics (width:depth ratios). Increased turbidity loads in summer typically modify rates of water heating, while low sun zenith angles in winter lead to altered reflectance (Rivers-Moore et al*.,* 2004). Each of these time series will be used as inputs to simulate downstream water temperatures (i.e. **between reach water temperature simulations**). This will be achieved using a differential equation, which also includes the buffering effect of groundwater on water temperatures. Specific to the generic water temperature model, we recommend the following approach:

- Progression from a conceptual to a working generic water temperature model for South Africa, through development of a simple, process-based water temperature model proposed in this report. We recommend that model development and testing take place in association with calibration using site surveys;
- Comparison of water temperature simulations using the above-mentioned model and simulations using SSTEMP;
- Further investigation into the relationship between turbidity and water temperatures;
- Further investigation, based on empirical data, into the relative sensitivity of water temperatures to groundwater versus surface water inputs, and the relative contributions of groundwater and surface water to water temperatures along river longitudinal axes.

- Testing of the use of simplified geological maps to derive surrogates for hyporheic influences, with model input parameters chosen using a decision flowchart. This could be developed in conjunction with broad river profile types, where different regions of the profile are associated with different hydraulic (and geomorphological) characteristics. Certainly for the Sabie River, Rivers-Moore and Jewitt (2004) have shown a distinct downstream gradient in water temperatures, which could be related to river profiles.

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