# Water Temperatures and the Ecological Reserve

Report to the
**Water Research Commission**

by

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# EXECUTIVE SUMMARY

## Introduction and Aims

Freshwater systems, both globally and within South Africa, are under pressure, and are amongst the most deteriorated and worst off systems, due in part to water abstraction, flow regulation and pollution. Successful implementation of environmental flow management requires taking cognizance of the full spectrum of flows together with thermal regimes, including their temporal and spatial variability. Water temperature is recognized as an important abiotic driver of aquatic ecosystems, and understanding the role that temperature plays in driving ecosystem change is important if effective management of thermal stress on aquatic ecosystems is to be achieved. Only through a foundation of fundamental research linking water temperatures and biotic response will the water temperature requirements for the ecological Reserve be met.

The main aims of this research were to:

- Collect baseline water temperature data in a range of rivers in the Western and Eastern Cape, South Africa
- Develop a generic water temperature model for South Africa
- To develop an understanding of the response of aquatic organisms to water temperature regimes in South African rivers
- Identify a suite of suitable aquatic macroinvertebrates for use as bio-indicators of thermal change
- Develop preliminary guidelines for the water temperature component of the ecological Reserve
- Develop scenarios of the potential biotic responses to changes in water temperature regimes as a result of climate and hydrological changes.

## Baseline water temperature data collection

A total of 92 sites in rivers in the Eastern and Western Cape provinces were selected for installation of water temperature loggers. Sites were selected to cover a range of ecoregions and river longitudinal zones (geomorphological classes). An additional 50 sites from collaborative projects were also used. Hourly water temperature data were collected over a period of two years using Hobo UTB1-001 TidBit V2 loggers. Air temperature and relative humidity data were collected at 47 sites to facilitate linking water temperatures to air temperatures for water temperature modelling purposes.

## Synthesis of main findings

This research project ran for in excess of three years, with a total of 31 deliverables and 14 final reports on various aspects of water temperatures and river ecology and management resulting from this process. These individual reports are included as PDF files in the CD at the back of this report. Each of these reports have been synthesized into a main report, which has been divided into three main themes, viz. modelling and mapping of water temperatures, biotic responses to thermal change and stress, and applications for management and inclusion into South Africa's ecological Reserve planning. Through this process, the collection of hourly water temperatures at a spread of seemingly unrelated sites have been integrated into a broader context to illustrate how water temperature time series can be linked to biotic responses, and integrated into a spatial framework.

## Modelling and Mapping

Measured water temperatures are the end result of complex interactions between solar radiation (heat inputs), flow rates, groundwater inputs and mixing with surface flows, shading, and land use (affecting turbidity levels). Increased turbidity can affect maximum daily water temperatures by up to 1°C, while groundwater inputs reduce daily ranges of temperatures and seasonal variability. Groundwater temperatures can be approximated by mean annual air temperatures, although their absolute impact on surface water temperatures, based on recharge rates and flow mixing, requires further research, as does the effect of land use and sediment load on turbidity levels and water temperatures.

Because water temperature time series across South Africa are limited in distribution and length of data (i.e. > two years), it is more typically necessary to simulate water temperature time series from data which are readily available, viz. daily air temperature data. The statistical relationship between air and water temperatures is well established, and is usually linear over the range of water temperatures currently measured from South African rivers. Water temperature time series can be simulated using linear models, either as stand-alone exercises, or to generate inputs for established process-based water temperature models. The process-based models are more complex to populate, but have the capability of running more sophisticated water temperature scenario analyses. For ecological Reserve assessments, modelling outputs should include daily mean, minimum and maximum temperatures. Good quality input data (air and water temperatures) are key to good temperature simulations and ideally each site should be modelled individually after a site visit.

A water temperature time series can be broken down into metrics that describe it in terms of thermal magnitudes, and frequency, timing and duration of extreme events. Using this approach, variation in water temperatures could be quantified, and was shown to vary between hydraulic biotopes, and regionally. In the former case, flow differences translated more markedly in affecting daily maximum temperatures during the hotter time of year, with shallower biotopes (riffles) being more vulnerable to thermal extremes. Regionally, water temperature sites could be classified according to the metrics, which statistically describe their thermal regimes. By linking thermal metrics to catchment conditions already mapped (altitude, mean annual air temperature), thermal metrics and thermal groups were mapped. There was poor correlation between thermal groups/metrics and existing ecoregions. In spite of an average prediction success of *ca.* 40% for thermal groups, this research has provided a foundation for including water temperatures in a regional approach to managing river systems. Further research is required to refine regional water temperature mapping.

### Biotic Responses

The importance of water temperatures to aquatic biota has been well documented from Northern hemisphere research. Southern hemisphere studies, including South Africa, are relatively scarce, with this project contributing substantially to the growing body of knowledge on biotic responses to water temperature. Studies from this research showed that water temperature regimes have a measurable impact on aquatic macroinvertebrate life histories, and life cycle cues. Through a combination of field surveys and laboratory experiments, it was shown that life histories of three target macroinvertebrate species showed differing degrees of flexibility in life history responses – from subtle changes in the timing of emergence and egg hatching to more extreme differences involving the production of additional generations within a year given differing environmental conditions. These responses were primarily related to water temperature and flow. Life-history studies inform all areas of aquatic ecological research, whilst also providing information relevant for conservation, and management of river systems. Life history data are useful in that they can be linked to *in situ* thermal data and be used in the development of thermal guidelines and scenario analysis.

At a community level, and in agreement with existing international studies, the level of thermal variability in a river system affects aquatic macroinvertebrate community structure. More variable (less predictable) river systems are more likely to have greater numbers of generalist species, and higher levels of taxonomic turnover and diversity than more predictable rivers.

Thermal stress is likely to increase in the south-western Cape as a result of predicted climate change, viz. reduced rainfall and elevated air temperatures. Sampling macroinvertebrate assemblages, with the aim of assessing if biota differed in rivers with different thermal signatures, did not however, reveal any distinct faunal trends that could be related to thermal conditions, even though water temperature data showed distinct differences in thermal conditions across the range of sites. Likely reasons for this are the relatively course level at which the sampling took place, mesh size and the likelihood of catchment signatures playing a role.

A number of laboratory experiments of short (Critical Thermal Maxima, CTM) and long (96h-$LT_{50}$) duration led to the identification of a suite of macroinvertebrate taxa that were thermally sensitive. These taxa have potential for use as bio-indicators of thermal change. Thermal stress is taxon-dependant, and specialist taxa were typically more sensitive to thermal stress than more generalist species. The relationship between CTM and 96-$LT_{50}$ experiments has been established, which facilitates future experimental work based on CTM. Laboratory data on thermal tolerances such as incipient lethal limits may be used to generate biological temperature thresholds. These may then be used in, for example, the thermal component of the ecological Reserve and the determination of exceedance of these biological temperature thresholds.

## Management

Understanding the predictability or cyclical constancy of water temperatures, and how this changes with downstream distance, is an important predictive tool in relating biotic responses to abiotic change. Temperature regimes can be summarized and quantified using statistics that describe distribution. As mentioned earlier, a total of 37 metrics to describe water temperature time series were developed. The temperature metrics define statistics of a river’s thermal regime with respect to magnitude of water temperatures, frequency, timing and duration of thermal events. The metrics were shown to be sensitive enough to distinguish adjacent sites based on their thermal signatures, but also robust enough to remain relatively consistent (assuming no trends) inter-annually. Metrics provide a structured approach to disaggregating and describing water temperature time series over annual periods, and to compare inter-annual differences in thermal data at a single site. Such metrics are first and foremost a tool which has a number of applications. For example, the usefulness of metrics in classifying sites based on thermal characteristics has applications at a number of scales, from making distinctions between hydraulic biotopes to defining thermal regions for management actions. Linking metrics to temperature thresholds for selected macroinvertebrates provide the capacity to define components of the thermal ecological Reserve, and to assess flow reduction and climate change impacts on biota.

Determination of environmental flows and the ecological Reserve are incomplete without incorporating water temperatures into their assessments. Water temperatures have been shown to be as important as flows in determining aquatic biotic patterns. Impacts on river systems which particularly affect water temperatures include impoundments and flow abstractions. These impacts are felt as thermal stress by aquatic macroinvertebrates. Stress can be the result of water either being too cold or too hot. Departures from reference (natural) thermal conditions can be assessed using the thermal metrics, frequency and duration of exceedance of biological temperature thresholds, or whether a thermal regime falls within a daily range buffer or acceptable band. Such bands should be defined for reference sites per thermal region (as described in the mapping section). Successful application of this approach requires refinement of thermal regions, definition of average reference site conditions per region linked to these spatial units. Also required is further research linking biotic response to thermal changes, and a better understanding of links between flows and temperatures following disruptions to the river continuum (flow abstraction and impoundments).

In this research, response of aquatic macroinvertebrates to thermal conditions was successfully achieved using thermal stress thresholds and cumulative measures of heating. Cold-adapted specialist aquatic macroinvertebrates which breed only once a year (which are usually of high conservation importance) are most vulnerable to small increases in water temperatures. Pest species of aquatic macroinvertebrates, which typically breed throughout the year and which are widespread, are likely to benefit from increased water temperatures. Outbreaks of pest species are likely to become more sever. Under conditions of increased water temperatures, aquatic macroinvertebrate communities could become increasingly dominated by warm water, widespread generalist species.

## Conclusions and key messages

The body of research in this project represents a considerable advancement in understanding thermal patterns in South African rivers, and how biota (individual species and aquatic macroinvertebrate communities) respond to thermal variability and stress. Understanding spatio-temporal thermal patterns in the Eastern and Western Cape provinces requires a multi-scale approach. Linking biotic response to thermal drivers is naive if mean temperature values only are used. Spot measurements of water temperatures are at best inadequate when used in conjunction with other data. Rather, understanding biotic responses to thermal regimes not only involves fundamental research on life histories of taxa, but also in relating these to the subtler statistics of a thermal regime. The collection and/or modelling of sub-daily temperatures (mean, minimum and maximum values) is fundamental to describing thermal regimes relative to timing, frequency, duration and magnitude of thermal events.

Any ecological Reserve determinations in South Africa would be incomplete unless thermographs are considered together with hydrological assessments. Our research has gone a considerable way to providing the tools for accomplishing this. In summary, we have:

- provided the models for simulating water temperatures in the absence of water temperature data;
- automated the calculation of temperature metrics that facilitate the conversion of sub-daily temperature data into statistics that define a river's thermal regime with respect to magnitude of water temperatures, frequency, timing and duration of thermal events;
- identified thermally sensitive macroinvertebrate taxa that may be used as bio-indicators of thermal alteration;
- identified key life history cues for selected macroinvertebrates in the context of water temperatures;
- demonstrated the importance of maintaining thermal variability in river systems for aquatic macroinvertebrate community structure;
- generated a preliminary map of thermal regions that can provide an initial framework within which the thermal ecological Reserve is applied;
- developed a thermograph that incorporates the natural range of variability and the concept of reference sites (and condition) with which an assessed (impacted) site can be compared, and the effect (if any) quantified; and

- provided a decision tree for determining thermal ecological Reserve exceedance.

Future areas of research should focus on the following:

- Expansion of the geographic range of this study to a national scale.
- Establishment of a water temperature monitoring network to collect long-term water temperature data. This facilitates detecting warming or cooling trends, as well as departures from current conditions.
- Development of management tools from this research for use by practitioners and managers in ecological Reserve determinations. Links between environmental flows, thermal regimes and biotic responses need to be made more explicit.
- Continuation of life history studies to expand our knowledge of the extent to which species are cued into water temperature and flow.
- Continuation of thermal tolerance studies to determine the extent to which bio-indicators are regionally applicable and the potential to utilise family level data.

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| | |
|---|---|
| 7-D | Seven-day |
| ACRU | Agricultural Catchments Research Unit Agro-hydrological model |
| ALT | Altitude |
| AT | Air temperature |
| CART | Classification and Regression Tree analysis |
| CCA | Canonical Correspondence Analysis |
| CTE | Critical thermal endpoint |
| CTM | Critical thermal method |
| CTmax | Critical thermal maximum |
| CTmin | Critical thermal minimum |
| CV | Coefficient of variation |
| DWA(F) | Department of Water Affairs (and Forestry) |
| IHA | Indicators of Hydrologic Alteration |
| ILT | Incipient lethal temperature |
| ILUT | Incipient lethal upper temperature |
| $LT_{50}$ | Lethal temperature where 50% mortality achieved |
| MWAT | Maximum weekly allowable temperature (chronic threshold) |
| NMS | Non-metric multidimensional scaling |
| OT | Optimal temperature |
| PCA | Principal Components Analysis |
| RH | Relative Humidity |
| RI | Return/recurrence Interval |
| RMSE | Root mean square error |
| SD | Standard deviation |
| SSTEMP | Stream Segment Temperature model |
| TSR | Thermal sensitivity rank |
| WSTAT | W-Statistic |
| WT | Water temperature |

# 1 INTRODUCTION, AIMS AND OJECTIVES

Freshwater systems, both globally and within South Africa, are under pressure. According to the Millennium Ecosystem Assessment (2005), they are amongst the most deteriorated and worst off systems, due in part to water abstraction, flow regulation and pollution. Continual pressure from ever increasing population and the demand for freshwater, coupled with additional pressure from global climate change, make it critical to develop tools to reduce the chances of mismanagement of these systems. Successful implementation of environmental flow management requires taking cognisance of the full spectrum of flows together with thermal regimes, including their temporal and spatial variability (King et al. 2003), such that variability in discharge and temperature are considered simultaneously (Jackson et al. 2007; Olden and Naiman 2010). Of particular importance is the relationship between, and significance of, extreme low flows and elevated water temperatures.

Water temperature is recognised as an important abiotic driver of aquatic ecosystems (Smith, 1972; Ward, 1985; Caissie, 2006; Dallas, 2008; Webb. et al., 2008). Understanding the role that temperature plays in driving ecosystem change is important if effective management of thermal stress on aquatic ecosystems is to be achieved. Lotic ecosystems are subject to flow regulation and water abstraction to a much greater degree than lentic systems, and there are clear links between quantity and quality of water, with water temperature typically classified under "quality". Management approaches would be short-sighted not to take cognisance of the fact that flow and thermal regimes vary geographically in response to climate and catchment characteristics (geology, stream order, topography, land cover) (Poff and Zimmerman 2010). Consequently, a "one-size-fits-all" approach will apply to some systems some of the time, and be wrong most of the time: what is needed is a patchwork of management approaches which are tailor-made to the thermal regions.

It was recognised that only through a foundation of fundamental research linking water temperatures and biotic response will the water temperature requirements for the ecological Reserve be met. Previous research indicated that there were likely to be significant differences between northern hemisphere and southern hemisphere aquatic thermal regimes (Dallas 2008, 2009, Rivers-Moore et al. 2008c). To better understand these differences, a series of carefully constructed *in situ* and *ex situ* projects aimed at linking biotic response to thermal triggers, were undertaken. The usefulness of these data were further enhanced through a more complete spatial understanding of water temperatures, and a series of scenario analyses based on temperature simulations using a suitable water temperature model.

This report is a summary document that consolidates research undertaken as part of this project (K5/1799). It is divided into three main sections addressing 1) modelling and mapping, 2) biotic responses and 3) management. The relationship between components and links to management is indicated in Figure 1.1.

The main aims of this research were to:

- Collect baseline water temperature data in a range of rivers in the Western and Eastern Cape, South Africa
- Develop a generic water temperature model for South Africa
- Develop an understanding of the response of aquatic organisms to water temperature regimes in South African rivers
- Identify a suite of suitable aquatic macroinvertebrates for use as bio-indicators of thermal change
- Develop preliminary guidelines for the water temperature component of the ecological Reserve
- Develop scenarios of the potential biotic responses to changes in water temperature regimes as a result of climate and hydrological changes.

# 2 METHODS

## 2.1 Site selection

Preliminary sites for collection of baseline and project-specific data were identified using several spatial coverages, including DWA primary catchments, Ecoregions Level I (Kleynhans et al., 2005), longitudinal zones (Geomorphological classes, Moolman et al, 2006; 2008), Hydrological Index per quaternary catchment (Hughes and Hannart, 2003), River Health Programme monitoring Sites (Rivers Database, DWAF 2007), RHP national monitoring sites (Dallas, 2005a, b), flow types (Joubert and Hurley, 1994; King and Tharme, 1993), climatic zones of South Africa, 1:500 000 DWA rivers coverage, 1:500 000 DWA dams coverage and DWA flow gauging stations. Sites were selected at desktop level to represent a range of ecoregions and longitudinal zones, within each primary catchment and province (Western and Eastern Cape). Different flow types were incorporated, which provided the basis for identifying rivers with potentially different thermal properties. This process was followed to allow for future testing of the correlation between existing spatial coverages (e.g. Ecoregions) and aquatic thermal regions. Final site selection was undertaken in the field during logger installation, as all sites needed to be ground-truthed before deemed suitable for logger installation. Aspects considered included accessibility, safety (personnel and equipment), representativeness, suitability for project objectives and suitability for attachment of loggers. A total of 92 sites were selected for installation of water temperature loggers. An additional 50 sites from collaborative projects were also used. Figure 2.1 indicates the locality of each logger relative to the primary drainage regions and river network. Site information for each logger is summarised in Appendix 1 of this report.

## 2.2 Data collection

Hourly water temperature data were collected using Hobo UTB1-001 TidBit V2 loggers. The criteria for selection of loggers included logger accuracy and resolution, the degree to which the logger was waterproof, the amount of memory for data storage, cost (including value for money) and appropriateness of the technology for the environment. Air temperature and relative humidity data were collected using Dallas I-Buttons (DS1923 Dallas I-button), which included a radiation shield. The specifications of each logger are provided in Appendix 2. Details of logger assemblage and installation are provided in Rivers-Moore and Dallas (2008). Briefly, water temperature loggers were housed in protective steel casings which allowed for flow through of water. These were attached to an immovable object such as rocks (or trees) using a drill and Dyna bolts. To increase longevity of the logger housing it is recommended that stainless steel be used for all metal parts. Where possible, loggers were positioned in the thalwag of the stream, normally in a run. Installing loggers during the low flow period ensures that loggers remained submerged in the water. Water and air temperature data are provided on the CD that accompanies this report and are also archived at SAEON (South African Environmental Observation Network). Details of the data quality are tabulated together with site information in Appendix A on the CD.

### 2.2.1 Development of a simple site characterisation datasheet

A datasheet for initial characterisation of sites where water temperature loggers were installed was compiled (Appendix 2). It was based on the River Health Programme Site Characterisation Manual (Dallas 2005c) and was aimed at facilitating the standardised capture of all relevant site data. It included general site information, spatial information, location details, channel condition, condition of local catchment, channel morphology, water level, water chemistry, stream dimensions and substratum composition.

**Related Publication**

Rivers-Moore N.A. and Dallas H.F. 2008. Water temperatures and the Reserve (WRC Project: K5/1799): Report on water temperature logger assemblage. Report Number 1799/2 produced for the Water Research Commission. The Freshwater Consulting Group.

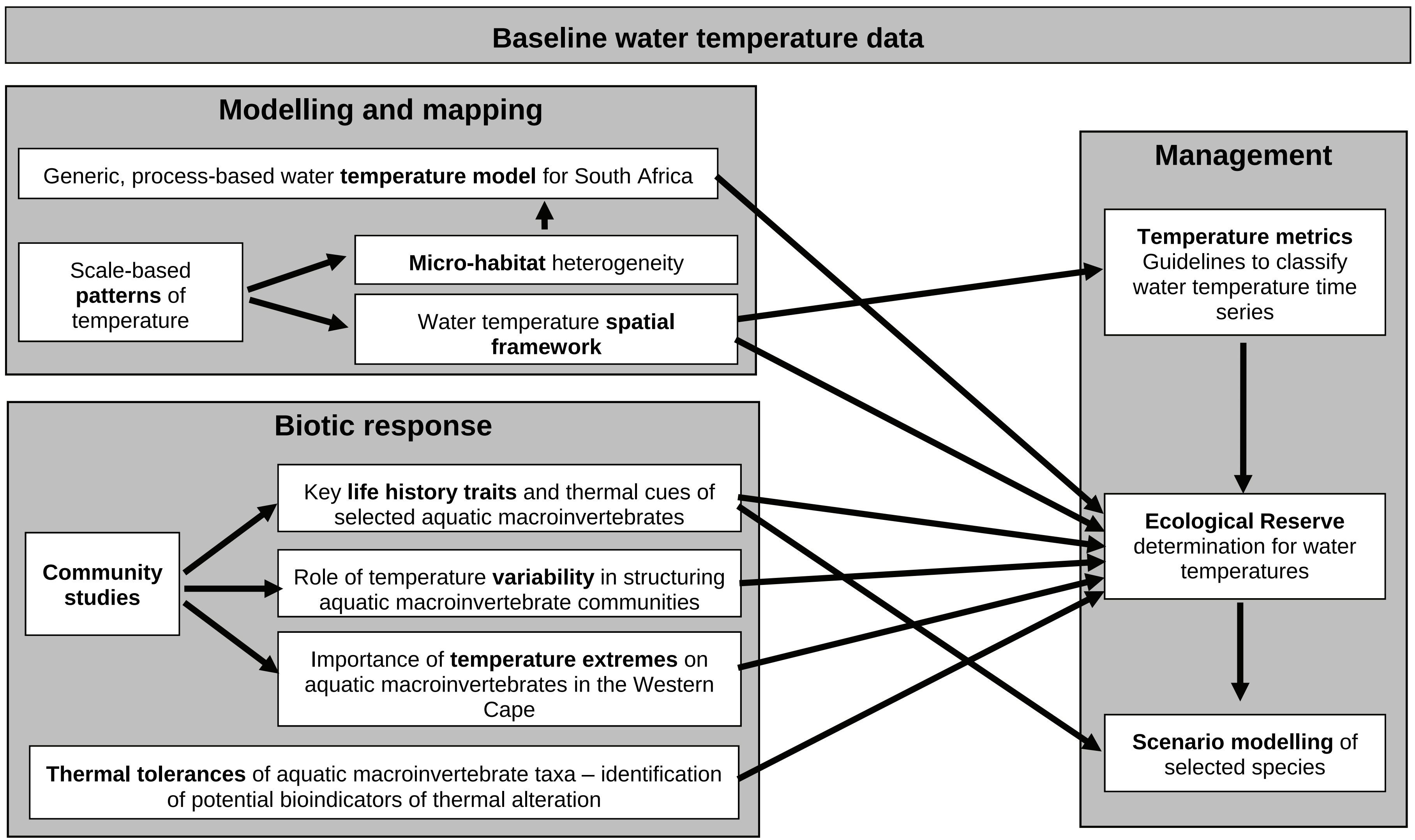


**Figure 1.1 Schematic diagram of the research project indicating the links between components**

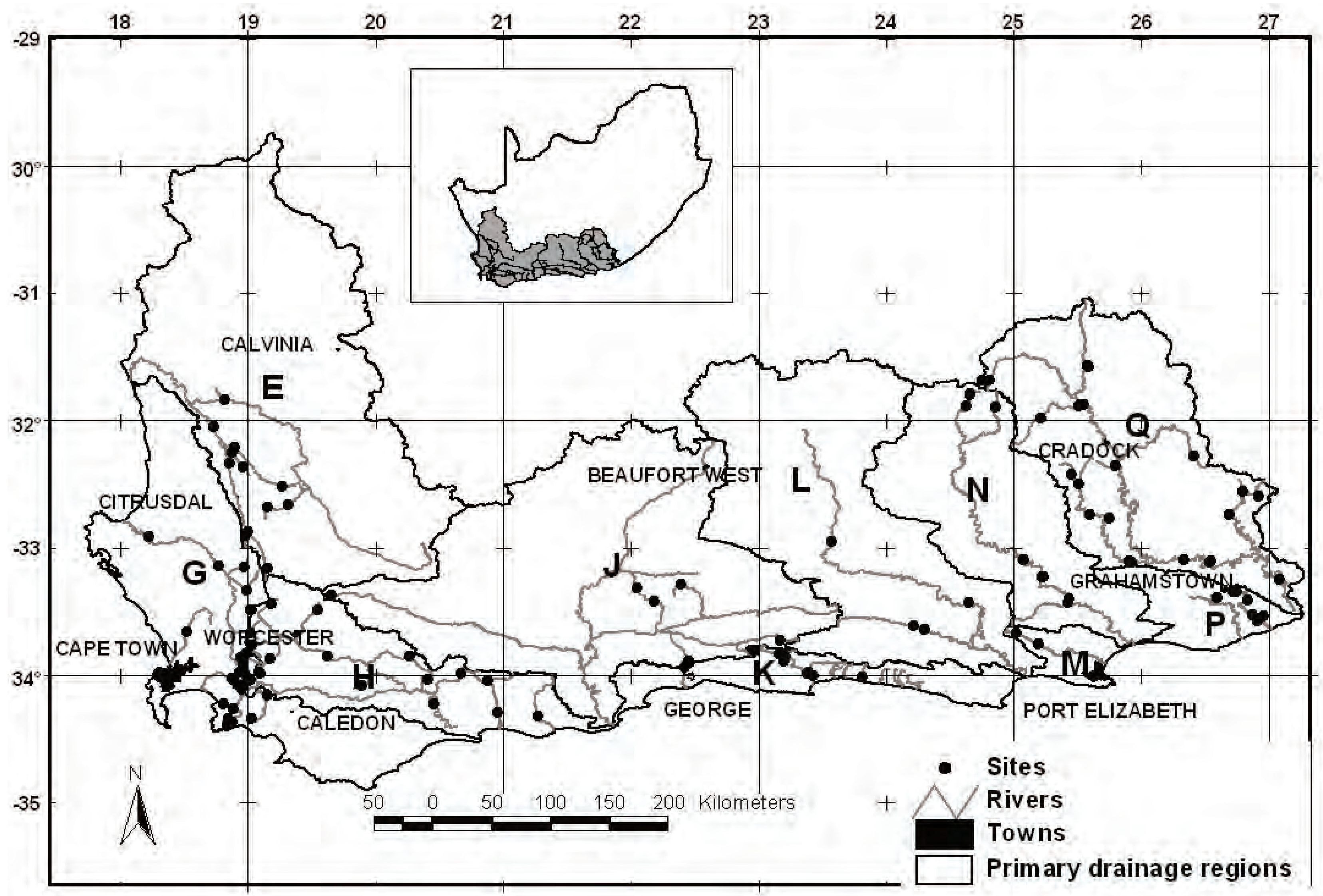


**Figure 2.1** Sites in the Western and Eastern Cape where water temperature data loggers were installed

# 3 MODELLING AND MAPPING

## 3.1 Generic, process-based water temperature model for South Africa

### 3.1.1 Relationship between water temperature and selected thermal buffers

**Related Publications**

Rivers-Moore NA, Mantel S and Dallas HF 2010a. Water temperatures and the Reserve (WRC Project: K5/1799): Development of a process-based understanding of selected water temperature modifiers (turbidity and groundwater contribution) in South African rivers. Report Number 1799/11 produced for the Water Research Commission. The Freshwater Consulting Group and the Institute for Water Research.

Rivers-Moore NA, Mantel S and Dallas HF 2010b. Water temperatures and the Reserve (WRC Project: K5/1799): A generic, process-based water temperature model for South Africa. Report Number 1799/16 produced for the Water Research Commission. The Freshwater Consulting Group and the Institute for Water Research.

Rivers-Moore NA and Mantel S 2011. Water temperatures and the Reserve (WRC Project: K5/1799): Final report on the evaluation of the water temperature model. Report Number 1799/24 produced for the Water Research Commission. The Institute for Water Research.

### Introduction and aims

Measured stream temperatures result from a number of assimilated processes, essentially summarized by Expressions 3.1 and 3.2 (Poole and Berman 2001). Water temperature is most simply described as the result of heat energy inputs diluted by flow volumes. Ward (1985) shows that a water temperature regime is the result of a complex interplay of variables at different spatial and temporal scales. Incoming solar radiation is, however, the major variable influencing both air and water temperatures, with air temperatures acting on water temperatures via convective energy exchanges (Johnson 2003). While air temperature (derived from solar radiation) is the variable of greatest leverage in water temperature models (Bartholow 1989), regional knowledge of the role of additional drivers, which are in turn acted upon by insulators and buffers, is required (Poole and Berman 2001) (Figure 3.1):

- **Drivers** operate beyond the boundaries of the stream, and control the rate at which heat and water are delivered to the stream system;
- **Insulators** influence the rate of heat exchange with the atmosphere;
- **Buffers** store heat already in the system, and integrate the variation in flow and temperature over time.

WT° α Heat energy/water volume [3.1]

WT° α Heat load/flow rate [3.2]

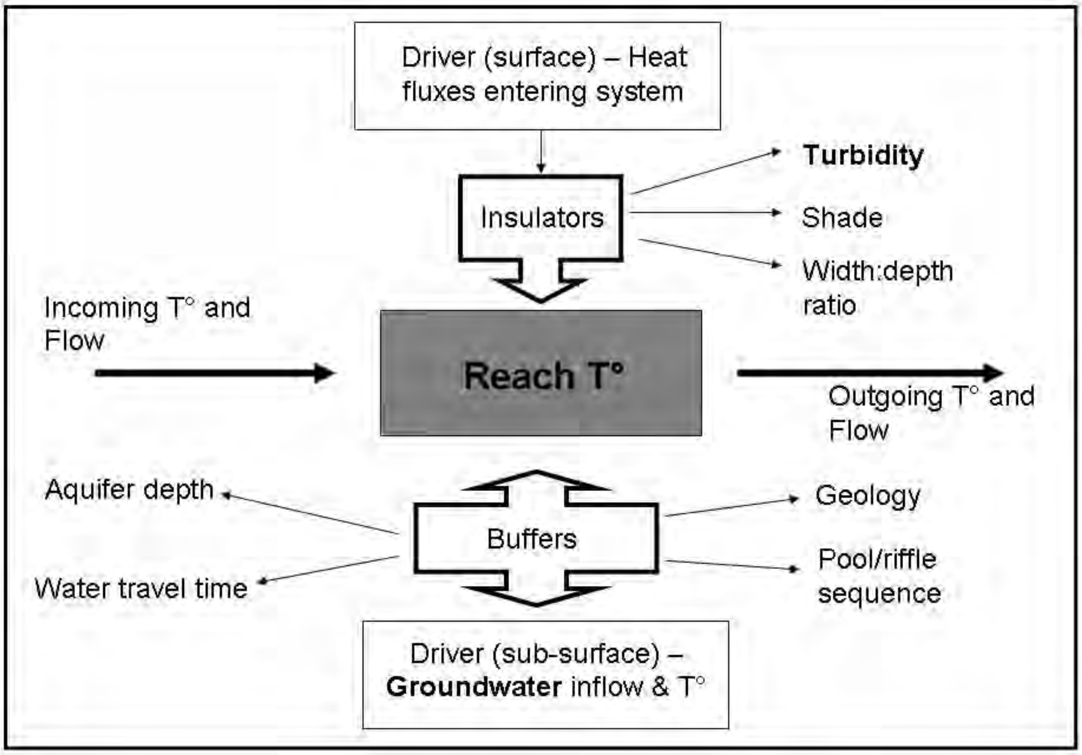


**Figure 3.1 Conceptual model of reach water temperatures (after Becker et al. 2004 and Poole and Berman 2001)**

In general, as streams become larger, insulating processes become less effective and buffering processes become more important (Poole and Berman 2001). The extent to which drivers and buffers modify the thermal regime of rivers in South Africa is not known. The identification of the magnitude of selected thermal modifiers within the South Africa context would provide insight into management of thermal changes in river systems. Human alteration of insulators and buffers, for example changing groundwater dynamics, channel morphology and turbidity, are critical pathways of human influence on channel water temperatures (Poole and Berman 2001).

The aim of this section was to further an understanding of groundwater as a driver of water temperatures in a river reach with respect to air temperatures, to consider the relationship between water temperature and aquifer depth (buffer effect of aquifer-surface water interactions), and to gain an understanding of the effects of turbidity (as an insulator) on diurnal water temperatures.

## Methods

### *Groundwater*

Water temperature data were related to broad-scale predictors (mean annual air temperatures), and groundwater contributions to flow (water volumes and groundwater depths). In the first instance, mean borehole temperatures were obtained from DWAF (2006) and recalculated as means per quaternary catchment. These mean values were correlated with mean annual air temperatures per quaternary catchment, obtained from Schulze (2007).

For the second component of this study (relating water temperatures to groundwater depths), a grid image of median average groundwater levels for South Africa was obtained from Colvin et al. (2007). Using this image, groundwater depths were assigned to each water temperature data logger site for this project (92 loggers plus an additional 30 from collaborative projects in the Western Cape). From this, mean daily water temperature range was calculated for a subset of 28 water temperature logging sites from the Eastern and Western Cape provinces, chosen to represent a range of groundwater depths. Mean daily temperature ranges were correlated with groundwater depths. At a finer scale, piezometer data (water levels in metres below ground level) and associated half-hourly water temperature data for 16 sites in the Table Mountain aquifer were analysed to determine relationships between water temperatures and groundwater dependence in rivers (City of Cape Town 2009).

### *Turbidity*

Experiments were undertaken for the heuristic potential in informing a water temperature model on the relative importance of turbidity as a model parameter. Two trials were undertaken to compare the effects of turbidity on water temperatures. In both trials, two Hobo TidbiT® v2 loggers (Onset 2008) programmed to record water temperatures at five minute intervals were placed in paired 20ℓ white buckets (clear and "turbid") on a sunny and overcast day for 26 and 44 hours respectively. Descriptive statistics and time series were based on mean values over hourly periods, using mean values from the paired data loggers. Solar radiation was not quantified beyond the qualitative observation of "sunny" versus "overcast".

## Summary of Major Results

### *Groundwater*

Mean borehole temperatures were available for 445 of the total 1946 quaternary catchments in South Africa. These temperatures ranged from 2.4 to 49.6°C, with a mean of 21.78±4.87°C. A trend of borehole temperature increase with longitudinal distance downstream was apparent, which was possibly due to groundwater levels being closer to the topographic surface further downstream, and thus influenced more by air temperatures, although borehole depths could not be obtained to verify this trend. The borehole temperatures showed anomalies with a number of extreme values, and these points were excluded as outliers. Mean borehole temperatures showed a weak correlation ($R^2$ = 0.19, $p < 0.0001$) with mean annual air temperatures (Figure 3.2). The slope line indicated that mean borehole temperatures were *ca.* 1.2 times higher than mean annual air temperatures.

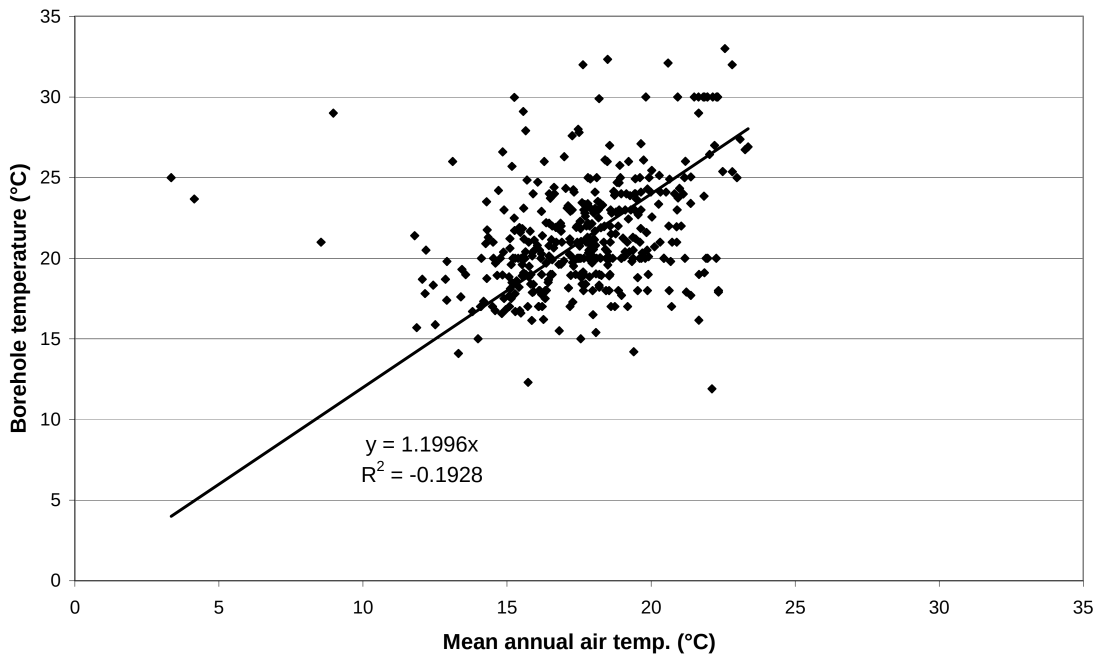


**Figure 3.2 Correlation between mean borehole temperatures and mean annual temperatures (n = 402), with regression line forced to zero**

Median average groundwater levels for South Africa ranged from 0 to 147 metres below ground level, with a median value of 22.02±57.94 m. At a regional scale, the mean daily water temperature ranges for 28 loggers showed no relationship with groundwater levels. At a finer scale, where water temperatures could be directly related to groundwater depths using piezometers (City of Cape Town 2009), mean daily water temperatures decreased with increasing groundwater contributions (negative j-curves, Figure 3.3). Data from 16 sites in the Western Cape showed two distinct patterns: In case A, correlations between rainfall and groundwater depths have been shown to be non-significant and these systems have been classified as groundwater dependant, while the latter (B) are likely to be more dependent on rainwater (surface flows). Of the total 16 sites examined, nine sites showed water temperature-groundwater depths similar to pattern A and had $R^2$ values of 0.36 to 0.86. High groundwater levels corresponded with decreased mean daily water temperatures, while decreases in groundwater levels corresponded with increases in water temperatures.

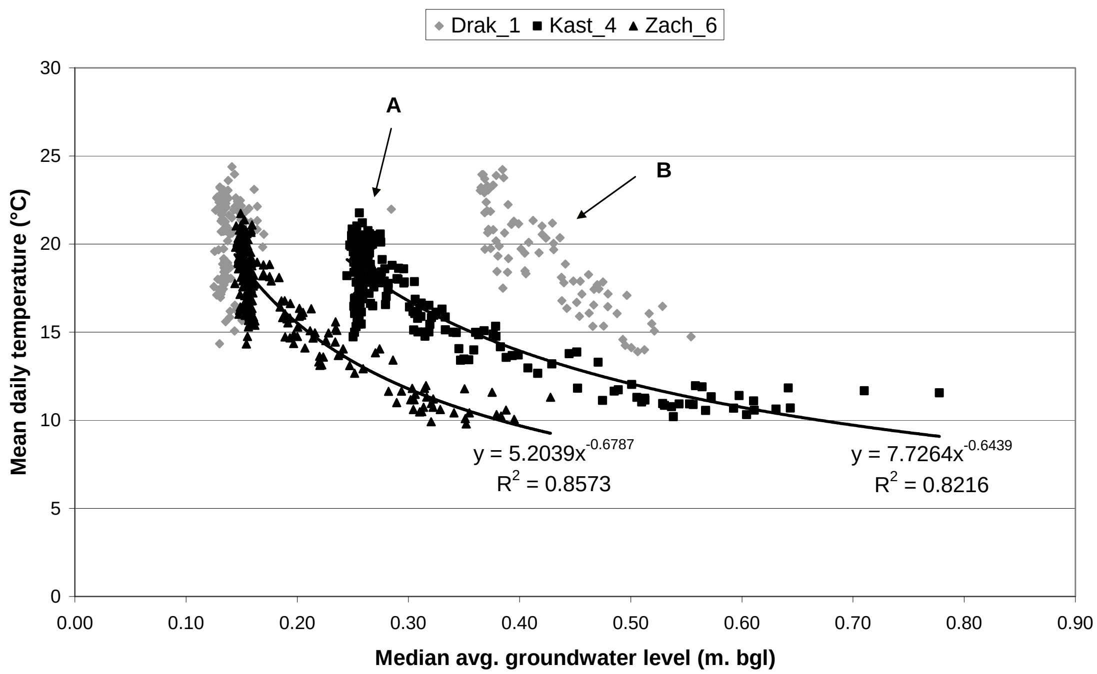


**Figure 3.3 Mean daily water temperatures vs. groundwater levels (metres below ground level) derived from piezometers for three tributary sites of the Wemmers River in the Western Cape province (Drak_1 = Drakenstein River; Kast_4 = Kasteelskloof River; Zach_6 = Zachariashoek River) located in the Table Mountain aquifer region**

### *Turbidity*

Water temperatures in clear versus "turbid" water showed distinctly different diurnal patterns, which further varied based on solar radiation. In both experiments, temperature time series for clear versus turbid water showed different heating patterns (Table 3.1). For example, under conditions of higher solar radiation (Figure 3.4), water temperatures in the bucket with turbid water were higher than in the clear water, and the daily maximum was 1.1°C higher. Differences between water temperatures in clear versus turbid water for experiment 1 (more sunshine and solar radiation) were more pronounced than for experiment 2 (longer periods of being overcast) (Figure 3.5).

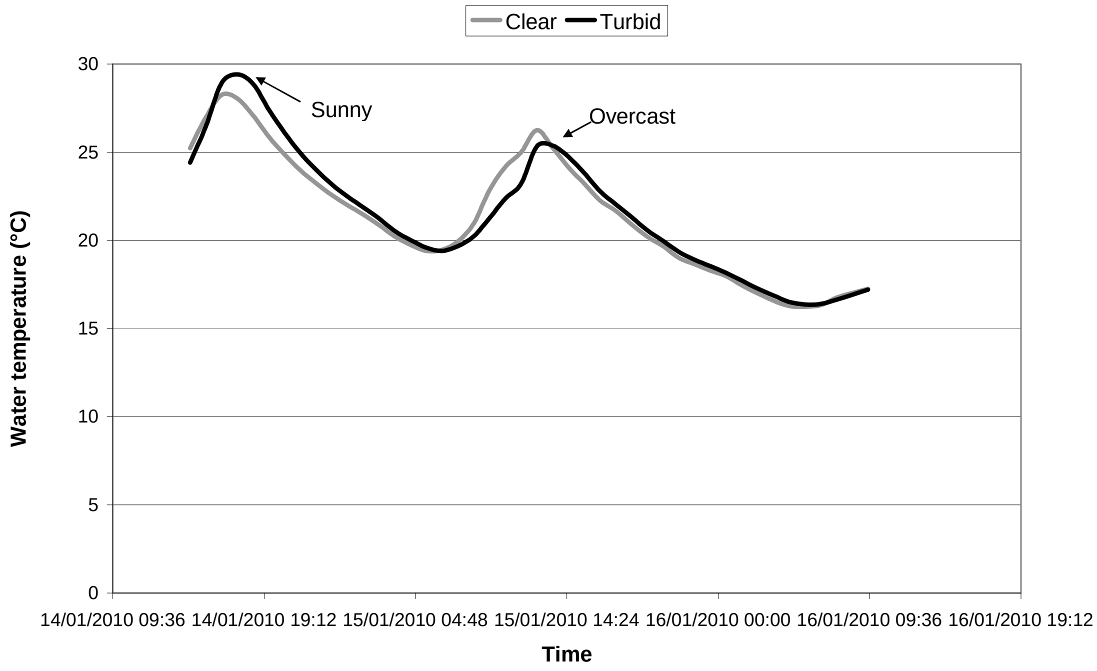


**Figure 3.4 Time series plot of mean hourly water temperatures in clear vs. turbid water (Experiment 1)**

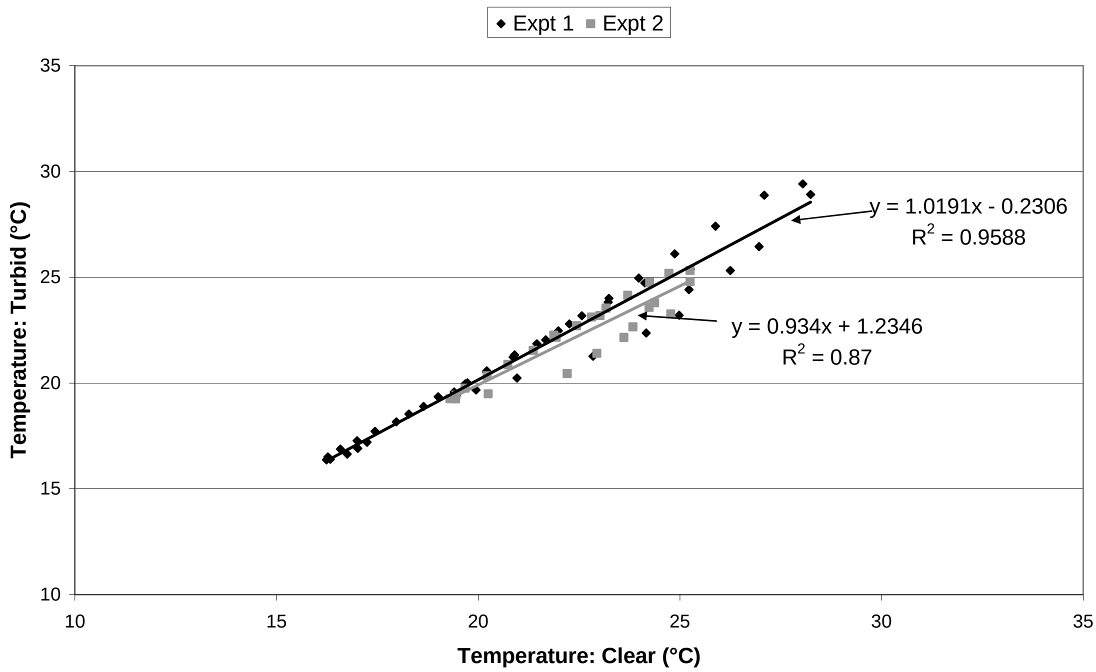


**Figure 3.5 Regression of mean hourly water temperatures from clear water vs. turbid water for two experiments**

**Table 3.1 Descriptive statistics of water temperatures logged at five-minute intervals from clear vs. turbid water in 20ℓ buckets**

| | Experiment 1 | | | Experiment 2 | | |
|---|---|---|---|---|---|---|
| | **Clear** | **Turbid** | **Diff.** | **Clear** | **Turbid** | **Diff.** |
| **Mean±SD** | 21.32±3.51 | 21.50±3.65 | +0.18 | 22.49± 1.92 | 22.24± 1.90 | -0.25 |
| **Max** | 28.32 | 29.45 | +1.11 | 25.76 | 25.38 | -0.38 |
| **Min** | 16.23 | 16.33 | +1.13 | 19.19 | 19.20 | 0.01 |
| **Range** | 12.09 | 13.12 | +1.03 | 6.57 | 6.18 | -0.39 |
| **Cumulative** | 11407.12 | 11501.09 | +93.97 | 7015.43 | 6937.76 | -77.76 |

## Conclusions and Recommendations

**Groundwater and surface water temperatures –**

- In the absence of better data, and based on international literature, mean annual air temperatures are a useful surrogate for baseline groundwater temperatures.
- Aquifer depth is an important buffer of groundwater temperatures. The influence of groundwater temperatures on the measured water temperature time series is a function of groundwater levels and volumes of groundwater contributing to overall flow volumes. Further analyses of water temperature time series, particularly with respect to seasonal variability patterns, are required.
- Further analysis of the deviation of water temperature time series from baseline temperatures is a suggested avenue of future research.

**Turbidity** –

- Is unlikely to play a large role in a model to simulate mean daily water temperatures.
- Affects maximum daily water temperatures and should be a variable in a daily maximum water temperature model, which will affect cumulative maximum degree days. A daily maximum water temperature model should have a constant value (>1°C) added to this daily maxima for turbid rivers.

### 3.1.2 Model development, testing, evaluation and verification

## Introduction and aims

Direct and indirect impacts on water temperatures occur through reductions of flows (groundwater and surface water) and catchment land use (indirect), all of which affect a river's natural thermal regime (Olden and Naiman 2010). Since the primary source of energy for warming streams is the sun, increasing or decreasing the amount of energy reaching a stream causes increases in maximum temperatures (Brown and Krygier 1970) or decreases in minimum temperatures. For example, forest canopy (shading) modifies the amount of solar radiation influencing stream temperatures: Johnson (2003) showed that for conditions at midday in summer, an open reach under full sun experienced a net energy gain of 580 $Wm^{-2}$ but a reach under full shade had a net loss of 149$Wm^{-2}$.

Additionally, relative flow contributions from different sources with different water temperatures should be incorporated into water temperatures leaving a reach through flow mixing. Understanding the dynamics of confluence zones and tributaries in maintaining thermal heterogeneity is one example (Olden and Naiman 2010). Surface-subsurface exchanges (typically expressed as groundwater inflow rate: $m^3s^{-1}km^{-2}$) of water have also been recognised as significantly influencing the heat budgets of small streams, because groundwater inflows add to heat storage capacity, and moderate annual variation. Story et al. (2003, cited in Webb et al. 2008) showed that groundwater inflow was responsible for about 40% of an approximately 3°C gross cooling effect in the daily maximum temperature. However, a complication to modelling is the temporal and spatial variability in groundwater-surface water interaction, and impacts on daily range and maximum values, and there are few studies which consider the heat fluxes that influence water temperatures in groundwater-dominated water courses (Webb and Zhang 1999).

Attempts at modelling water temperatures have been focused on either empirical models based on statistical analysis of meteorological data (for example, Rivers-Moore and Lorentz 2004; Rivers-Moore et al. 2005a), or physically-based models solving the heat budget equation (Brown 1969; Bartholow 1989, 2002; Sullivan et al. 1990). One of the first widely used process-based models was developed by Brown (1969), and is regarded as being simple to use, and of value in providing a reasonable index of change in the maximum temperature with a change in stream shading (Sullivan et al. 1990).

Many advances in modelling have come through the need to assess, understand and mitigate human impacts on thermal behaviour in water courses, and further impetus for models has come from the impacts of climate change on river systems (Webb et al. 2008). A simple, parsimonious model, while being the most pragmatic to use because it requires fewer data inputs, does not have the functionality required to assess direct and indirect impacts on water temperatures. A more complex, process-based model, however, has the potential to incorporate multiple heat flux variables, providing decision makers with the facility to assess different scenarios.

For a water temperature model useful to water managers in South Africa, Rivers-Moore et al. (2010a) recommended that it was most practical to separate a hydrograph into surface and subsurface (groundwater) flow components, and that groundwater temperatures could be approximated using mean annual air temperatures. Turbidity was also a necessary variable to incorporate into a temperature model, which could be achieved by changing the albedo value of water (Paaimans et al. 2008a), which was shown to have an effect particularly on maximum daily water temperatures.

The aim of this section was to develop a generic water temperature modelling system for South Africa, using a suitable process-based water temperature model, to simulate daily minimum, maximum and mean water temperatures.

**Conceptual requirements**

Rivers-Moore et al. (2008c) list a number of attributes which an “ideal” water temperature model for South African rivers should:

- use data which are readily and widely available (air temperatures and solar radiation as major drivers),
- integrate with other models, and use outputs from these,
- be useful in scenario analyses (e.g. incorporate flow, turbidity and riparian shading terms, such as for dam releases; climate change). Evaluation of temperature signals needs to be attached to different flow components, i.e. disaggregation of (monthly) flows into groundwater vs. instream (surface and interflow) flows.
- have a model time step 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,
- be generic i.e. be applicable at a range of spatial scales and applicable throughout South Africa,
- be simple, with as few terms as possible.

## Methods

Four study sites were selected based on their heuristic value for testing and evaluating different water temperature models (Table 3.2 and Figure 3.6). Data logger codes (see Appendix 1) associated with each study are:

1. Cluster of loggers = K6Keur-Outen1; K6Keur-Outen2 and K6Keur-Kwaai
2. Q9Fish-Gh012
3. Cluster of groundwater- and non-groundwater dependant sites = G4Palm-Kogfr; K4Dwars and K2a-Oudebos
4. Q1Brak-Dekeur

**Table 3.2 Study sites and heuristic value of sites**

| Site | Latitude | Longitude | Heuristic Value |
|---|---|---|---|
| 1 | -33.821880 | 23.182140 | Impact of segment length and tributary on water temperatures |
| 2 | -33.087540 | 26.436360 | Impact of flow regulation on water temperatures |
| 3 | -34.327380 | 18.961330 | Investigate different contributions of groundwater volumes on water temperatures |
| 4 | -31.600660 | 25.492870 | Investigate impact of air temperatures at different distances on water temperatures |

### *Modelling approach and development*

Hourly relative humidity (percentages), air and water temperature data were converted to daily statistics (mean, maximum and minimum). Time periods for model testing were limited to periods where all data were overlapping: 11 months in the case of sites 1, 2 and 4 (25 June 2009 - 24 May 2010) and five months for site 3 (12 December 2008 - 27 April 2009). Three different models were used to evaluate water temperature predictions, viz. statistical, and two process-based models with differing levels of complexity. These models were assessed using a number of different scenarios (change in turbidity; proximity to air temperature station; varying segment lengths, etc.) for sites 1 to 4.

### Statistical model

Statistical models to simulate primarily daily maximum, and in limited cases daily mean, water temperatures, have previously been developed for site- or river-specific reasons (Rivers-Moore et al. 2004; Rivers-Moore et al. 2008a). For this study, more generic models based on a wider array of sites were required. Data from 39 sites from rivers in the Eastern and Western Cape (13 lowland and 26 upland) were used to generate models for daily water temperatures (mean, minimum and maximum). Two sets of water temperature models were developed – simple models that used data for air temperature (AT) and % relative humidity (%RH), and more complex models that additionally used data for stream order (SO) and altitude. Stepwise forward multiple regression was used to generate the models (Table 3.3).

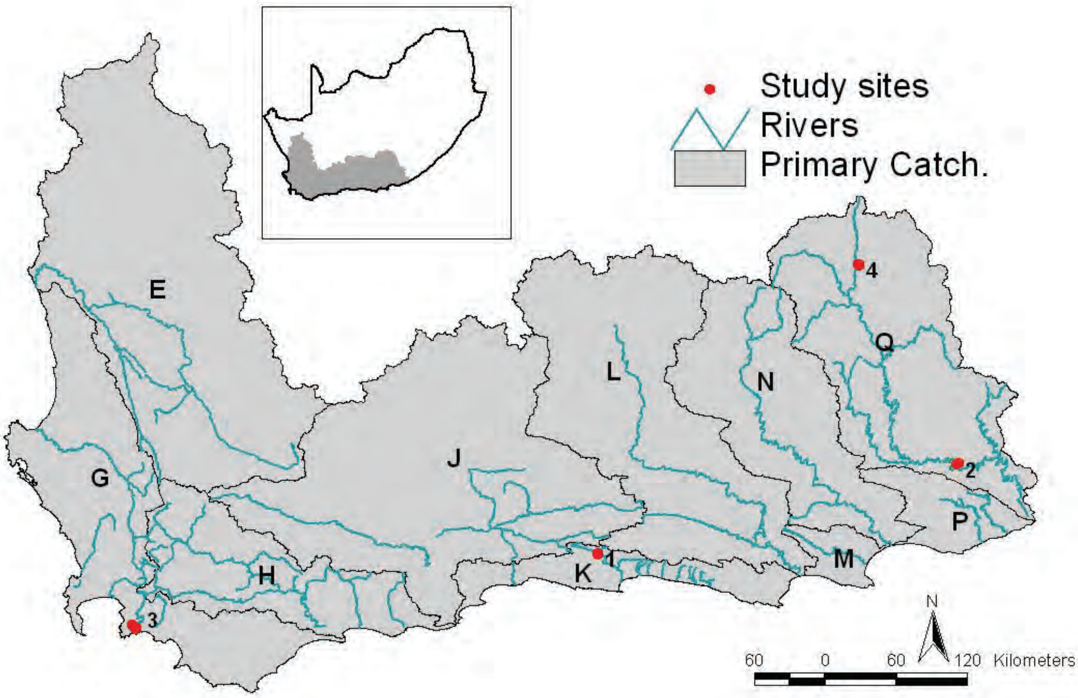


**Figure 3.6 Study sites selected for model assessment based on their heuristic value (see Table 3.2)**

**Process-based model SSTEMP**

The Stream Segment Temperature (SSTEMP) model is a widely used daily time-step process-based temperature model, and is an adaptation of the stream network temperature model (SNTEMP) by Theurer et al. (1984), whose algorithms continue to be used in thermal studies (Webb and Zhang 1999). The model predicts maximum, minimum and mean temperatures at user-specified points and relies on measurement of a number of site and basin-specific variables, typically applied as data per ecoregion. SSTEMP uses 24-hour averages of input values to predict daily temperatures. "To predict the daily maximum, the model begins with the 24hour mean value at solar noon, and models the stream's response up to solar sunset, predicting the maximum. To estimate the minimum, the model makes a mirror image of the curve between the mean and the maximum by subtracting their difference from the mean (Sullivan et al. 1990, p. 110).

**Table 3.3 Generic regression models developed from Eastern and Western Cape temperature data, based on inputs of mean daily air temperature (AT), mean daily percentage relative humidity (RH), stream order (SO) and altitude (Alt)**

| Model Family | Metric | Equation | Mult. $R^2$ |
|---|---|---|---|
| All data (simple) | Mean – F(2, 9878) = 8062 | $WT = 0.46 + 0.70(AT) + 0.07(RH)$ | 0.62 |
| | Min– F(2, 98786) = 8062 | $WT = -2.34 + 0.72(AT) + 0.08(RH)$ | 0.61 |
| | Max – F(2, 9878) = 8062 | $WT = 3.97 + 0.69(AT) + 0.06(RH)$ | 0.47 |
| All data (complex) | Mean – F(4, 9876) = 8062 | $WT = -2.33 + 0.70(AT) + 0.08(RH) + 0.93(SO) + 0.001(Alt)$ | 0.69 |
| | Min – F(3, 9877) = 8062 | $WT = -4.33 + 0.71(AT) + 0.09(RH) + 0.87(SO)$ | 0.66 |
| | Max – F(4, 9876) = 8062 | $WT = 0.70(AT) + 0.07(RH) + 1.01(SO) + 0.001(Alt)$ | 0.53 |
| Upland (simple) | Mean – F(2, 6493) = 4838 | $WT = -5.07 + 0.68(AT) + 1.27\sqrt{RH}$ | $0.60^{*}$ |
| | Min – F(2, 6493) = 5204 | $WT = -3.24 + 0.70(AT) + 0.09(RH)$ | 0.62 |
| | Max – F(2, 6493) = 2313 | $WT = 4.13 + 0.65(AT) + 0.06(RH)$ | 0.42 |
| Upland (complex) | Mean – F(4,6491) = 3390 | $WT = -5.32 + 0.73(AT) + 0.10(RH) + 1.50(SO) + 0.001(Alt)$ | 0.68 |
| | Min – F(4,6491) = 3029 | $WT = -6.57 + 0.73(AT) + 0.11(RH) + 1.02(SO) + 0.001(Alt)$ | 0.65 |
| | Max – F(4,6491) = 1841 | $WT = -3.5 + 0.74(AT) + 0.09(RH) + 2.12(SO) + 0.002(Alt)$ | 0.53 |
| Lowland (simple) | Mean – F(2, 3382) = 3411 | $WT = 1.87 + 0.71(AT) + 0.06(RH)$ | 0.67 |
| | Min – F(2, 3382) = 2523 | $WT = -3.77 + 0.69(AT) + 1.08\sqrt{RH}$ | 0.62 |
| | Max – F(2, 3382) = 2362 | $WT = 3.48 + 0.73(AT) + 0.06(RH)$ | 0.58 |
| Lowland (complex) | Mean – F(4, 3380) = 1875 | $WT = 0.76 + 0.69(AT) + 0.06(RH) + 0.48(SO)$ | 0.69 |
| | Min – F(4, 3380) = 1392 | $WT = 3.89 + 0.60(AT) + 0.19(AT_{\min}) - 0.10(AT_{\max}) + 0.04(RH) + 0.37(SO) - 0.08\ln(Alt)$ | 0.66 |
| | Max – F(4, 3380) = 1284 | $WT = 2.19 + 0.72(AT) + 0.05(RH) + 0.48(SO) + 0.001(Alt)$ | 0.60 |

**Process-based flow component model**

The third model was based on existing one-dimensional process-based models (e.g. used by Becker et al. 2004 – Figure 3.7; Marcé and Armengol, 2008) for estimating groundwater contributions to overall flows using temperature signals was adapted to estimate downstream water temperatures (Equation 3.3). Included in this model is a flow mixing component to incorporate the relative contribution of different components of the flow hydrograph at different temperatures. In this model, surface heat fluxes (F) were reduced to incoming solar radiation only, since this accounts for 90% of the heat fluxes, while surface areas was a product of segment length and width, as estimated for the SSTEMP model. Model variables were populated using the same data as derived for the SSTEMP model.

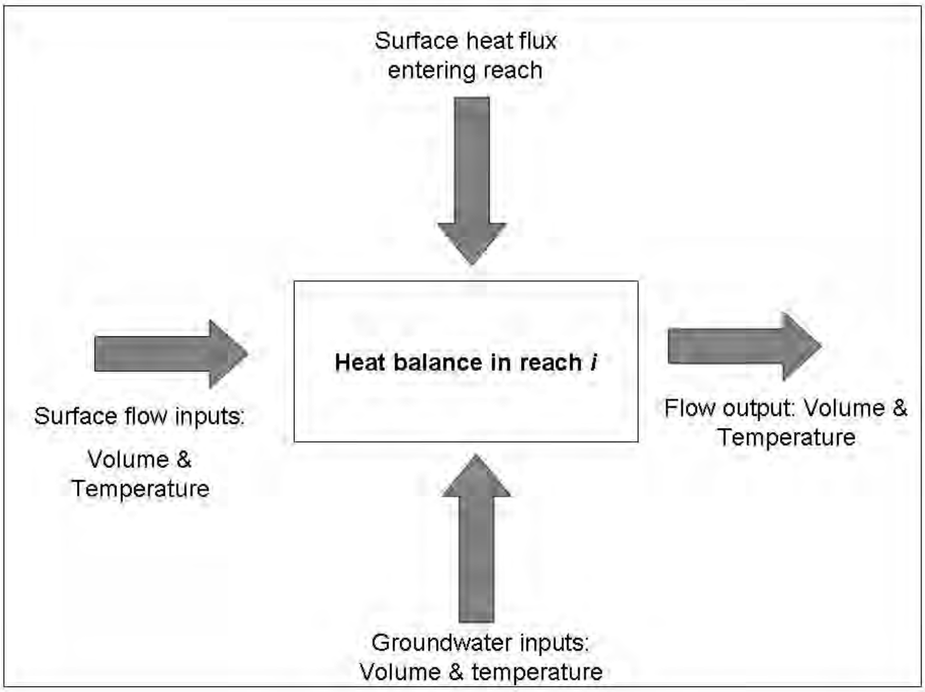


**Figure 3.7 Stream inputs model, showing conceptual approach to estimating heat balance in reach *i*. In this conceptual diagram, heat balance in reach *i* is a function of inflow and its associated water temperature. In reach *i*, inflow water temperatures are modified by groundwater inputs (as estimates of its heat dilution capacity based on groundwater volumes and the temperature of the groundwater), and energy inputs from incoming solar radiation (heat flux entering reach)**

$$T_i = \frac{\rho c Q_{i-1} T_{i-1} + \rho c Q_{gi} T_g + FA}{\rho c Q_i} \qquad [3.3]$$

where $T_i$ is water temperature in reach *i*; $\rho$ and *c* are water density and heat capacity; *Q* is flow volume, *F* is surface heat flux and *A* is stream surface area; and the subscript *g* is groundwater (after Becker et al. 2004 p. 227)

### Model testing

For the model testing, daily mean, minimum and maximum time series were simulated for the four study sites (Table 3.2), using the statistical models, and both process-based models (SSTEMP and flow-component). Four techniques were used for quantitative evaluation of models. The first approach was to measure correlation between observed and predicted data, where the regression slope and $R^2$ values provide a measure of deviation of the residuals from the mean. The second approach was based on Rivers-Moore et al. (2008c), who calculated the mean percentage difference between observed and predicted water temperatures (Equation 3.4). A third approach used was the WSTAT (W-Statistic; Sullivan et al. 1990) (Equation 3.5), where a negative sign shows under-simulation, while a positive sign shows over-simulation. The fourth approach, similar to the WSTAT test, was to calculate the RMSE (root mean square error), based on the square root of summed differences between expected and observed data (Equation 3.6) (also see, for example, Benyahya et al. 2009 citing Janssen and Heuberger (1995); Rivers-Moore et al. 2005a). The last three techniques provide a measure of the mean value of the residuals between observed and predicted data.

$$\frac{\overline{P_i} - \overline{O_i}}{\overline{O_i}} * 100 \qquad [3.4]$$

$$WSTAT = \frac{\sum_{i=1}^{N} (P_i - O_i)}{N} \qquad [3.5]$$

$$RMSE = \sqrt{\frac{\sum_{i=1}^{N} (O_i - P_i)^2}{N}} \qquad [3.6]$$

where N = number of daily water temperature observations, $O_i$ is observed and $P_i$ is predicted water temperature

To test the models under different scenarios (clear versus turbid; upstream dam versus no dam, etc.), observed and simulated (SSTEMP and simple, lowland regression models) time series data for site 2 (Great Fish River) were analysed using the temperature metrics developed by Rivers-Moore et al. (2010b, and Section 5.1) to facilitate further model comparisons. Metric data were first correlated to reduce redundant data, prior to assessing which model was closest to observed data using a principal components analysis.

**Model evaluation**

For model evaluation, models were assessed according to criteria by Rivers-Moore et al. (2008c), and models were assigned a qualitative score of 0 to 2 (poor/addresses issue/addresses issue well).

**Model verification**

Models developed from one year's temperature data were used to simulate daily mean, minimum and maximum water temperatures for a subsequent year. Statistical models were selected from Rivers-Moore and Mantel (2011) based on which model had the highest $R^2$. Water temperature data were simulated for 1 to 2 sites per thermal region (Section 3.3.2) using hourly (or half-hourly for some sites in the Western Cape) water temperature data from 1 January 2010 to 31 December 2010. Daily water temperatures were simulated using inputs of mean daily air temperatures (°C – *AT*), mean daily relative humidity (% – *RH*), stream order (*SO*) and altitude (m – *Alt*). Time series were simulated for mean daily (Equation 3.7), minimum daily (Equation 3.8) and maximum daily (Lowland sites = Equation 3.9, and Upland sites = Equation 3.10).

$$MeanWT = -2.33 + 0.70(AT) + 0.08(RH) + 0.93(SO) + 0.001(Alt) \quad [3.7]$$

$$MinWT = -4.33 + 0.71(AT) + 0.09(RH) + 0.87(SO) \quad [3.8]$$

$$MaxWT = 2.19 + 0.72(AT) + 0.05(RH) + 0.48(SO) + 0.001(Alt) \quad [3.9]$$

$$MaxWT = -3.5 + 0.74(AT) + 0.09(RH) + 2.12(SO) + 0.002(Alt) \quad [3.10]$$

where *WT* is water temperature, *AT* is mean daily air temperature, *SO* is stream order, and *Alt* is altitude (m amsl)

Next, model performance in the process-based model SSTEMP was verified using simulations of mean, minimum and maximum water temperatures for three sites on the Great Fish River (Figure 3.8 and Table 3.4, where the most upstream site – Q9Fish-Mort – was used as the input site to similate water temperatures for the first of three downstream sites). These sites were chosen because of access to good water and air temperature data, and daily flow data. The flow characteristics of the Great Fish River for the segment of interest were described by Rivers-Moore et al. (2008a). Of relevance is that the three simulated sites are downstream of a major impoundment, while input water temperature for the first site (Q9Fish-Carli) was from a site upstream of the dam. The third site (Q9Fish-Trum) has a gauging weir associated with it, but flow data ceased in March 1956. Consequently flows for this site were based on flows from the upstream weir (Q9H012) with a 10% reduction for abstractions. Model simulations were based on input water temperatures from observed data, as well as simulated mean daily water temperatures using Equation 3.7. Sensitivity analyses were undertaken using the algorithms contained in SSTEMP, which were based on a 10% variation for all input data. For both levels of model verification, quantitative assessments were undertaken using the four techniques described above. Qualitative assessments were also done using visual comparisons between observed and simulated time series.

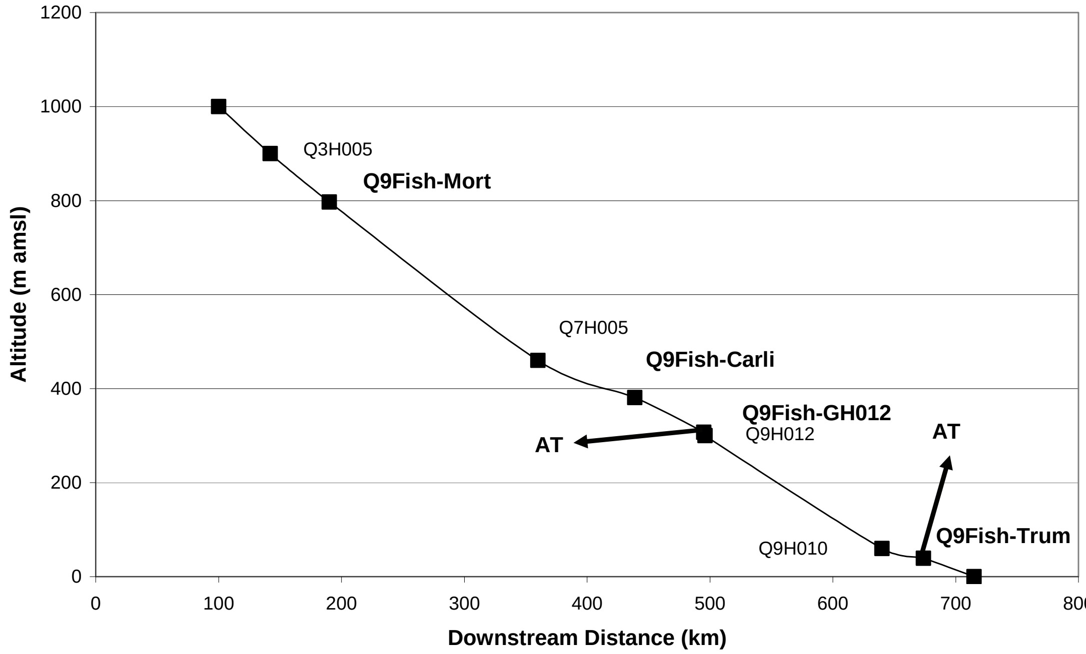


**Figure 3.8 Four sites on the Great Fish River sites used for data patching, showing altitude and downstream distance of sites. Location of closest flow gauging weir and air temperature is indicated**

**Table 3.4 Great Fish River water temperature sites, and associated gauging weirs and air temperature (AT) sites**

| Name | Distance Downstream (km) | Altitude (m amsl) | Weir | Data quality | AT |
|---|---|---|---|---|---|
| Q9Fish-Mort | 190 | 797 | Q3H005 | Good; Missing 19 June 2009 - 7 March 2010 | Q9Fish-Con[1] |
| Q9Fish-Carli | 439 | 381 | Q7H005 | Good | Q9Fish-Con[2] |
| Q9Fish-GH012 | 495 | 307 | Q9H012 | Good; Missing from 13 July 2010 | Q9Fish-Con |
| Q9Fish-Trum | 674 | 39 | Q9H010 | Exposed from 15 August 2010 | Q9Fish-Trump |

[1]3.43°C adiabatic lapse rate correction

[2]0.52°C adiabatic lapse rate correction

## Summary of Major Results

### *Model testing*

Time series of simulated data showed a range of correlations with observed data for some models, and obvious poor simulation for other models (Figure 3.9). From a more quantitative assessment of the models using the different statistics (Table 3.5), a number of pertinent points emerged. For mean daily water temperatures, the flow component model (model 3) gave the poorest results; in relative terms, this model performed best (compared against itself at different sites) for the groundwater-dependant sites. Using default values as input into the SSTEMP model gave the best results for site 2, although differences did emerge when input variables (turbidity, upstream dam) were included. Choice of segment length (site 1) affected model accuracy. Distance of air temperature station for water temperature site made little difference to regression models.

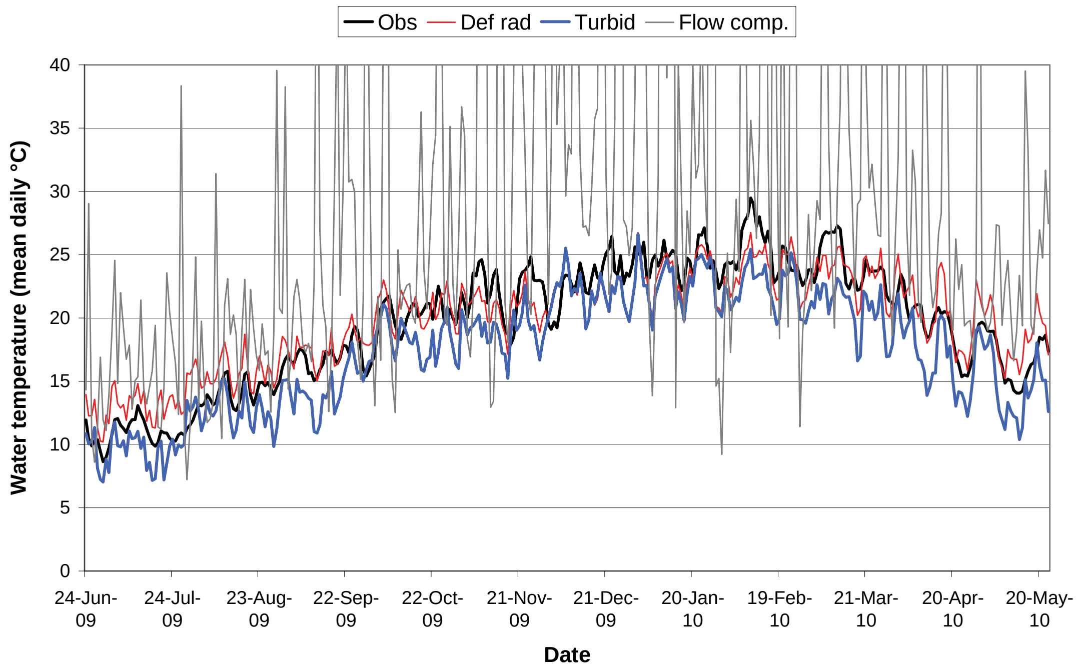


**Figure 3.9 Time series of observed versus simulated mean daily water temperatures for the SSTEMP model (two scenarios – Def rad. = default solar radiation, and Turbid = solar radiation values from ACRU scenario and multiplied using an albedo coefficient of 0.92 – after Paaijmans et al. 2008) and the process-based flow component model**

Prediction of minimum daily water temperatures was generally poorer than mean daily water temperatures, although the same trends as described above still applied. In contrast to the mean daily temperature models, the closer air temperature station yielded considerably better results than the station further away. All models were better at predicting maximum rather than minimum daily water temperatures. Accuracy of maximum daily temperatures was improved by including the upstream dam function for the Great Fish River, which is an impounded river.

**Table 3.5 Model performance for daily mean water temperatures (Models 1-3 refer to statistical, SSTEMP and Flow component models respectively). Tables for minimum and maximum water temperature models in Rivers-Moore et al. (2010c)**

| Site | Model | Variation | RMSE | Mean±SD | WSTAT | $R^2$ |
|---|---|---|---|---|---|---|
| 1 | 1 | All | 2.15 | -0.22±14.13 | -0.52 | 0.808 |
| | 1 | Upper | 2.11 | 3.96±15.25 | 0.22 | 0.799 |
| | 2 | Short segment | 1.43 | 1.20±9.54 | -0.09 | 0.907 |
| | 2 | Long segment | 1.49 | -0.84±9.12 | -0.43 | 0.909 |
| | 3 | | 7.60 | -38.86±18.01 | -6.78 | 0.453 |
| 2 | 1 | All | 2.62 | -1.15±15.49 | -0.83 | 0.727 |
| | 1 | Lower | 2.46 | 3.61±16.69 | 0.04 | 0.802 |
| | 2 | Default radiation | 1.95 | 3.38±11.27 | 0.24 | 0.866 |
| | 2 | ACRU radiation | 2.43 | -8.81±8.93 | -1.72 | 0.875 |
| | 2 | Turbid | 2.75 | -10.87±8.96 | -2.12 | 0.871 |
| | 2 | Clear | 3.00 | -12.41±9.01 | -2.43 | 0.867 |
| | 2 | No dam | 2.75 | -10.87±8.96 | -2.12 | 0.871 |
| | 3 | | 17.17 | 48.24±67.52 | 9.43 | 0.220 |
| 3 | 1 | All | 2.98 | 16.15±4.34 | 2.86 | 0.771 |
| | 1 | Upper | 2.94 | 16.05±4.04 | 2.84 | 0.782 |
| | 2 | Gw input | 3.44 | -14.70±3.83 | -3.29 | 0.742 |
| | 2 | Non-gw input | 3.75 | -15.92±4.34 | -3.57 | 0.742 |
| | 3 | | 4.21 | -16.19±16.26 | -3.02 | 0.117 |
| 4 | 1 | All – AT close | 2.24 | -3.85±13.60 | -0.90 | 0.791 |
| | 1 | Lower | 2.10 | 1.90±14.73 | 0.00 | 0.779 |
| | 1 | All – AT far | 2.18 | -2.14±14.02 | -0.61 | 0.781 |
| | 1 | Lower | 2.17 | 3.85±15.19 | 0.32 | 0.769 |
| | 2 | | N/A | N/A | N/A | N/A |
| | 3 | | N/A | N/A | N/A | N/A |

The PCA using a reduced set of metrics (Section 5.1) (monthly values were excluded as these were highly correlated with a number of other metrics) showed that the simulated time series most closely relating to the observed data was for the SSTEMP model where default values were used to calculate incoming solar radiation (Figure 3.10). The first two axes explained 86% of variation in data. The first PCA axis was positively correlated with minimum and mean metrics, and negatively correlated with SD, CV and Julian maximum values. The second PCA axis was negatively correlated with maximum metrics and positively correlated with Julian minimum value. The regression models were strongly biased by axis 1, while the SSTEMP model (under different scenarios) was more strongly affected by axis 2 (Table 3.6).

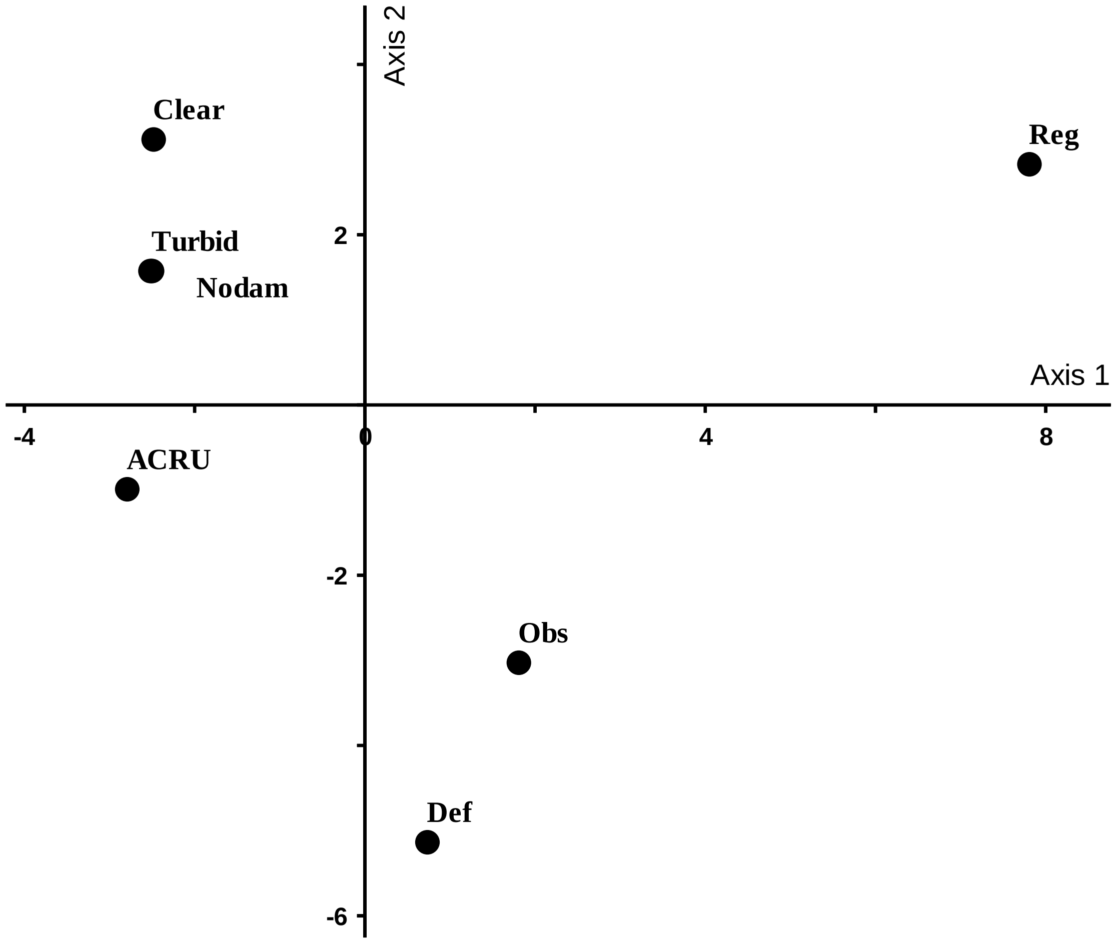


**Figure 3.10 PCA of observed versus simulated (SSTEMP and statistical models) daily mean, maximum and minimum water temperature data for site 2 (Great Fish River). Scenario labels are: Obs = observed data; Reg = simulated data using linear regression models; Def = default solar radiation used in SSTEMP based on fixed solar radiation; ACRU = raw simulated daily solar radiation values; Clear = solar radiation values from ACRU scenario and multiplied using an albedo coefficient of 0.86; Turbid = solar radiation values from ACRU scenario and multiplied using an albedo coefficient of 0.92 – after Paaijmans et al. (2008); Nodam = as per the Turbid scenario but with no upstream dam indicated**

***Model evaluation***

SSTEMP performed well for all model criteria used, while the flow component model performed poorly (Table 3.7). The statistical model performed well and would be suitable for simulating water temperatures as input into a process-based model, such as SSTEMP, for scenario analysis.

**Table 3.6 Eigenvalues for PCA of metrics of observed and simulated water temperature time series**

| | Axis 1 | Axis 2 |
|---|---|---|
| **Eigenvalue** | 13.06 | 8.45 |
| **% Cum var.** | 52.26 | 86.07 |
| Mean temperature | 0.2193 | -0.2091 |
| SD of mean temp. | -0.2231 | -0.0230 |
| CV of mean temp. | -0.2453 | 0.0590 |
| Predictability | 0.0504 | -0.2782 |
| Mean of minima | 0.2436 | -0.0775 |
| Mean of maxima | 0.0852 | -0.2624 |
| Mean Range | -0.1695 | -0.0947 |
| Min_1 | 0.2746 | -0.0121 |
| Min_3 | 0.2759 | 0.0014 |
| Min_7 | 0.2760 | -0.0091 |
| Min_30 | 0.2756 | -0.0009 |
| Min_90 | 0.2722 | -0.0405 |
| Max_1 | -0.0042 | -0.2884 |
| Max_3 | -0.1383 | -0.2808 |
| Max_7 | -0.0132 | -0.3399 |
| Max_30 | -0.1707 | -0.2581 |
| Max_90 | -0.1314 | -0.2989 |
| Degree days | 0.2193 | -0.2091 |
| Max. Range | -0.2369 | -0.0715 |
| Min T threshold count | -0.2518 | 0.1221 |
| Max T threshold count | -0.0866 | -0.3106 |
| Min T dur | -0.2644 | -0.0248 |
| Max T du | -0.1168 | -0.3036 |
| Julian date of min. | -0.0296 | 0.3094 |
| Julian date of max | -0.2027 | 0.0832 |

**Table 3.7 Model criteria from Rivers-Moore et al. 2008c and model performance of three different models**

| Model criterion | SSTEMP | Flow comp. model | Statistical model |
|---|---|---|---|
| Readily available data | 1 | 1 | 2 |
| Integrate with other models | 2 | 1 | 1 |
| Scenario analyses | 2 | 1 | 0 |
| Suitable timestep | 2 | 2 | 2 |
| Time series disaggregation | 2 | 1 | 1 |
| Build on existing research | 2 | 1 | 2 |
| Simple | 1 | 1 | 2 |
| Dynamic potential | 1 | 0 | 0 |
| **Score** | 13 | 8 | 10 |

***Model verification***

For the statistical models, it was difficult to assess model performance using a single accuracy metric over another, as each metric yielded different site rankings. In terms of three of the accuracy metrics (RMSE, WSTAT and % mean difference), best model performance was for daily minimum water temperatures, followed by mean daily and last of all, maximum daily water temperature simulations. Based on correlations between observed and predicted data ($R^2$), once again models for maximum daily water temperatures were worst, but conversely mean daily models performed better than minimum daily models using this criterion (Table 3.8; results from daily minima and maxima are included in Rivers-Moore and Mantel, 2011). Nevertheless, correlations between observed and predicted daily water temperatures were generally between 0.6-0.9, with mean $R^2$ > 0.75 for all three metrics (mean, minimum and maximum). The WSTAT metric was useful in indicating whether models over- or under-simulated. No trends were apparent here, with all models generally over-simulating, and fewest under-simulations occurring for daily maximum water temperatures. Again, over-simulation for one metric at a site did not translate into over-simulation for a different metric at the same site. Percentage mean differences between observed and simulated data showed a wide range of standard deviation in over- and under-simulation of data. Inaccuracies in simulations for different metrics (minimum, maximum and mean daily water temperatures) were not consistent between sites i.e. sites ranked differently in model performance depending on the temperature metric modelled. Even paired sites within the same thermal group exhibited different levels of modelling success, indicating high levels of thermal variability even within groups.

In spite of good model performance at a quantitative level, qualitative differences were not accounted for unless time series of observed and predicted data were plotted. For example, simulations at an upland Western Cape site showed that simulations were better for summer months than for winter months for mean daily water temperatures, which may have been due to influences of winter rainfall and groundwater inputs not accounted for in the linear regression models (Figure 3.11). This site was located in the hot Cederberg, but has good flows throughout the year. Qualitative assessments are useful in showing up strengths and weakness in model performance in the simulation components of the time series (magnitude, frequency, etc.).

For the process-based model SSTEMP, simulations of daily water temperatures (mean, minimum and maximum) were generally good and comparable at the three selected sites on the Great Fish River (Table 3.9). Simulated time series for daily maximum water temperatures were weaker than those for mean and minimum daily temperatures. In all instances, simulated time series were an overestimate of actual temperatures, most typically during the winter months (Figure 3.12). Simulations based on inputs of observed mean daily water temperatures were more accurate than the same simulations but using simulated mean daily water temperatures as incoming temperatures. Sensitivity analyses showed that the two input parameters which had the greatest influence on model output were mean daily air temperatures and inflowing mean daily water temperatures. For two of the sites (Q9Fish-Carli and Q9Fish-Trum) varying mean daily air temperatures by 10% impacted on output water temperatures by ca. 1°C, while varying mean daily incoming water temperatures affected output water temperatures by ca. 0.5°C. The converse of this was true at Q9Fish-GH012, i.e. input water temperatures affected outflow water temperatures by ca. 1.5°C while input air temperatures affected outflow water temperatures by ca. 0.5°C.

**Table 3.8 Data on mean daily temperature model performance of statistical models**

| Group | Zone | Site | *n* | RMSE | % mean diff ±SD | WSTAT | $R^2$ |
|---|---|---|---|---|---|---|---|
| 1 | Lowland | L8Bav-Res1 | 564 | 2.07 | -3.57±10.79 | -0.59 | 0.67 |
| 4 | Upland | T4Rse3-Channel | 585 | 3.10 | 22.68±17.47 | 2.56 | 0.70 |
| 7 | Lowland | P4Blou-Res | 545 | 2.31 | 6.47±16.51 | 0.60 | 0.71 |
| 5 | Upland | E1Dwar-Kraka | 730 | 2.58 | 13.54±10.92 | 2.04 | 0.74 |
| 3 | Upland | Q4Tark-Oak | 545 | 2.42 | 15.00±27.98 | 1.14 | 0.78 |
| 11 | Upland | N1Gats-Aasvoel | 604 | 2.19 | -0.23±14.86 | -0.45 | 0.79 |
| 2 | Upland | Q9Balf-Wfall | 545 | 1.98 | 4.18±14.38 | 0.30 | 0.80 |
| 9 | Upland | K3Keur-Monta | 411 | 1.58 | 1.36±10.37 | -0.01 | 0.80 |
| 13 | Lowland | Q8Kvis-Cookh | 546 | 2.21 | 4.70±15.66 | 0.29 | 0.80 |
| 10 | Upland | H1Witr-Monum | 730 | 3.02 | 15.74±21.42 | 1.43 | 0.81 |
| 13 | Lowland | H4Bree-Lecha | 730 | 2.90 | 12.04±17.79 | 1.47 | 0.81 |
| 1 | Lowland | L8Bav-Res2 | 564 | 1.36 | -1.21±7.72 | -0.26 | 0.82 |
| 11 | Upland | K6Keur-Outen2 | 606 | 2.59 | -4.87±13.11 | -1.36 | 0.82 |
| 12 | Lowland | Q9Fish-Trum | 598 | 3.08 | 14.34±13.42 | 2.45 | 0.82 |
| 5 | Lowland | K4Dwars-Channel | 410 | 2.35 | 13.99±10.37 | 1.97 | 0.84 |
| 7 | Upland | E1Rate-Beave | 730 | 1.84 | 6.64±11.63 | 0.77 | 0.84 |
| 10 | Upland | G2Eers-Jonke | 586 | 2.88 | 20.24±15.79 | 2.37 | 0.84 |
| 12 | Upland | M1Eland-Cyph | 377 | 2.60 | -4.86±12.09 | -1.38 | 0.84 |
| 9 | Upland | W7-4 | 730 | 2.32 | 14.90±9.39 | 2.04 | 0.89 |
| 3 | Upland | W7-1Drakenstein | 730 | 2.73 | 16.44±18.93 | 1.77 | 0.90 |
| Mean±SD | | | | 2.41±0.49 | 8.38±14.53 | 0.86±1.25 | 0.80±0.06 |

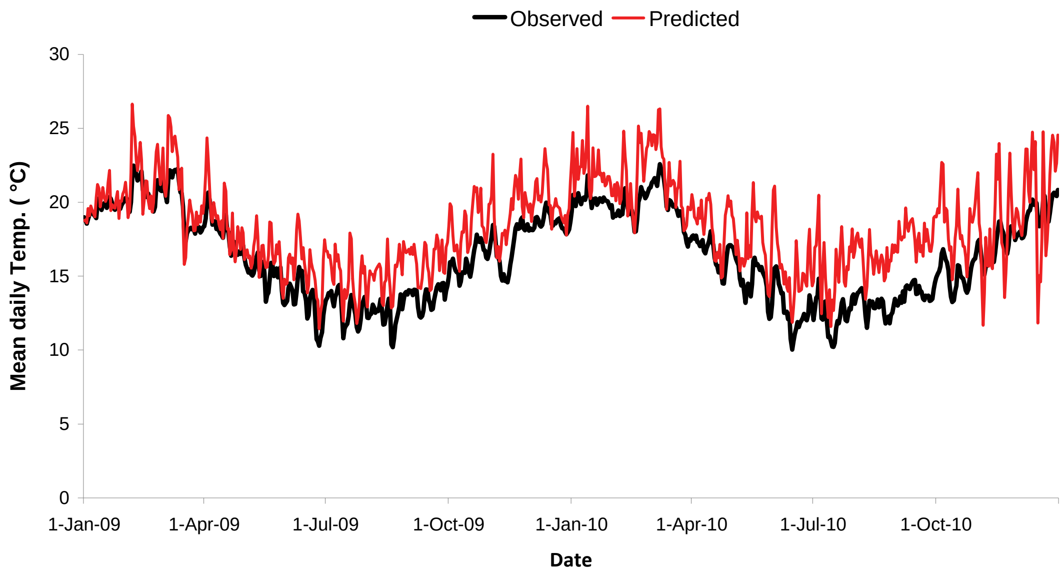


**Figure 3.11 Time series plot of observed and simulated mean daily water temperatures for E1Dwar-Kraka site in the Western Cape**

**Table 3.9 Accuracy metrics for daily mean, minimum and maximum water temperatures at three sites on the Great Fish River. Simulations were based on inputs of mean daily water temperatures using observed and simulated* time series**

| Metric | Site | RMSE | % mean | % s.d. | WSTAT | $R^2$ |
|---|---|---|---|---|---|---|
| Mean | Q9Fish-Carli | 5.26 | 28.20 | 17.92 | 4.89 | 0.83 |
| | Q9Fish-Carli* | 5.42 | 29.04 | 19.05 | 5.01 | 0.80 |
| | Q9Fish-GH012 | 3.80 | 19.32 | 14.60 | 3.33 | 0.85 |
| | Q9Fish-GH012* | 6.61 | 35.32 | 22.50 | 6.19 | 0.81 |
| | Q9Fish-Trum | 5.81 | 29.27 | 13.29 | 5.59 | 0.83 |
| | Q9Fish-Trum* | 5.86 | 29.54 | 13.58 | 5.63 | 0.83 |
| Min | Q9Fish-Carli | 3.06 | 16.35 | 15.82 | 2.46 | 0.83 |
| | Q9Fish-Carli* | 3.27 | 17.39 | 17.42 | 2.60 | 0.80 |
| | Q9Fish-GH012 | 1.83 | 6.84 | 11.43 | 0.88 | 0.89 |
| | Q9Fish-GH012* | 4.49 | 25.11 | 21.42 | 3.90 | 0.78 |
| | Q9Fish-Trum | 3.67 | 19.11 | 13.21 | 3.26 | 0.81 |
| | Q9Fish-Trum* | 3.74 | 19.43 | 13.64 | 3.30 | 0.80 |
| Max | Q9Fish-Carli | 7.63 | 37.81 | 21.52 | 7.19 | 0.69 |
| | Q9Fish-Carli* | 7.76 | 38.50 | 22.32 | 7.29 | 0.67 |
| | Q9Fish-GH012 | 6.08 | 29.12 | 18.52 | 5.57 | 0.75 |
| | Q9Fish-GH012* | 8.74 | 43.19 | 25.12 | 8.29 | 0.72 |
| | Q9Fish-Trum | 8.09 | 37.89 | 15.56 | 7.84 | 0.72 |
| | Q9Fish-Trum* | 8.13 | 38.11 | 15.71 | 7.88 | 0.72 |

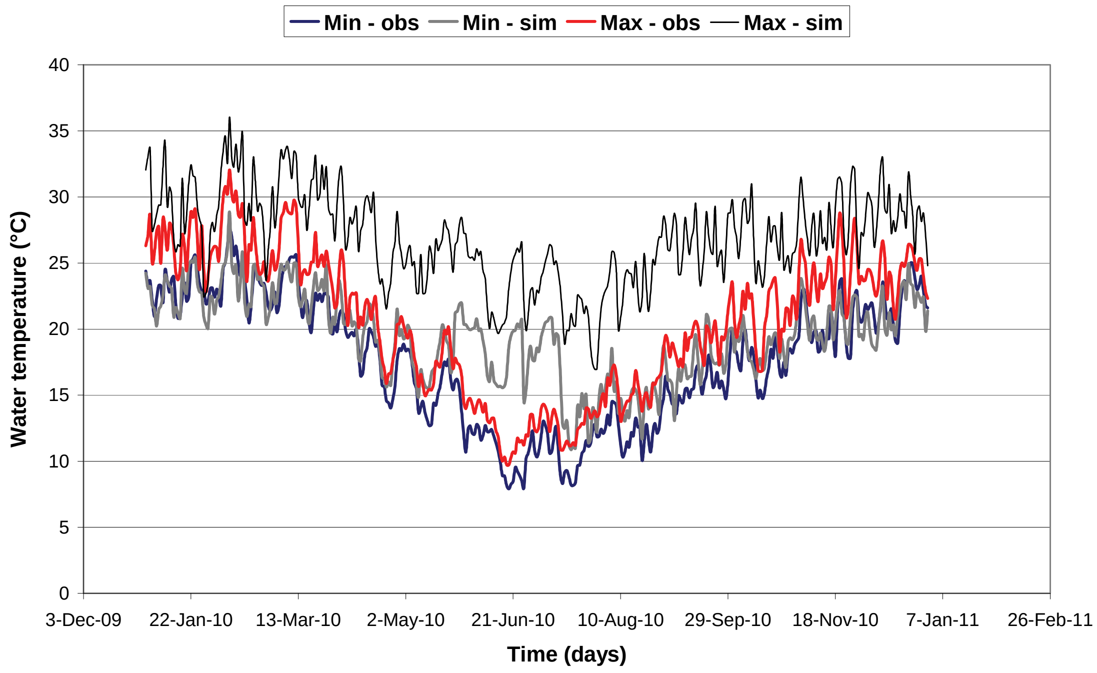


**Figure 3.12 Time series of observed and simulated daily water temperatures (minimum and maximum) at Q9Fish-GH012**

## Conclusions and Recommendations

### *Modelling and management*

The absolute accuracy of models should not cloud the issue of whether a model improves a management decision or not. For example, model assessments based on 2009 water temperature time series showed that simulations of daily maxima were better than daily minima. Conversely, model verifications using different sites and 2010 data showed that simulations of daily minima were generally better than those of daily maxima, but that simulations for different metrics (minimum, maximum and mean daily water temperatures) were not consistent between sites. What is important is that "models are useful tools in resource management. They allow managers to analyze the effects of management decisions based on objective criteria applied to the physical and biological systems influenced by management practices." (Sullivan et al. 1990 p. 63). The modelling context ultimately becomes a process of testing temperature models and data collection, leading to a greater understanding of management actions by those who have the responsibility for managing rivers and setting water quality criteria related to temperature (Webb et al. 2008).

To date, water temperatures in South Africa have been simulated incidentally while focusing on evaporative heat fluxes. Such models (Everson 1999; Savage 2010) have estimated energy fluxes relying on the Penman-Monteith equation to estimate evaporation from a water body. For these models, as with the ones assessed in the study, the weakness with any physically based model is that it requires input temperatures. Indeed, sensitivity analyses showed that SSTEMP was sensitive to the input water temperature value, depending on stream size (Sullivan et al. 1990). Appendix 3 includes guidelines for screening input data, and for choosing and populating suitable models to simulate water temperature time series.

From a pragmatic point of view, the most efficient approach would be to simulate input water temperatures (mean daily values) using a simple linear regression model using air temperatures. This research, in agreement with other research (Benyahya 2009) indicates that the absolute distance of an air temperature station for the water temperature site is not critical to the modelling process. The linear regression approach is not, however, the “silver bullet” to temperature modelling, due to problems with strength and sensitivity of statistical water-air temperature relationship – streams affected by human impacts generally have weaker statistical correlations or lower regression slopes, and a departure from linearity in the air: water temperature relationship typically occurs at high and low temperatures (Webb et al. 2008 and citing others – p 905).

In agreement with the findings of Sullivan et al. (1990) (similar WSTAT values), the SSTEMP model predicted mean daily temperatures well, but performed poorly in predicting maximum, minimum and diurnal fluctuations. This lower accuracy in predicting minimum and diurnal fluctuation by SSTEMP may result because “minimum stream temperatures are more strongly affected by factors such as groundwater temperature and mean air temperature than by solar insolation, which affects maximum temperatures more strongly” (Sullivan et al. 1990, p 110).

Model output is a function of the quality, quantity and accuracy of the data inputs. SSTEMP has been found to be most sensitive to air temperature inputs (Bartholow 1989; Sullivan et al. 1990). A second issue is that of segment length, as this variable relates directly to the concept of equilibrium temperatures, where the model has been found to produce poor results for the maximum water temperature when short reaches were specified (Sullivan et al. 1990, p. 70)

***Recommendations***

For managers assessing different catchment development scenarios and wanting to incorporate water temperatures into ecological Reserve and flow assessments, we recommend the following approach:

- Generate input water temperatures (see also Sullivan et al. 1990; Marcé and Armengol 2008) using simple linear regression models (air water temperature relationship based on equilibrium temperature concept).
- A basin approach of estimating water temperatures is not warranted, and it is preferable to use site-specific simulations. Adding dynamism to the modelling system is not advised: Sullivan et al. (1990) found that basin models were cumbersome to use due to intense data and modelling requirements, and that such systems yielded poor predictions. We recommend that a dynamic basin-level model is too complex for managers to use, and their use is not warranted.
- Use SSTEMP as the current preferred model to estimate mean, minimum and maximum daily water temperatures. While data simulations may not be entirely correct, this model did perform well, and had the added advantage of being able to be used in conjunction with existing hydrological modelling systems used in South Africa (ACRU – Schulze 2007). This model has a full online help system, and has been extensively tested and peer-reviewed.
- While default values can be used for SSTEMP simulations, it is strongly recommended that the model be populated after a site visit. When using default values, it is still recommended that monthly coefficients rather than single values be used, which in the absence of better data can be derived using a sine-function to adjust for seasonality (Marcé and Armengol 2008).
- The statistical distribution, including skewness and kurtosis, of water temperatures over a twenty four hour period has important implications on the decision of whether to simulate mean, minimum or maximum daily water temperatures. As a general rule, where data tend towards a normal distribution, the most accurate approach in simulating water temperatures is to simulate mean daily time series. However, choosing whether to simulate maximum/minimum versus mean daily water temperatures is also dictated by the modelling objectives (Section 3.3.1).

## 3.2 Scale-based patterns of temperatures

### 3.2.1 Micro-scale heterogeneity of water temperature in rivers

**Related publications**

Dallas H.F. and Rivers-Moore N.A. In Press. Micro-scale heterogeneity in water temperature. Water SA.

Dallas H.F. and Rivers-Moore N.A 2010. Water temperatures and the Reserve (WRC Project: K5/1799): Micro-scale heterogeneity of water temperature in rivers. Report Number 1799/18 produced for the Water Research Commission. The Freshwater Consulting Group.

### Introduction and aims

Natural variation in water temperature occurs regionally at the catchment scale; longitudinally down a river system; and at a finer scale due to geomorphic variation, i.e. that of biotope (riffles, pools, backwaters, etc.). Broad scale differences between river catchments are driven by differences in climate, geography, topography and vegetation (Poole et al., 2001). Longitudinal variation often occurs down a river system, with headwaters typically cooler than lowland areas, with maximum temperatures increasing downstream (Ward, 1985; Rivers-Moore et al., 2004); while the maximum range is often found in the middle reaches (Vannote and Sweeney, 1980). At a local scale, channel complexity influences water temperature, with variation occurring laterally across the channel and in relation to side-channels and different biotopes (e.g. wood snags, off-channel biotopes) (Poole et al., 2001). Complex channels with backwaters, shallow margins, deep pools, side channels, etc. have more diverse temperature regimes, whereas simple uniform channels have more homogenous temperature regimes (Poole et al., 2001; Dallas, 2008, 2009). Backwaters may attain higher summer maxima than water of the main channel (e.g. Appleton, 1976; Harrison and Elsworth, 1958; Allanson, 1961) and marginal/lateral areas have been reported to have higher temperatures than mid-channel ones (Clark et al., 1999). Biotopes such as riffles may have a different temperature profile from pools (e.g. Nordlie and Arthur, 1981), which often exhibit vertical stratification in relation to water depth (e.g. Elliot, 2000). For example, Harrison and Elsworth (1958) reported a 10$^{o}$C gradient in a 2 m deep pool on the Berg River, Western Cape, South Africa, while Appleton (1976) observed a 0.9$^{o}$C and 1.9$^{o}$C difference in temperature of 0.70 m and 1.4 m pools in the Gladdespruit, Mpumalanga, South Africa, respectively.

The inflow of groundwater has also been shown to greatly influence water temperature heterogeneity at a site, for example, Mosley (1983) observed a 17.7$^{o}$C range, from 17.2 to 34.9$^{o}$C, in temperature in a single reach of the Ashley River in New Zealand. This was the result of temperature differences in small side channels, which received seepage of cool underflow from the streambed. Variation within a riffle has also been linked to upwelling of groundwater at the tail of riffles compared to downwelling of surface water at the heads of riffles (Evans and Petts, 1997). Cold water patches, which were at least 3$^{o}$C cooler than ambient water temperature, were found to be associated with side-channels, alcoves, lateral seeps and floodplain spring brooks (Ebersole et al., 2003).

Understanding the variation in water temperature between shallow-water biotopes such as riffles and runs, and deep-water biotopes such as pools, will provide insights into potential thermal stress and thermal refugia under changing flow conditions. Riffles and runs have high productivity (Biggs and Hickey, 1994; Biggs et al. 1998), high concentrations of dissolved oxygen, accumulate organic material and generally support a high diversity and density of invertebrates. These biotopes may however be more susceptible to elevated water temperatures, as they are shallower and may be only 0.1 m deep during the low flow period. Pools, which are typically areas of deposition and settling out of suspended sediment, may provide refuge areas for more mobile organisms during periods of high water temperatures, and may become isolated pockets of water during low flows and droughts.

The extent to which water temperatures vary amongst biotopes will also have implications for the selection of an appropriate model to simulate water temperatures, which should be governed by the purpose of the research (for example, biological stress or habitat range studies versus meeting of ecological Reserve thresholds). The aim of this component was to examine micro-scale heterogeneity of water temperature in rivers in upper catchments based on hourly water temperature data. The biological consequences of micro-scale heterogeneity and the implications for water temperature modelling are discussed.

## Methods

Water temperature loggers, programmed to record hourly temperatures, were installed at upland sites on six rivers (Boesmans, Eerste, Duiwelsbos, Groot, Molenaars and Wit) in the Western Cape. These streams were chosen because of their diverse hydraulic biotopes; and where possible, loggers were positioned in three biotopes, namely riffle, run and pool. Daily mean, minimum, maximum (and their standard deviations) and degree-days (based on mean and maximum daily temperatures) were calculated from hourly water temperature data. Temperature metrics, for describing “Indicators of Thermal Alteration” (after Richter et al., 1996, see Table 5.1), were calculated for four sites where a full years data was available. These metrics are divided into six broad groups, the first of which are annual descriptive statistics, while the remainder relate to timing, frequency, duration, magnitude and rate of change (Rivers-Moore et al. 2010b). Sites include Wit river (riffle, run and pool), Eerste (riffle and run), Boesmans (riffle, run and pool) and Duiwelsbos (run and pool). Multivariate analyses were undertaken on the temperature metrics.

## Summary of Major Results

Hourly water temperature data indicate that there is generally a difference in water temperature between the shallow, fast-flowing biotope such as a riffle (average depth of 0.19 m), and the deeper, slow-flowing biotope such as a pool (average depth of 1.2 m). Temperature differences between riffle/run and pool biotopes appear to be related to pool depth, which varied from 0.7 m to 1.8 m, with pools with a depth of less than 0.9 m not exhibiting substantial differences in temperature, while pools greater than 0.9 m had temperature differences up to 4.5$^{o}$C. Greatest differences were apparent during the warmer, summer period and for the daily maximum values (Figure 3.13). Translated to degree-days these differences represented a maximum difference of 197 degrees between riffle and pool biotopes over a 2.5 month period. Greatest variance was observed in the Wit River, followed by the Eerste and Boesmans Rivers, most likely a reflection of greater dependence on surface water compared to groundwater. The greatest differences in daily maximum temperatures amongst shallow- and deep-water biotopes are likely to be observed in rivers that are less groundwater dependent and which have significant pool depth.

## Implications for aquatic organisms

Temperature influences many aspects of an individual organisms' existence including its metabolic (Eriksen, 1964), growth (e.g. Vannote and Sweeney 1980) and feeding rates (Kishi et al., 2005); fecundity (e.g. Brittain, 1991); emergence (McKie et al., 2004); behaviour and ultimately survival. Organisms have an 'optimum thermal regime' (Vannote and Sweeney, 1980) at which optimal growth, reproduction and general fitness occur. Temperatures outside of this range may have a sub-lethal effect on the organism by affecting, for example, its reproductive success; or a lethal effect causing its death. The effect is dependent on the end temperature, as well as the rate of increase and the duration (Dallas and Day, 2004). Short duration exposures, in particular, may be avoided through behavioural adaptation, with organisms utilising cooler biotopes during periods of heat stress, by for example, drifting, migrating to cooler instream areas or burying into the hyperheos.

Several studies have illustrated that fish use thermal refugia (e.g. undercut banks, deep pools) and often thermoregulate by migrating to areas of cooler water when surrounding water temperatures are outside of their preferred range or exceed their upper tolerances (e.g. Torgersen et al., 1999; Elliot, 2000; Ebersole et al., 2001; Gardner et al., 2003. Studies for aquatic macroinvertebrates are relatively scarce with Mundahl (1989) showing that excessively high water temperatures (>39°C) in an isolated stream pool forced freshwater crayfish, *Orconectes rusticus*, to abandon the pool and dig burrows, which were >6°C cooler. It is likely that some aquatic invertebrates that frequent shallow-water biotopes such as riffles may experience thermal stress as water levels drop and water temperature increases, and that the hyporheos, which has been shown to be colder than surface, presents one option for escaping temperatures outside an organism's optimum range.

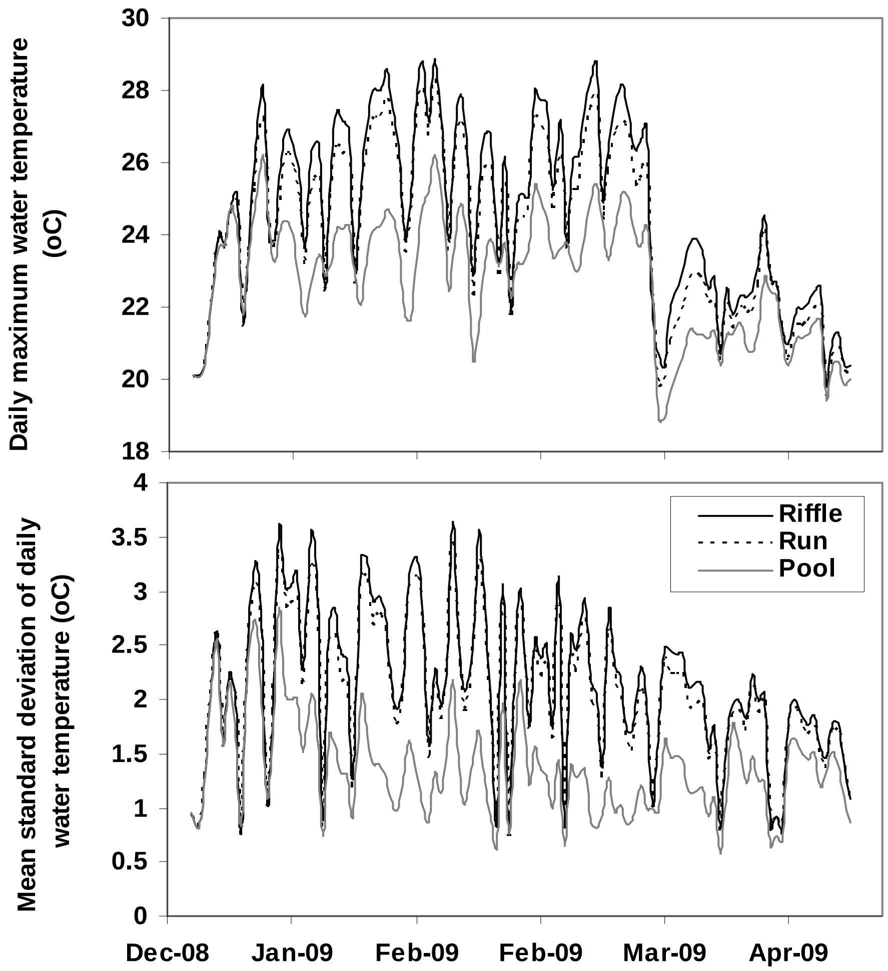


**Figure 3.13 Daily maximum values and mean standard deviation of daily water temperature in three biotopes for Wit River over the warmest period 01 January 2009 to 17 April 2009**

Associated studies (section 4.2) have identified thermally sensitive taxa. These taxa may utilize thermal refugia by moving to deeper biotopes such as pools during periods of thermal stress, although their other habitat requirements related to substrate, food, dissolved oxygen, as well as the risk of predation, may influence their behavioural response. Additional studies on diurnal migration of organisms will provide insight into the behavioural responses. It is also likely that, from an evolutionary perspective, thermally sensitive taxa have developed life cycles that enable them to avoid the hotter period, with development taking place during the cooler months and emergence occurring prior to the onset of temperatures exceeding their thermal tolerance. This is the focus of the life history study (section 4.1.1).

The maintenance of instream and riparian habitat, in the form of, for example, pools, undercut banks, marginal vegetation and an intact hyporheos, will ensure the presence of thermal refugia into which aquatic organisms may temporarily escape if water temperatures exceed their optimum or preferred temperature ranges.

### Implications for modelling

The observed difference in water temperature between shallow- and deep-water biotopes, and surface- versus ground-water dependent stream, highlight the need to consider several aspects before choosing an appropriate model to simulate water temperatures. Key questions to ask include:

- Is the stream a gaining or losing one (i.e. is there recharge from groundwater);
- What are the appropriate temporal and spatial scales for simulation of water temperatures;
- Is the model process-based or statistical?

In answer to the first point, the relative contribution of groundwater to overall streamflow should be assessed either directly (piezometers) or indirectly (water temperature data). Diurnal variation in stream temperatures is reduced in the gaining reaches of a stream as a result of discharging groundwater of relatively constant temperature. This is because groundwater inflows vary diurnally and annually, but with very little diurnal variation in groundwater temperatures and a small annual variation. Therefore variation in groundwater discharge generally increases variation in streamflow but decreases variability in water temperatures. For streams with large groundwater contributions, differences in water temperatures between different hydraulic biotopes are likely to be less, and a model to simulate mean versus maximum daily water temperatures is a moot point. Conversely, for stream with little groundwater input, differences will be more pronounced, and the choice between using a mean versus maximum model becomes more critical. This choice should be dictated by the overall study objectives – is the purpose of the study to relate biotic response to temperatures (in which case a maximum temperature model should be used); to study general trends down a river's axis (mean temperature model) or relative groundwater trends (minimum and maximum models). Additional considerations are 1) the turbidity of the water, since differences in maximum daily temperatures between a pool and a riffle will be more pronounced in a turbid river than in a clear river; and 2) residency time of water in the system, which is a function of stream velocity. The ratio of pool to riffle should inform the choice of model i.e. a model for maximum water temperatures might be more appropriate for a stream with a large amount of riffle.

For the second point, the choice of mean versus maximum temperature model becomes less important as the temporal scale increases. In other words, daily mean and maximum temperatures will be able to be derived from an hourly water temperature model; a daily temperature model should be explicitly mean or maximum, and the choice will be dictated by the larger study objectives; a weekly or monthly model will be less sensitive to whether mean or maximum temperatures are being used.

The third point is partially informed by the choice of time step. Statistical models are less complex and more suited to larger time steps (weekly or monthly temperatures), while process-based models are more suitable for hourly time-step models. At a daily time step, a statistical model will be less complex and can either be used to simulate mean or maximum daily temperatures. A process-based model should have the capability to include water velocity and groundwater contributions, and be able to simulate "average" reach conditions or temperatures specific to hydraulic biotopes. Finally, temperature time series for a pool versus a riffle might have different residuals in the data, ultimately affecting model accuracy. A statistical model might not have the capability to reflect the nuances in the data as accurately as a process-based model. Thus, in choosing an appropriate model to simulate water temperatures, aspects such as groundwater-dependency, temporal and spatial scale, and study objectives should inform the choice of model used.

## Conclusions and Recommendations

Micro-scale heterogeneity in water temperature was observed during this study. Greatest differences were apparent in daily maximum temperatures between shallow- and deep-water biotopes during the warmest period of the year. The depth of the pool biotope affected water temperature differences, with deeper pools creating a more stable and cooler thermal environment. Groundwater-dependency affected water temperature differences with less groundwater-dependent rivers exhibiting greatest differences in daily maximum temperatures. The maintenance of instream and riparian habitat, in the form of, for example, pools, undercut banks, marginal vegetation and an intact hyporheos, will ensure the presence of thermal refugia into which aquatic organisms may temporarily escape if water temperatures exceed their optimum or preferred temperature ranges. In choosing an appropriate model to simulate water temperatures, aspects such as groundwater-dependency, temporal and spatial scale, and study objectives should inform the choice of model used.

### 3.2.2 Water temperature spatial framework

**Related Publications**

Rivers-Moore N.A., Mantel S. and Dallas H.F. 2010d. Water temperatures and the Reserve (WRC Project: K5/1799): Development of a spatial framework for managing water temperature. Report Number 1799/19 produced for the Water Research Commission. The Freshwater Consulting Group and the Institute for Water Research.

Rivers-Moore NA, Dallas HF and Morris C 2011. Water temperatures and the Reserve (WRC Project: K5/1799): Ecological Reserve and Management Guidelines for River Water Temperatures for setting water temperature guidelines for the ecological Reserve in South Africa. Report Number 1799/23 produced for the Water Research Commission. Institute for Water Research and The Freshwater Consulting Group.

### Introduction and aims

Water temperature varies spatially (site, zone and region) and temporally (diel, annually and inter-annually). Understanding the dynamics of water temperatures (the extent of this variation in time and space) reduces the chances of mismanagement of natural systems, through a better knowledge of natural system variability in river systems. There has been a renewed interest recently in statistical analysis to tease out the relationships between water temperature metrics and potential controlling factors operating over sizeable areas (e.g. Nelitz et al. 2007).

A temperature classification offers numerous advantages to conservation and water managers, because it provides the spatial tool through which management programmes can be applied uniformly to broad geographic regions (Hart and Campbell 1994). For example, it becomes possible to predict the natural biota from a modified river by extrapolating from a similar but unmodified river (O'Keeffe *et al*. 1994). A good classification forms a necessary prerequisite to select "representative" river systems, reference sites and monitoring programmes (Hart and Campbell 1994), and forms a necessary basis from which to undertake ecological and condition assessments (Hart and Campbell 1994). While the criteria chosen for the classification are crucial to the utility of it, one classification will not satisfy all potential users (O'Keeffe *et al*. 1994). Classifications should ideally be based on natural rather than modified conditions.

In monitoring networks, data are collected at several locations over time, primarily to detect trends but also spatial pattern (Esterby 1996). Regional estimates are important for environmental variables when effects are expected to be exhibited on a regional basis. Thus more is needed than the estimates at the monitoring locations. The results of many studies involving regional estimates have been given as hand-drawn isopleths or computer-interpolated maps (contour maps) (Esterby 1996).

The aim of this section was to link key water temperature metrics to readily mapped environmental surrogates, to produce spatial images of temperature metrics and thermal regions.

## Methods

### *Source data and data screening*

Hourly water temperature data from a total of 90 sites in the Western and Eastern Cape, representing eight ecoregions (Kleynhans et al. 2005) and seven longitudinal zones (geomorphological classes, Moolman 2008), were processed for correlation with spatial surrogates (Figure 3.14). For the Eastern Cape, one year of hourly data for 2009 at 28 sites were analyzed, and supplemented with two-hourly data from 2008 at 15 sites in the Southern Cape (De Moor, 2009, unpublished data). For the Western Cape, one year of hourly data for 2009 were collected from 47 sites, including 16 sites of half-hourly data from the Table Mountain Aquifer region (City of Cape Town 2009). Sites with data missing for one to two days, or where the logger was known to have been out of water for a period and therefore recording air temperatures, were in-filled using averages of hourly data from one to two days preceding and succeeding the event. Sites were separated into upper and lower river sites based on their longitudinal zone: upper (70 sites) = mountain streams, transitional and upper foothill sites; lower (20 sites) = lower foothills, rejuvenated foothills, and lowland sites.

***Data processing***

***Calculation of temperature metrics***

Hourly data for a full year (January to December) were converted to daily data (average, minimum, maximum and range). The daily data were processed to calculate 37 ecologically meaningful metrics from the temperature time series (See Section 5.1). These temperature metrics define statistics of a river's thermal regime with respect to magnitude of water temperatures, frequency, timing and duration of thermal events (Richter et al. 1996; Olden and Naiman 2010; Rivers-Moore et al. 2010b).

***Environmental variables***

Sixteen environmental surrogates were selected because of published correlations with water temperatures (for example, Vannote and Sweeney 1980; Ward 1985; Sullivan et al. 1990; Rivers-Moore et al. 2008b, c) (Table 3.10). All 37 temperature metrics were correlated with environmental variables, to reduce the number of site characteristics for analyses and highlight environmental variables most important for spatial modelling. Principal Components Analysis (PCA) together with correlation matrices were conducted to determine which variables were important predictors of groupings of sites (Primer 5 for Windows 2002; Clarke and Gorley 2001), using the approach of Rivers-Moore and Goodman (2010). These correlative relationships were further explored using simple linear regression models for selected temperature metrics versus latitude, altitude, percentage downstream distance of river, and stream order. Rose diagrams to explore relationships between aspect and selected temperature metrics were also undertaken (Oriana 2009).

***Spatial modelling of metrics***

As a first step in the spatial modelling process, stepwise forward multiple linear regression models were developed for selected water temperature metrics and five environmental variables (latitude, altitude, ecoregion, geology and stream order), based on their degree of correlation between selected temperature metrics, and the degree to which they could be represented spatially.

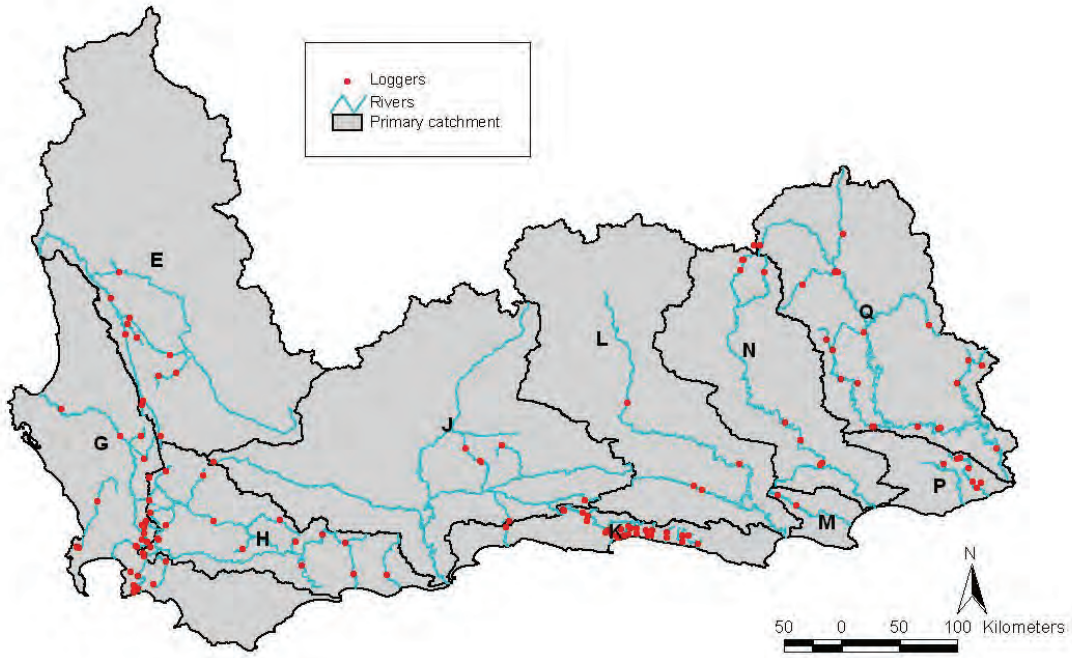


**Figure 3.14 Map of water temperature of loggers used in Western, Southern and Eastern Cape**

**Table 3.10 Environmental variables selected for correlation with temperature metrics for Eastern and Western Cape logger sites**

| Environmental Variable & description | Source |
|---|---|
| Distance from stream source to logger site (1:500 000 scale) | DWAF (2005) |
| Total downstream distance (total downstream distance from stream source to mouth) | DWAF (2005) |
| % downstream distance | DWAF (2005) |
| Aspect, measured as degrees downstream | N/A |
| Latitude | N/A |
| Longitude | N/A |
| Ecoregion | Kleynhans et al. (2005) |
| Longitudinal zone | Moolman (2006, 2008) |
| Stream order (Strahler's stream orders) | DWAF (2005) |
| Altitude (m above mean sea level) | N/A |
| Geology | Vegter (1995) |
| Catchment area ($km^2$) per secondary catchment | Schulze (2007) |
| Catchment perimeter (km) per secondary catchment | Schulze (2007) |
| Secondary catchment | Schulze (2007) |
| River length per secondary catchment (1:500 000 scale) | N/A |
| Drainage Density (river length/catchment area) | N/A |

Next, raster images for the independent environmental variables were derived. A national grid image of altitude was used from Schulze (2007), and stream order per quaternary catchment was calculated by assigning the highest stream order to occur in each quaternary catchment, based on input images of rivers (DWAF 2005; Schulze 2007). A latitude grid was created from a quarter-degree point coverage which was interpolated for latitude. The grid images were manipulated (using regression coefficients and algebraic procedures according to the multiple linear regression models) in a grid-based GIS (Clark Labs. 2009). Image verification opportunities were limited, owing to a paucity of comprehensive water temperature data in South Africa. Temperature metrics from data for 20 unrelated sites previously described by Rivers-Moore et al. (2008c) were correlated with selected spatially modeled temperature metrics.

### *Classification of thermal regimes and spatial mapping of thermal groups*

Cluster analysis was used to define thermal groups (McCune and Mefford 1999) using the group averaging agglomerative option and Euclidean distance as the distance measure. For each group, expected ranges for selected metrics were calculated from the data from the group metrics, which has bearing on the calculation of thresholds.

Groups were related to thermoclines/contour lines describing sites by thermal gradients for particular thermal metrics (mean and coefficient of variation of annual water temperatures). Thermal groups were related to environmental variables using classification and regression tree (CART) analysis (Steinberg and Colla 1997). Environmental variables were chosen based on how effectively they could be spatially mapped and included: location (latitude and longitude), rainfall (mean monthly and annual, annual coefficient of variability, and rainfall season – Schulze 2007), altitude, geology and mean annual air temperature (Schulze 2007). Thermal regions were spatially mapped using an iterative process of multiplying the different environmental grids which had been reclassified into classes based on the CART analyses (Clark labs. 2009).

## Summary of Major Results

### *Mapping of metrics*

Of the 90 sites, downstream distance for 20 sites could not be calculated as they were on tributaries that were not indicated on the 1:500 000 rivers coverage. Thus the analyses were conducted using site characteristics and metrics based on 1 year’s data for 70 sites. A correlation matrix of the 16 site characteristics found that three of them (secondary catchment, catchment perimeter (km) and total river length per secondary catchment) were highly correlated (correlation coefficient >0.8) with other characteristics and could be deleted from further analyses. These parameters were also not significantly correlated with the first two axes of a PCA conducted using all site characteristics, although the remaining variables explained 76% of the variation in the data for the first two axes of the PCA (Table 3.11; Figure 3.15). From the biplot, the environmental variables contributing most to explaining site variance were altitude, followed by stream order, latitude and ecoregion. Stream orders were correlated with downstream distances, and are less intensive to calculate than downstream distances. Correlation matrices between selected environmental and temperature metrics showed that different environmental predictors applied to upper versus lower sites. Furthermore, while catchment area, downstream distance and aspect were not significantly correlated with any of the temperature metrics, the most universal environmental surrogate for temperature metrics was altitude.

**Table 3.11 Cumulative variation of site groupings for PC Axes 1 and 2 based on environmental variables**

| Axis | Eigenvalue | % Total var. | Cum. eigenvalue | Cumulative % |
|---|---|---|---|---|
| 1 | 18.74 | 50.64 | 18.74 | 50.64 |
| 2 | 9.50 | 25.68 | 28.24 | 76.32 |

***Mapping of thermal regions***

Of the five environmental variables used (latitude, altitude, ecoregion, geology and stream order), ecoregion was included in only one of the models. Multiple linear regression models (n=90; $p < 0.001$) for mean annual temperature at three different levels of parsimony (Equation 3.11 – all terms: $R^2 = 0.67$; Equation 3.12 – geology and ecoregions excluded: $R^2 = 0.64$; and Equation 3.13 – only latitude and altitude included: $R^2 = 0.49$ including and excluding stream order respectively), standard deviation of mean daily water temperature ($R^2 = 0.29$; Equation 3.14), coefficient of variation of mean daily water temperature ($R^2 = 0.23$; Equation 3.15) and seven-day (7-D) moving average for minimum ($R^2 = 0.45$; Equation 3.16) and maximum temperature ($R^2 = 0.12$; Equation 3.17) are provided. An example of how each metric could be spatially mapped is provided in Figure 3.16, for mean annual water temperatures. Model verification between modelled mean annual temperatures and observed, independent data showed a good correlation ($R^2 = 0.77$), although the modelled values tended to be over-simulated. Correlation between observed and modelled coefficients of variation showed a weak, negative correlation ($R^2 = 0.18$) which would need to be investigated further.

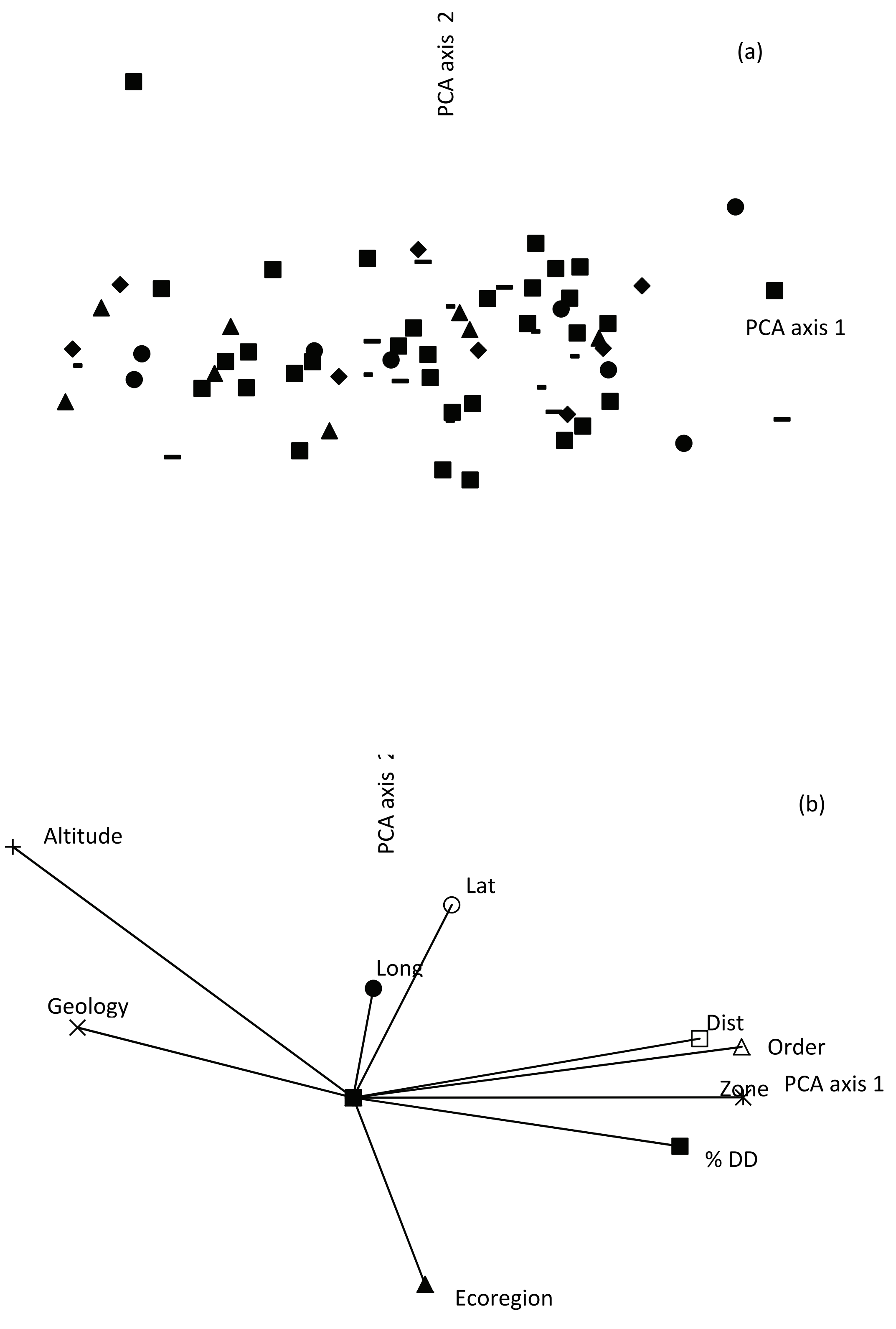


**Figure 3.15 Correlation biplots of sites (a) and main environmental variables characterising sites (b) for the first two axes of the PCA**

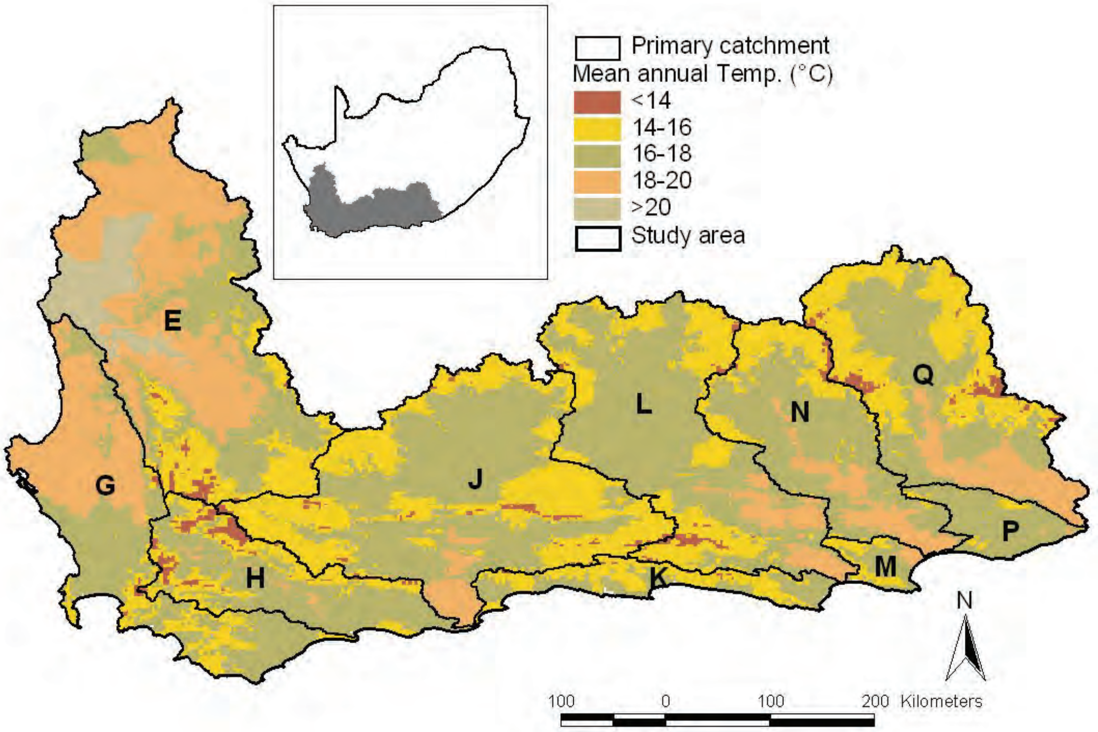


**Figure 3.16 Map of mean annual water temperatures, reclassified into 2°C class intervals, in the Western and Eastern Cape provinces, and spatially modelled using a multiple regression model to estimate mean annual water temperatures from environmental variables (Equation 3.12)**

$$MAT = 54.34 + 1.10(Lat) + 0.57(SO) - 0.003(Alt) - 0.19(Geo) \quad [3.11]$$

$$MAT = 49.05 + 0.98(Lat) + 0.69(SO) - 0.003(Alt) \quad [3.12]$$

$$MAT = 73.90 + 1.67(Lat) - 0.004(Alt) \quad [3.13]$$

$$MAT_{SD} = 16.68 + 0.39(Lat) + 0.16(SO) \quad [3.14]$$

$$MAT_{CV} = 21.28 + 0.007(Alt) \quad [3.15]$$

$$Min_7 = 8.01 + 0.43(SO) - 0.003(Alt) + 0.28(Ecoregion) \quad [3.16]$$

$$Max_7 = 79.52 + 1.62(Lat) \quad [3.17]$$

where *Lat, Alt, Geo, SO* refer to latitude (-ve values), altitude (m amsl), geology and stream order respectively.

***Thermal management regions***

Based on the cluster analyses, 13 thermal groups were derived (Figure 3.17), and two of these groups were excluded because their membership was too small. For each group, thermal ranges were calculated for selected thermal metrics (Table 3.12). Thermal groups could be represented along gradients of selected thermal metrics, including water temperature means (Figure 3.18) and coefficients of variations. These thermal groups correlated well with selected environmental variables (Table 3.13). For the first spatial mapping exercise, based on the CART analyses, the following environmental predictor variables contributed to the groups, with 64 and 38% correctly predicted for learned and tested criteria respectively: rainfall (mean May, June, November and annual), stream order, altitude and longitude (Figure 3.19; Table 3.14). Groups seven and nine could not be predicted using the CART analysis, with the latter group being dominated by groundwater-dependant sites. Of the thirteen initial groups derived from the cluster analysis, nine could be mapped spatially using environmental surrogates. Furthermore, the level of success with which groups could be predicted varied depending on the group (Table 3.14). Using these binary groups, thermal regions were mapped (Figure 3.20).

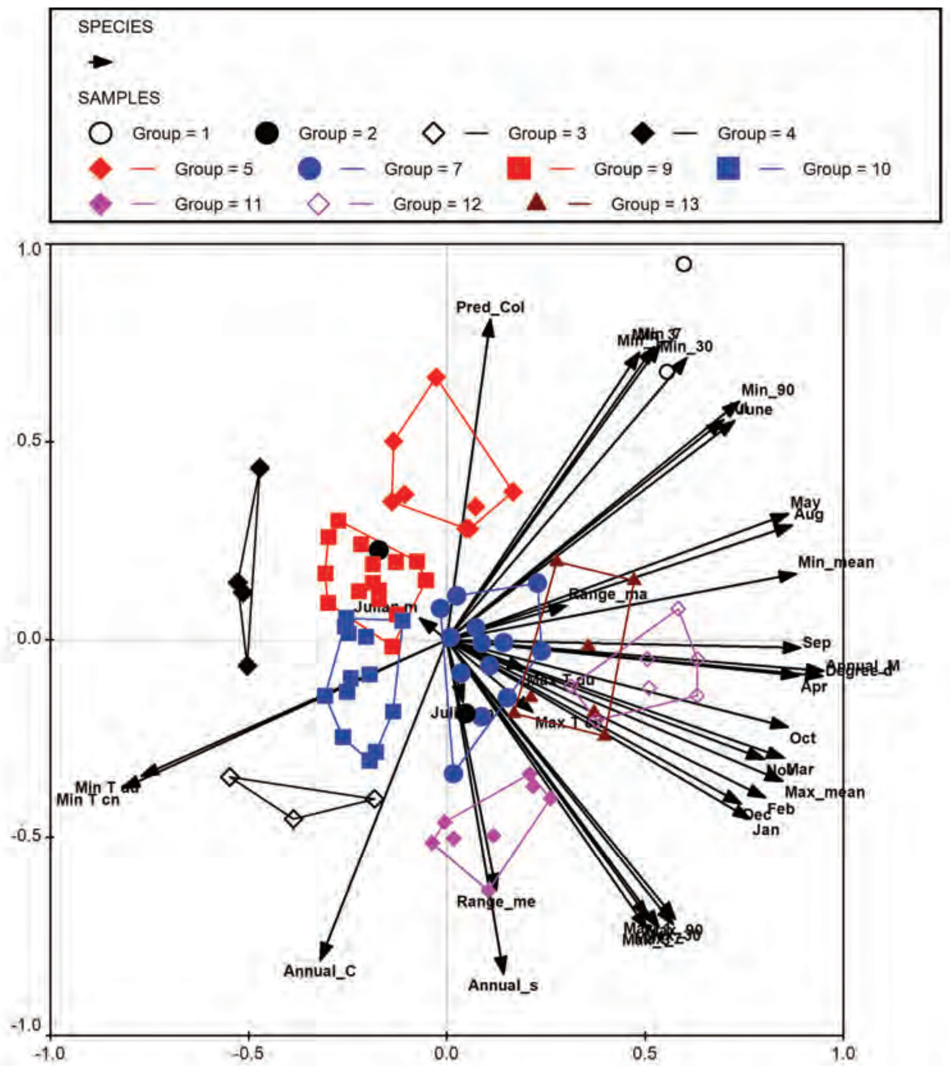


**Figure 3.17 PCA of thermal groups (showing thermal metrics which were best correlated with thermal groups)**

**Table 3.12 Thermal group descriptive statistics**

| Group | Mean | CV (mean) | Predictability | Range (mean) | Total degrees |
|---|---|---|---|---|---|
| 1 | 18.98 | 7.71 | 0.87 | 2.10 | 6828.16 |
| 2 | 15.96 | 23.46 | 0.63 | 1.76 | 5823.84 |
| 3 | 14.47 | 32.94 | 0.60 | 3.25 | 5280.25 |
| 4 | 13.51 | 24.49 | 0.67 | 1.76 | 4929.17 |
| 5 | 15.82 | 17.14 | 0.71 | 1.71 | 5773.68 |
| 6 | 16.51 | 26.45 | 0.76 | 2.66 | 6026.93 |
| 7 | 16.74 | 24.31 | 0.63 | 2.65 | 6108.01 |
| 8 | 15.43 | 30.65 | 0.62 | 3.37 | 5632.23 |
| 9 | 15.04 | 22.22 | 0.66 | 2.09 | 5488.31 |
| 10 | 14.95 | 26.94 | 0.61 | 3.35 | 5456.05 |
| 11 | 16.76 | 26.86 | 0.59 | 6.34 | 6116.29 |
| 12 | 19.39 | 22.94 | 0.63 | 2.75 | 7077.50 |
| 13 | 18.29 | 24.32 | 0.63 | 2.39 | 6677.12 |

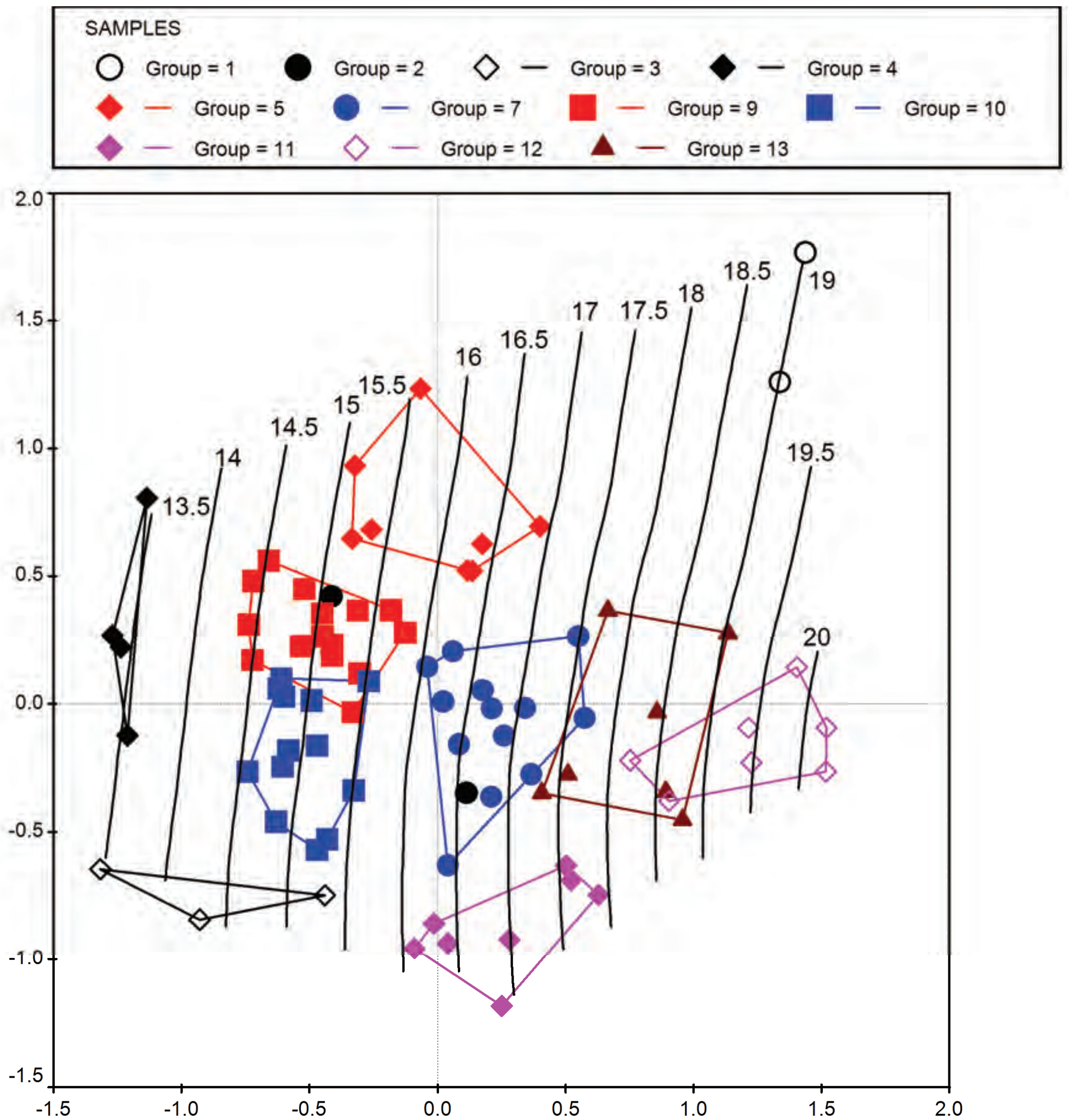


**Figure 3.18 Contour lines (annual mean water temperatures) relative to thermal groups derived from a cluster analysis**

**Table 3.13 PCA for environmental variables lined to thermal groups derived using a cluster analysis**

| | Axis 1 | Axis 2 |
|---|---|---|
| **Eigenvalue** | 0.468 | 0.280 |
| **% Cum var.** | 46.800 | 74.900 |
| Latitude | 0.235 | -0.383 |
| Stream order | 0.657 | -0.167 |
| Altitude | -0.396 | -0.450 |
| Rainfall | -0.006 | 0.133 |

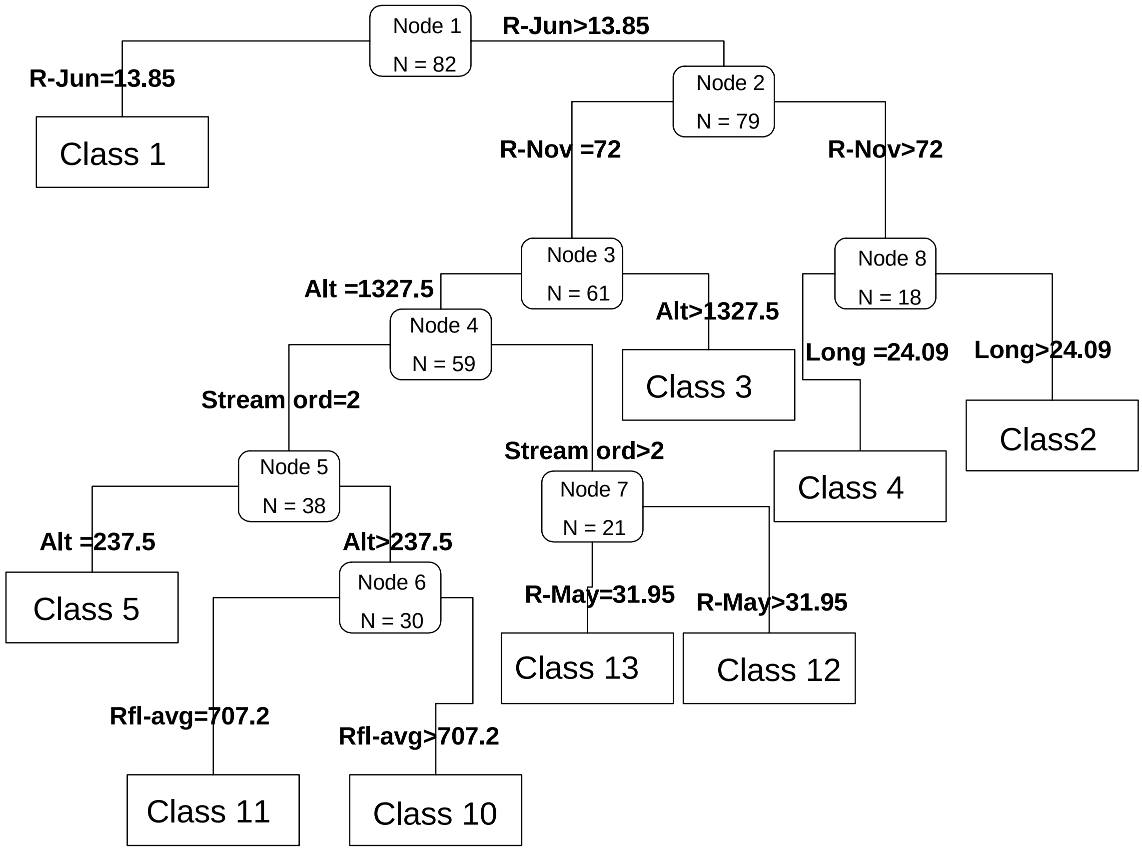


**Figure 3.19 CART classification tree diagram illustrating the relationships respective thermal groups have with environmental variables**

**Table 3.14 Results of classification tree (CART) analyses of thermal groups using geographic and environmental variables**

| Group | *n* | Prediction success (Learn %) | Prediction success (Test %) | Mis-classification (Learn data) | Mis-classification (Test data) |
|---|---|---|---|---|---|
| 1 | 2 | 100 | 100 | 0 | 0 |
| 2 | 2 | 100 | 0 | 0 | 100 |
| 3 | 3 | 66.67 | 33.33 | 33.33 | 66.67 |
| 4 | 4 | 100 | 75 | 0 | 25 |
| 5 | 8 | 62.5 | 50 | 37.5 | 50 |
| 7 | 13 | 0 | 0 | 100 | 100 |
| 9 | 15 | 0 | 0 | 100 | 100 |
| 10 | 13 | 53.85 | 46.15 | 46.15 | 53.85 |
| 11 | 8 | 62.5 | 37.5 | 37.5 | 62.5 |
| 12 | 7 | 100 | 57.14 | 0 | 42.86 |
| 13 | 7 | 57.14 | 14.29 | 42.86 | 85.71 |
| Total | 82 | | | | |
| Average | | 63.88 | 37.58 | | |
| Overall % correct | | 46.34 | 29.27 | | |

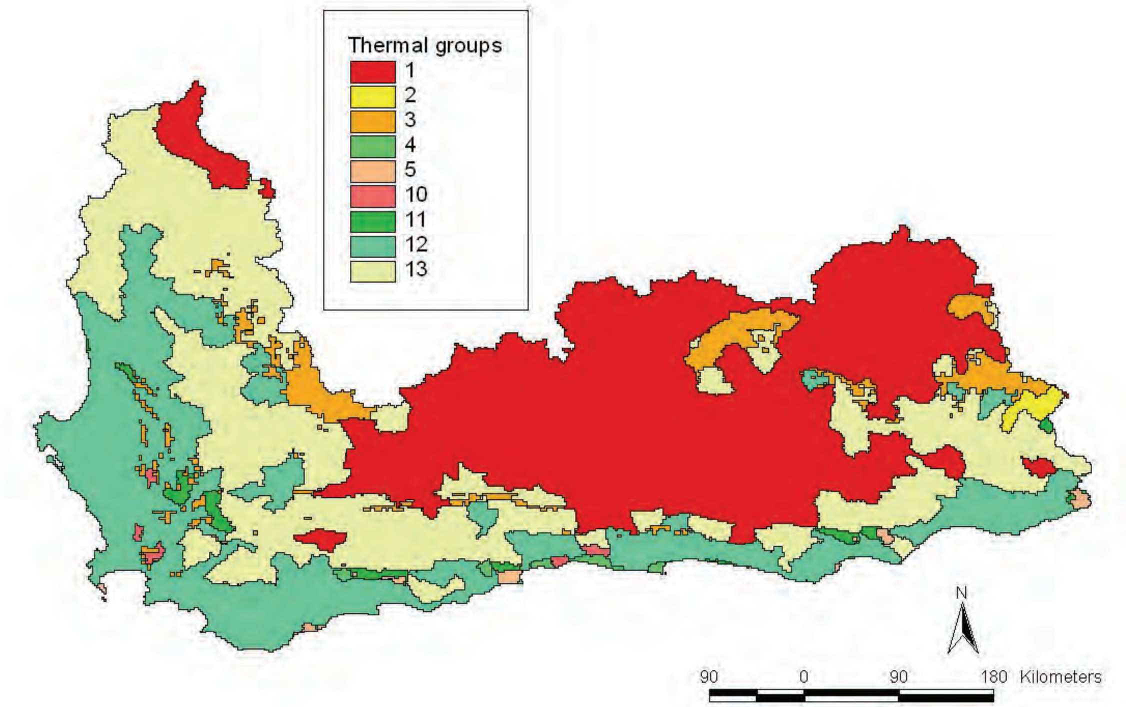


**Figure 3.20 Map of thermal regions in the Western and Eastern Cape provinces based on combinations of environmental variables best predicting membership of thermal groups using CART analysis**

## Conclusions and Recommendations

While many of the temperature metrics were highly correlated, all 37 temperature metrics were used in the analyses, and they could be correlated with a number of environmental surrogates. Correlation with environmental variables which can be spatially represented provides the "Rosetta Stone" to spatially represent water temperature metrics. These analyses showed that data for the period used are adequate in defining initial thermal management regions for the study area.

There are two spatial avenues which could be followed based on this study. The first is to produce a series of images for different temperature metrics, and using the quaternary catchment (or quinaries as a further subdivision) as the basic unit of representation. The second approach is to use thermal regions, to which could be added a relational database for all 37 water temperature metrics, which in time, could indicate baseline reference conditions and associated natural ranges of variability. Whichever approach is adopted retains the utility value of these maps, which will facilitate a regional approach in deriving temperature guidelines as part of the ecological reserve for rivers in South Africa.

For the temporal component, the spatial units become the management regions, and the emphasis becomes one of detecting trend(s) over time, with departure from baseline conditions and potential triggers of management intervention through exceedance of defendable thresholds. Such an approach would require monitoring at selected sites, and annual updates of the temperature metrics from a national monitoring network to identify potential trends (Esterby 1996).

Prediction success at this, the first iteration of the thermal regions concept, was poor (*ca*. 40%). In spite of such current inaccuracies, the utility value of this approach is substantial. Ultimately "reference" thermal graphs for each thermal region (i.e. hypothetical 7-D moving averages of daily means plus daily ranges within a 95% confidence envelope – see Section 5.2) could be incorporated into this approach, and which provides a relatively simple means of detecting whether a site meets its thermal ecological Reserve. It is recommended that such a tool be developed in tandem with improving the accuracy of the thermal regions. Various avenues for research are available, which include, *inter alia*, using species distribution patterns and thermal isoclines similar to the concept of Stuckenberg's (1969) effective temperature maps; expanding the spatial distribution of thermal metric analyses to include more sub-tropical sites on other regions of South Africa (see for example Rivers-Moore et al. 2004); and to refine the robustness of the thermal graphs by paired site studies viz. above an impoundment (reference site) versus below and impoundment (modified site).

# 4 BIOTIC RESPONSES

The importance of water temperatures to aquatic biota has been well documented (e.g. Claska and Gilbert, 1998; Eaton and Scheller, 1996; Elliott, 1994; Sullivan et al., 2000; Dallas, 2008) and temperature is a key factor affecting the number and kinds of species in a stream (Vannote and Sweeney 1980). All organisms have a range of temperatures at which optimal growth (adult size), reproduction and general fitness occur. This is often termed the 'optimum thermal regime' (Vannote and Sweeney, 1980). Temperature outside of the 'optimum thermal regime' may affect the metabolism, growth, behaviour, food and feeding habits, reproduction and life histories, geographical distribution and community structure, movements and migrations, and tolerance to parasites, diseases and pollution; of aquatic organisms (Dallas, 2008). Section 4 focuses on community and individual level biotic responses to thermal regimes and changes in water temperature. It investigates the key life history traits and thermal cues of selected aquatic macroinvertebrates; the role of temperature variability in structuring aquatic macroinvertebrate communities; and thermal tolerances of macroinvertebrates.

## 4.1 Community studies

### 4.1.1 Key life history traits and thermal cues of selected aquatic macro-invertebrates

**Related publication**

Ross-Gillespie V. and Dallas H.F. 2011. Water temperatures and the Reserve (WRC Project: K5/1799): Key life-history traits and thermal cues of selected aquatic macroinvertebrates. Report Number 1799/20 produced for the Water Research Commission. Freshwater Research Unit, University of Cape Town and the Freshwater Consulting Group.

#### Introduction and aims

The effects of water temperature on the growth of aquatic insect taxa from North America, parts of Europe, as well as Australia and New Zealand have been documented within the growing body of international literature relating to the subject. Data from the majority of countries in the Southern hemisphere however remain scant. This is particularly true in the case of South Africa, where detailed life-history data (including information regarding egg development, nymphal growth and oviposition) remain to be collected for almost all aquatic taxa. Additionally, high resolution water temperature data for rivers across the country do not exist. The need for relevant long and short term water temperature data is therefore apparent, as Northern hemisphere practices cannot be blindly applied to the management issues and vastly different river systems of South Africa. This parent project aims to provide information regarding the thermal profiles and regimes of a large number of rivers spanning the country from the Western Cape to the southern and Eastern Cape, and aims to relate these temperature data to the ecology of the aquatic biota present in these rivers (including life-history data).

Several environmental factors are known to govern the life-histories of aquatic insects. Amongst others, water temperature, flow, photoperiod, food and water quality are considered to be the most important. Of these factors, water temperature and flow have received the most attention from limnologists around the globe and are regarded as having the greatest influence on the timing and the duration of aquatic insect life-history traits *viz*. voltinism (the number of generations per year for a particular species), degree of synchrony of hatching/growth, precision and rigidity of emergence, as well as life-history flexibility. Research suggests that differences exist between the life-histories of aquatic insects in the Northern and Southern hemispheres, largely as a response to the unpredictability of lotic environmental regimes, which are primarily affected by temperature and flow (Hynes 1970, Hynes and Hynes 1975, Poff and Ward 1989, Poff et al. 2006).

The central aim of this study was to provide information concerning southern African aquatic insects and their life histories in relation to thermal and hydrological regimes. Based on a review of the literature related to this topic, several general opinions relevant to Southern hemisphere river systems were noted:

- Variable climatic conditions are responsible for more variable flow and thermal regimes in the rivers of the Southern hemisphere, specifically in Australia and South Africa.
- As a result of variable environmental conditions, aquatic insects in lower latitudes and in the Southern hemisphere should exhibit faster life-histories, with a higher frequency of taxa producing multiple generations per year.
- With fewer variables or seasonal time constraints acting on aquatic insects in the Southern hemisphere, life-histories should be asynchronous.
- More favourable conditions year round should result in aquatic insects in the Southern hemisphere exhibiting longer emergence and flight periods.
- Additionally greater environmental variability in the Southern hemisphere should favour the growth of species with more flexible life-histories.

Without a sufficient body of supporting data or research currently available from South Africa to fully substantiate these opinions, more specific hypotheses based on these opinions were therefore proposed and investigated at a preliminary level in this project. Specific hypotheses are:

1a) Predictability of river systems (with regard to thermal and hydrological regimes) will influence life-history parameters such as, hatching, breaking of diapause, length of life cycle and voltinism differently.

1b) Rivers with unpredictable (variable from year to year) flow will be dominated by species exhibiting shorter life cycles (with possibility of multivoltinism) and staggered emergence.

2) Individual species responses will differ in predictable versus unpredictable rivers showing phenotypic plasticity in life-history parameters

3) Unpredictable rivers are disruptive to life cycles for various reasons – more extreme spates resulting in scouring, spates result in food reduction, unsuitable growth periods. As such higher numbers of individuals are present in predictable systems owing to more even recruitment and stability of resources and growth periods

4) Species with short generation times are less likely to be influenced by predictability of thermal/hydrological regime

## Methods

Biotic samples were collected monthly from six rivers within the Western Cape for the period April 2009 to April 2010. These rivers represented a range of thermal and hydrological regimes against which to compare the life-histories of target aquatic insect species. Species from the Ephemeroptera (*Lestagella penicillata*), Plecoptera (three species of *Aphanicercella*) and Trichoptera (*Chimarra ambulans*) were sorted from monthly samples. Hardened body parts were measured and their life history cycles assessed in relation to water temperature and flow, including the inference of timing of hatching, recruitment, emergence and flight duration. Laboratory experiments (including the rearing of nymphs through to emergence as well as the hatching/development of eggs of *L. penicillata* under several controlled temperatures) were conducted to provide finer resolution data relating biotic responses to water temperature that could ultimately be used to better interpret the collected field data.

## Summary of Major Results

## Site Selection

The intention in this study was to select rivers/study sites that exhibited differences in flow characteristics (and by proxy thermal characteristics) that would be reflected in the life-history traits of the same species studied in each of the rivers. While water temperature has been considered to have the greatest influence on the timing and duration of life history traits of aquatic insects, high resolution thermal data for the study rivers (and in fact all rivers in South Africa) did not exist prior to site selection procedures. Thus a number of ecologically meaningful variables were calculated from historical daily flow data, and were used as a proxy for water temperature in the site selection procedure. It was hoped that thermal metrics as well as predictability indices (a measure of inter-annual variability) would closely resemble those obtained using flow data – i.e. that the selected sites would show similar differences and/or groupings in terms of both thermal and hydrological characteristics. Sites selected for this study included the Eerste, Elandspad, Molenaars, Rooielskloof, Wit, and Wolwekloof rivers.

**Key findings:**

- Selected sites showed a range in flow characteristics and flow predictability. Predictability indices did not conform to the initial site groupings based on flow and were therefore not the primary factor, but rather one of several hydrological factors accounting for site specific differences.
- Site groupings based on water temperature data obtained in this study reflected the same site groupings as those based on flow data (Figure 4.1). Therefore, in the absence of water temperature data, flow appears to be a suitable proxy for predicting differences in temperature parameters amongst sites.
- Differences in *thermal* characteristics and predictability scores of each of the sites were not as pronounced as those differences observed in the flow data. This is however not unexpected, as the water temperature predictability indices were calculated from only a single year of collected temperature data, whereas the predictability indices for flow were calculated from 20 years flow data. (Generally, in order to derive reliable measures of inter-annual variability, predictability indices require 10 years data in the case of water temperature (Esterby 1996) and a minimum of 20 years for flow data (Taylor et al. 2003).
- Initial indications however, suggest that thermal characteristics are closely correlated to flow characteristics, as expected.

- Several additional years of water temperature data will likely reveal greater differences in the seasonality/predictability of the thermal characteristics and regimes at each of the sites.
- Sites provided a suitable template (in terms representing a range of hydrological *and* thermal regimes) against which to compare differences (both subtle and marked) in the life-histories of target species.

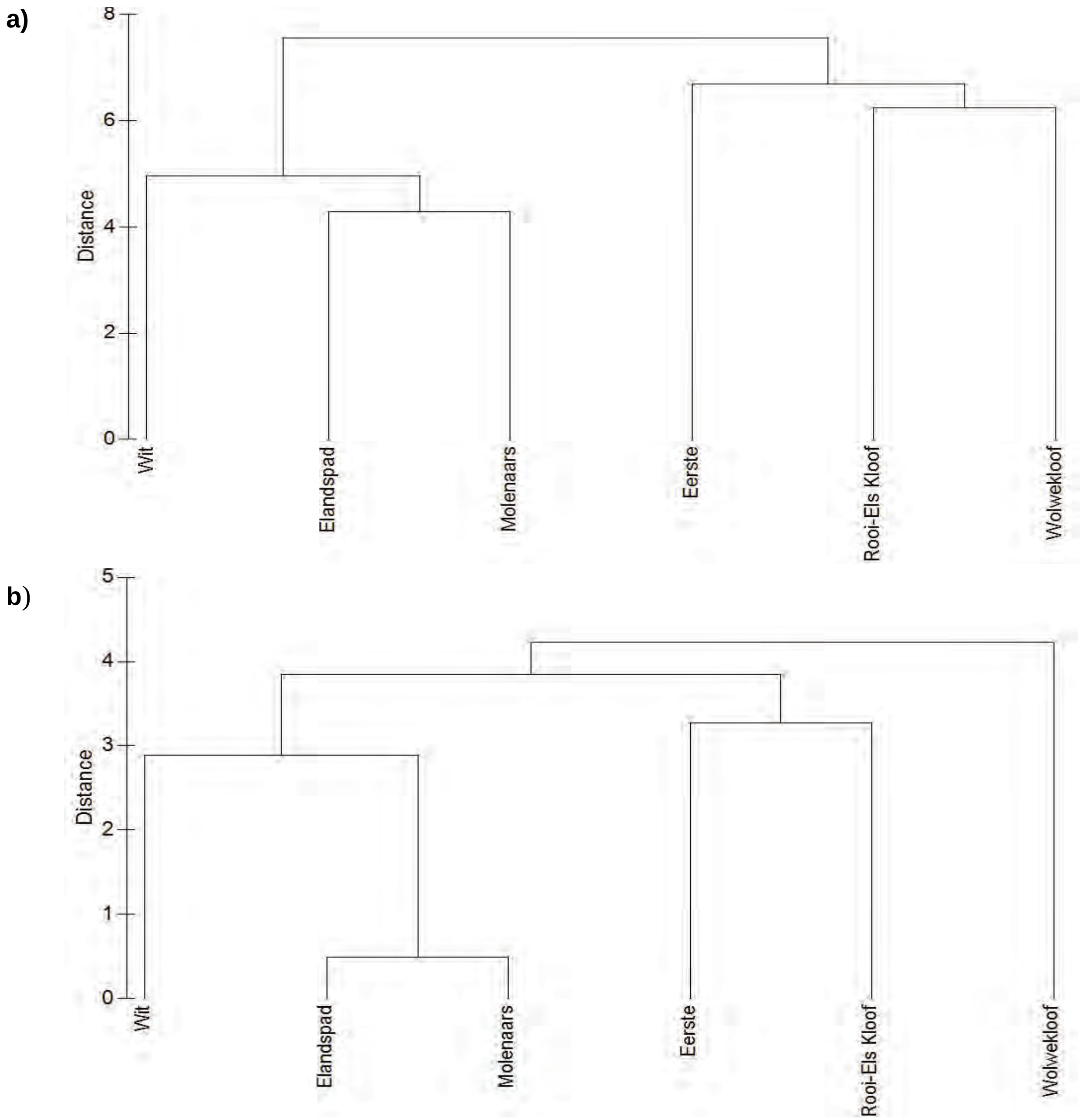


**Figure 4.1 Dendrograms indicating similarity between sites based on a) daily water temperature parameters and b) daily flow parameters. Data were normalised and analysed using the Euclidean distance measure and the group average clustering technique**

While thermal predictability indices were unable to provide meaningful results regarding *inter-annual* thermal variability, the thermal metrics calculated did however offer first insights into *intra-annual* variability in the rivers. Given a larger data set of daily water temperature records (5-10 years) it is expected that thermal predictability indices will closely mirror those predictability indices obtained using flow data.

### Physicochemical variables

Sites selected for this study all occurred in either mountain stream or transitional zones within protected or wilderness areas. This was done in order to as far as possible control for additional factors, such as water quality variables, which may have further confounded the analyses of biotic responses of aquatic invertebrates. Yearly water chemistry data obtained from the Department of Water Affairs (DWA), in addition to the physicochemical variables measured monthly from the six rivers in this study over the duration of the sampling period, were found to:

- be within ranges common to clear mountain fynbos streams of the Western Cape. pH ranged between approximately 4 and 7; dissolved oxygen saturation remained above 85% in all rivers over the duration of the sampling period; and electrical conductivity in all rivers was low, ranging between 8 and 50 µS/cm. Turbidity values were low and ranged between 0 and 2 NTUs in all rivers throughout the sampling period, with the exception of an elevated value of 10 NTUs, which was recorded from the Eerste in June 2009 as a result of ash entering the river from a fire in March 2009. Active channel width measurements of the sites ranged on average from 2 m to 10 m, with a peak in channel width (as great as 40 m) evident in the Wit river during June spates. Channel width measurements largely related to annual flow characteristics.
- generally follow similar trends among sites,
- be largely controlled by seasonal changes in flow and temperature, and
- pose no clear areas for concern regarding possible toxic effects or confounding effects on life-history responses.

Averages of yearly water chemistry variables collected by DWA generally showed similar values across all sites and, in general, none were observed to be at levels that would be either toxic to aquatic insects or result in differential timing of life-history traits. Thus in this study, water temperature and flow were considered to be the primary variables differing at each of the sites and therefore the variables most likely to influence differential life-history responses.

## Flow and Water Temperature

Both flow and water temperature characteristics varied in each of the rivers. Sites exhibited different scores for thermal and hydrological metrics and incurred differing numbers of significant regime shifts (statistically significant shifts from one stable state to another in a long term data series) in both flow and water temperature data.

### Flow

Regime shift analyses using flow data revealed two major groupings of sites: those with more than seven regime shifts (less predictable rivers: Eerste, Wit and Wolwekloof) and those with less than seven shifts (more predictable rivers: Elandspad, Molenaars and Rooielskloof). The numbers of regime shifts correlated negatively with predictability indices calculated from flow data for each of the sites.

Barring the additional regime shifts exhibited in less predictable rivers, the timing of flow regime shifts largely coincided between rivers. This suggested that the same climatic conditions (e.g. cold fronts) were responsible for common shifts in all rivers. Observed differences in the *number* and *magnitude* of shifts could largely be a result of the relative amount of precipitation occurring at each of the sites (linked also to the relative size of the catchments). As only shifts that are significantly different from prior conditions are detected, the same weather phenomenon that results in a regime shift in one study river might not cause a detectable shift in another river. This is likely as a result of site specific conditions such as the contribution of groundwater to summer base flows. Rivers with more stable/constant flows over the dry season as a result of groundwater inputs might not experience a significant regime shift with the onset of the first winter rains. In comparison, a less predictable river that incurs extreme low base flows over summer as a result of limited groundwater input would experience a large magnitude flow regime shift with the onset of the same first winter rains.

The flow regime in general was not observed to be as closely related to the timing of thermal regime shifts in respective study rivers as was initially expected. However in rivers where a greater number of flow regime shifts were evident, occurring later in the season (towards the end of winter and into early spring), the onset of positive thermal regime shifts (or warming of water temperatures) in spring were delayed. Additionally the flow regime shifts that occurred in late spring (November) appeared to be responsible for the untimely negative thermal regime shifts that were recorded in all but one of the rivers (the Rooielskloof) during November.

**Water temperature**

Using accumulated maximum degree days, study rivers were ranked in terms water temperature for the entire period for which temperatures were recorded (23 February 2009 - 25 April 2010). The warmest river, which accumulated the highest number of maximum degree days, was the Wit (7751°C) followed in sequence by the Elandspad (7597.5°C), the Molenaars (7533.1°C ), the Wolwekloof (7441.1°C), the Eerste (7389.0°C) and finally the Rooielskloof with the lowest (6827°C). While results revealed moderate site separation over the entire period, the most distinct separation in thermal rank was observed over the summer period.

Study sites exhibited similar timing, number and duration of thermal regime shifts. In less predictable rivers (in terms of both flow and water temperature metrics), positive spring and summer thermal regime shifts occurred later when compared to rivers experiencing more predictable flow. These less predictable rivers (e.g. Eerste, Wit and Wolwekloof) also showed a greater number of hydrological regime shifts. Unpredictable spates and slightly higher flows appear therefore to delay the onset of warmer spring and summer conditions in these rivers. The high flows that occurred in all of the rivers in November resulted in a negative thermal regime shift in all of the rivers except the Rooielskloof. This river on average was the coldest over the spring and summer period, exhibiting also the most predictable and constant flows (assumed to be the result of substantial colder groundwater inputs). This river in contrast to the others appeared to not be affected by the cooling effect of this late spring cold front.

Thermal shifts were found to be influenced to a degree by flow, but were largely driven by changes in photoperiod. Within the year over which the sampling was conducted, only two major periods of thermal regime shifts were evident. These shifts occurred simultaneously across all rivers and coincided closely (roughly within 2 weeks in both 2009 and 2010) with dates marking the winter and summer equinoxes (20-21 March and 22-23 September, respectively). Stark contrasts in the thermal regime and calculated water temperature metrics were provided by the Rooielskloof and Wit rivers, the coldest and warmest rivers on average respectively.

The colder Rooielskloof and Eerste rivers, when compared to the warmer Wit and Wolwekloof rivers, were observed to provide templates on either end of the gradient of flow and thermal conditions. It was expected that life-histories of the same species would exhibit the greatest differences between these groups of rivers. The Molenaars and Elandspad however showed almost identical thermal and hydrological regimes and for this reason it was expected that life-histories between these two rivers should not show major differences.

### Life Histories

Timing and duration of life-history traits (i.e. emergence, hatching and recruitment) were analysed in relation to the hydrological and thermal regimes. When comparing responses of different insect orders, evolutionary conservatism appeared to override environmental parameters, whereas plastic responses (site specific adaptation) were observed at a finer scale. Life-histories of the basal insect groups (Plecoptera: three species of Aphanicercella, and Ephemeroptera: *Lestagella penicillata*) were constrained by longer univoltine life-histories involving many instars. However the timing and duration of several life-history events was controlled by environmental factors, predominantly water temperature, flow and photoperiod. The trichopteran *Chimarra ambulans* showed the most plastic life-history (from univoltine to bivoltine and trivoltine) in different rivers, in response to thermal regime and water temperature.

For the purposes of this study the seasons referred to within a given year have been defined according to the calculations of metrics developed by Rivers-Moore et al. (2010b) and are as follows: Autumn (1 March - 31 May); Winter (1 June - 31 August); Spring (1 September - 30 November) and Summer (1 December - 28 February).

*Lestagella penicillata*

Life cycle:

- This species showed a slow seasonal univoltine life cycle (see Hynes 1970) (approximately 11-13 months, December/January - October/November/December) in all of the study rivers with a single cohort evident within a given year (Table 4.1, Figure 4.2).
- Noticeable differences in the timing of certain life-history traits (*viz.* emergence, hatching, recruitment, mortality and growth) were observed amongst the rivers studied.

Emergence:

- Emergence period was longer (up to a month) and occurred later (up to a month) in colder rivers (Figure 4.2).

**Table 4.1 The timing of key events in the life cycles of *Lestagella penicillata* from six streams in the Western Cape, South Africa. Asterisks denote estimates based on low numbers of individuals**

| River | Start of Emergence | Peak Emergence | Duration of Emergence period (months) | First presence of next generation nymphs |
|---|---|---|---|---|
| Wit | October * | October* | 1* | December |
| Elandspad | October | October | 2 | December |
| Wolwekloof | October | November | 2 | December |
| Eerste | October | November | 3 | January |
| Molenaars | October | November | 3 | December |
| Rooielskloof | October | December | 3 | January |

- The onset of the emergence period in this species was cued to the warming of water temperatures in the rivers after winter. This warming was linked to the spring thermal regime shift that resulted from day length (photoperiod) increases following the summer equinox (which occurred on approximately 22 September).
- The first adult emergence occurred in October in all but one study river, namely the Wolwekloof, in which this was noted later in November. Data suggest that this is a direct result of the delayed onset of spring warming in this river.
- Emergence period in all rivers extended up to 3 months (October-December)

Hatching:

- Hatching largely coincided with the onset and duration of the emergence period.
- Hatching occurred earlier (December) in warmer rivers (Elandspad, Molenaars, Wit and Wolwekloof) and later (January) in colder rivers (Eerste and Rooielskloof).
- In warmer rivers, it appeared that shorter emergence periods would most likely have resulted in more synchronized oviposition, earlier hatching (up to a month) and thus shorter periods of recruitment.
- Overlap occurred between the new hatchlings and large nymphs of the previous generation, suggesting oviposition without diapauses – this was confirmed through laboratory experiments.
- Optimal water temperature ranges, development time (degree day requirements) in relation to water temperature, thermal limits for egg development, and the degree of hatching synchrony in this species were obtained through laboratory experiments.

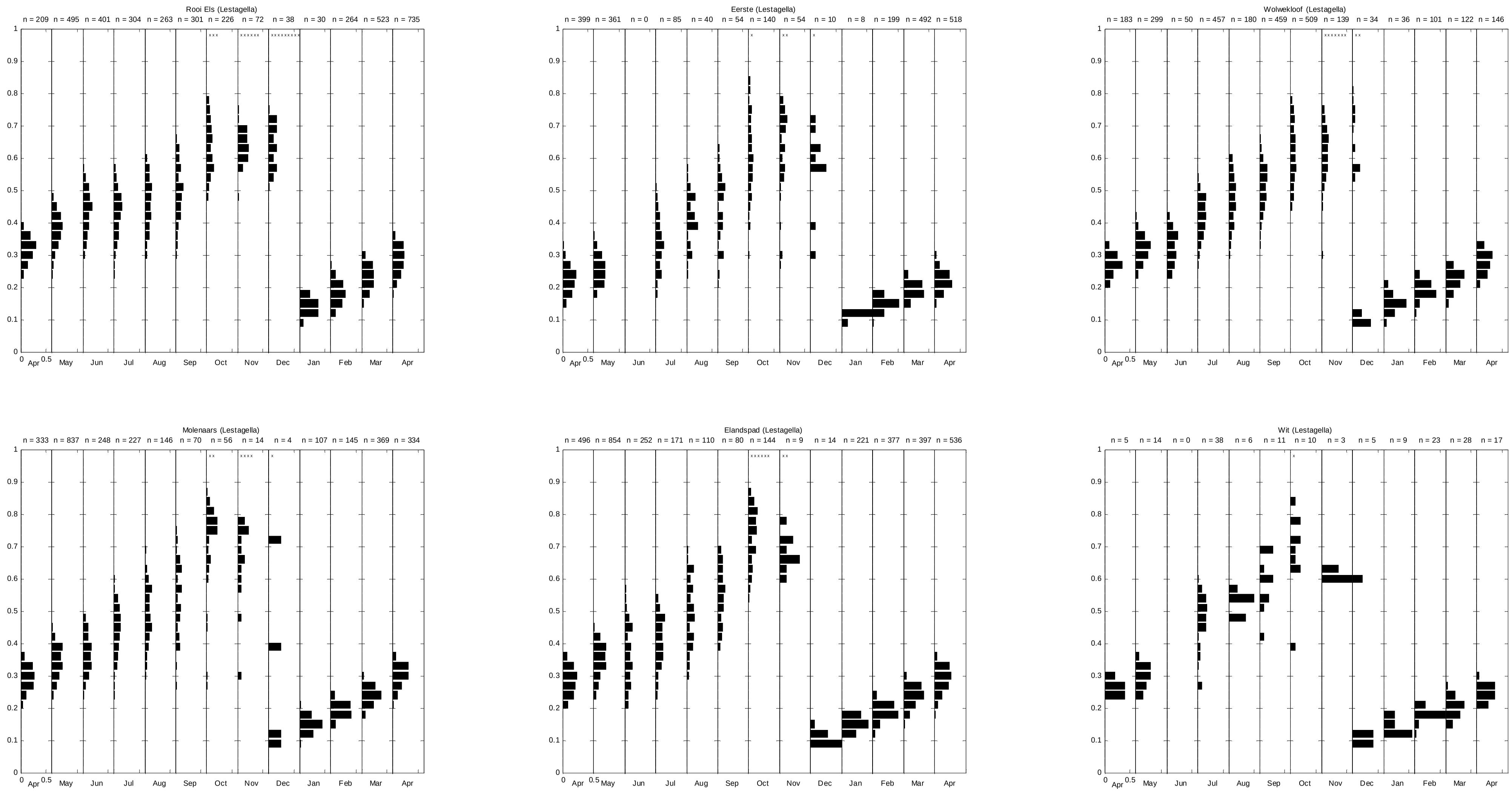


**Figure 4.2 Size frequency histograms of interocular distance measurements (mm) from individuals of Lestagella penicillata collected monthly from six streams in the Western Cape, South Africa for the period April 2009-April 2010. Bar length indicates relative numbers of individuals for a given size class in each month. Asterisks represent the absolute numbers of black wingpad nymphs collected in each month**

Recruitment:

- The period of recruitment largely coincided with the onset and duration of the emergence period (approximately 1-2 months)

Growth:

- Cohort analyses showed that growth was highest on average for warmer rivers in accordance with site groupings based on temperature and flow.
- Warmer rivers yielded larger individuals in samples at the time of emergence.
- Fits of growth rates to logistic functions revealed temperature-dependent growth for this species.
- Growth rates were observed to be lowest for temperatures ranging between 10 and 12°C whereas optimal growth rates were observed for temperatures between 16 and 22°C.
- For the warmer rivers (Elandspad, Molenaars and Wolwekloof), highest growth rates (between 1.4 and 1.6% increase in interocular distance per day) were found to correspond to mean temperatures during the period of hatching (11 December - 13 January) of 20.47, 20.58, 20.75°C.
- In contrast, the growth rates corresponding to the period of hatching (13 January - 16 February) for the colder Eerste (mean temperature 19.5) and Rooielskloof (mean temperature 18.8) rivers were markedly lower at 0.62 and 0.85% respectively.
- The highest mean temperature for the period of hatching (11 December - 13 January) occurred in the Wit River (21.52°C), but the corresponding growth rate was substantially lower than the other warmer rivers (approximately 0.8%) – this could suggest thermal constraints on growth beyond a critical limit.

Mortality:

- Mortality appeared to be predominantly induced by high flow events.
- Reduced hatching success occurred as a result of exposure of developing eggs to thermal critical limits.

*Aphanicercella species*

Three species of stonefly from the genus Aphanicercella (*Aphanicercella scutata, Aphanicercella flabellata and Aphanicercella barnardi*) were collected during the sampling period. The three species were not all present at each of the study sites and in general overall abundances were considerably lower in each month when compared to *Lestagella penicillata*.

Life cycle:

- Slow seasonal univoltine life cycle (see Hynes 1970) (approximately 11-14 months, August/September/October - May/June/July/August/September) were observed in all of the study rivers with a single cohort evident within a given year.
- Noticeable differences occurred in the timing of certain life-history traits (*viz.* emergence, hatching, recruitment, mortality and growth) amongst the rivers studied.

Emergence:

- For the three species emergence periods were less defined and extended for longer periods (3-5 months roughly May - September).
- Estimated peak emergence times appeared to be latest (August) in *A. scutata* in the Wit followed by *A. barnardi* in the Rooielskloof river (June/July).
- Extended emergence periods of up to five and four months were recorded in the Rooielskloof and Wit rivers respectively. In contrast, species occurring in the Wolwekloof and Elandspad and Molenaars rivers had shorter emergence periods spanning 1-2 months each.
- Emergence coincided closely with the onset of higher flows and spates (first regime shifts detected in flow for May) experienced in each of the rivers – possible avoidance strategy of unfavourable conditions.

Hatching:

- First hatching in the Aphanicercella was observed to occur in August and September following a period of between 1 to 3 months since the first and last emergence of adults was noted.
- The development time for Aphanicercella in the six rivers was estimated to take approximately a single month (291-297 Degree days at an average of approximately 9.7°C) in winter from the date of oviposition.
- Hatching largely coincided with the onset and duration of the emergence period.
- Estimated dates of hatching were found to differ according first to species and then to site groupings.

Recruitment:

- Recruitment largely coincided with the onset and duration of the emergence period.

Growth:

- Differential rates of size increase were noted according to species, with *A. flabellata* exhibiting a faster rate of increase in comparison to both *A. scutata* and *A. barnardi,* the latter two species revealing similar rates of increase.
- On a site level, as was observed for *L. penicillata,* warmer rivers showed faster rates of increase.
- Maximum head capsule sizes at emergence were generally similar for each species and across rivers, but sexually dimorphic growth was apparent.
- Sexually dimorphic growth appeared to coincide with onset of a) the first negative thermal regime shift in the year (March) – linked to changes in photoperiod/shorter day length as a result of the winter equinox in the Southern hemisphere b) as a result of increased food supply from autumnal leaf and c) as a result of the onset of winter high flows and spates.
- Growth did not show a clear relation to temperature. Generally growth for all species appeared to exhibit a trend similar to that of a quadratic function.
- Highest growth rates of approximately 2% increase in head capsule width per day were recorded at temperatures of between 14 and 18°C in conjunction with moderate flows.
- High growth rates were exhibited by both newly hatched nymphs towards late winter and early spring (August, September, October) and also larger nymphs prior to emergence (May-June).
- Optimum growth rates (approximately 1%) were otherwise observed to occur at temperatures ranging from 12-19°C, also in conjunction with moderate flows. Above temperatures of ~21°C and at lower flows, lower growth rates were observed.
- This could suggest an upper threshold tolerance of approximately 21-22°C for Aphanicercella similar to the critical thermal maximum and lethal temperatures presented by Dallas (2010a) and Dallas and Ketley (2011) for a closely related species.
- At temperatures below roughly 12°C growth was shown to be generally lower. High magnitude flows and temperatures below 12°C appeared to be suboptimal for growth in Aphanicercella.

Mortality:

- Mortality rates calculated for Aphanicercella were confounded by a prolonged period of recruitment of young instar individuals into the populations in each river.
- Mortality was induced through spates and high flow conditions.

- Numbers of individuals from the three species of Aphanicercella were considerably lower in all streams than those observed for *L. penicillata.*

<u>*Chimarra ambulans*</u>

Life cycle:

- *Chimarra ambulans* showed a non-seasonal or asynchronous cycle, with multiple, overlapping generations occurring within the period of a year.
- Voltinism or number of generations produced in a year by this species was found to vary amongst the study rivers in accordance to site groupings based on flow and temperature.
- Data suggested univoltine, bivoltine and trivoltine life cycles being exhibited by this species.
- Five larval instars were observed for this species.
- Differences in the timing of certain life-history traits (*viz.* emergence, hatching, recruitment, mortality and growth) were observed amongst the rivers studied.
- Total development time and growth rates remain unclear owing to overlapping generations.

Emergence:

- Continuous emergence was observed over spring and summer through to autumn – this can be linked to recruitment period.
- No emergence over winter was observed and the number of larvae declined rapidly in samples with the onset of winter spates and the first flow regimes shifts – response appeared more closely linked to changes in flow as opposed to regime shifts in temperature.
- Rivers experiencing the earliest flow regime shifts in late autumn showed clear signs of larvae disappearing earlier from samples in contrast to rivers showing later shifts.
- Following unfavourable winter conditions, the first adults emerged towards early spring (September-October) in the Molenaars, Elandspad and Rooielskloof.
- In rivers that experienced a later thermal regime shift in spring owing to higher flows over winter (namely the Eerste, Wit and Wolwekloof), first emergence of adults was also noted later (in November, October, and December respectively).
- Emergence of adults after the winter quiescent stage appeared to be thermally cued.

Hatching:

- Hatching largely coincided with the onset and duration of the emergence period.

- In general hatching was observed from late spring to as late as early autumn.
- Owing to the overlap of generations, no egg diapause is suspected in this species.
- The specific onset of hatching in each river however was closely cued to a combination of the abating of winter/spring flows and the onset of warmer water temperatures following winter.
- Hatching was later in the colder Rooielskloof and Eerste rivers and earlier in the warmer Molenaars, Elandspad, Wit and Wolwekloof rivers.

Recruitment:

- Recruitment largely coincided with the onset and duration of the emergence period (long extended period from early spring until autumn).
- It occurred earlier and for longer duration in warmer rivers (Molenaars, Elandspad, Wit and Wolwekloof).

Growth:

- Growth appeared to be temperature dependant.
- Very little larval growth over winter was suspected – larvae were largely absent from samples.
- Rapid growth was observed at warmer water temperatures. However it was not possible to quantify growth rates in response to water temperature, as individual growth cohorts could not be followed through time as a result of overlapping generations.

Mortality:

- Mortality appeared to be induced predominantly by high flow events.
- Colder rivers (Rooielskloof, Eerste, and Wolwekloof) regardless of the magnitude and number flow regime shifts were associated with lower total abundances and fewer generations within the year – slowed growth and development of eggs, fewer adults emerging mating and laying – smaller initial populations.

## Rearing and Egg Experiments

Eggs *of L. penicillata* were collected from the wild for use in egg development experiments at controlled temperatures of 5, 10, 15, 20, 25, and 30° C (Figure 4.3). Experiments showed that hatching did not occur at 5° C, representing a lower thermal limit for development.

Similarly hatching did not occur at 25°C, thus representing a developmental threshold temperature. The developmental temperature range for eggs of this species was therefore between 5-25°C, with an optimum developmental temperature range around 15-20°C. The implications of the developmental temperature threshold of 25°C are considered to be a key factor in the control of population sizes and mortality (as a result of hatching failure) of this species.

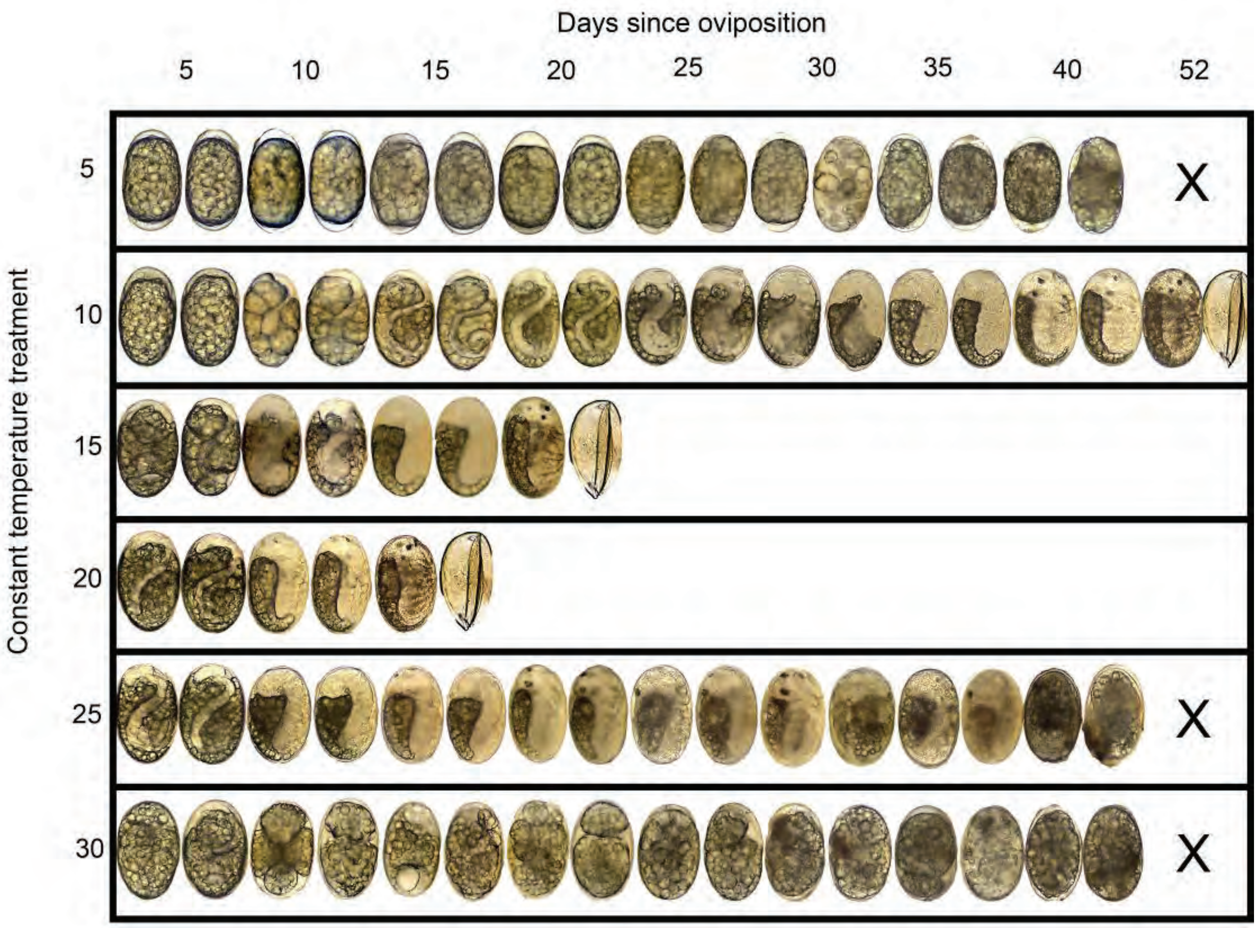


**Figure 4.3 Development of *Lestagella penicillata* eggs from Window Stream, Western Cape, South Africa, incubated at constant temperatures in the laboratory. Eggs used were collected from the wild. Crosses indicate no observed hatching, while hatched eggs represent first observed hatching.**

Experiments revealed that hatching at colder temperatures (15°C and 10°C) took longer from the date of oviposition (approximately 25 days at a constant temp of 15°C and 70 days at a temperature of 10°C) and was less synchronous occurring over a longer period (almost two weeks from the first hatch to the 90% hatch for the 10°C experiment) (Figure 4.4). Similar findings of asynchronous growth at lower temperatures have been reported in many studies of stoneflies and mayflies from the Northern hemisphere.

Brittain (1983) stated that the incubation/development period of eggs of many mayfly species is controlled predominantly by temperature. This was confirmed in the study where a clear relationship between egg development time requirements (and hatching) and water temperature was shown for this species.

Pilot studies involving the rearing of nymphs of Aphanicercella and *L. penicillata* allowed for the interpretation of growth and total development time, including estimates of the number of instars to be made through the monitoring of antennal growth (counts of the number of antennal segments present on individuals prior and post to moulting). The pilot studies have revealed the feasibility of conducting further rearing experiments in the future and allowed for the system design and laboratory setup to be modified and refined for future experiments.

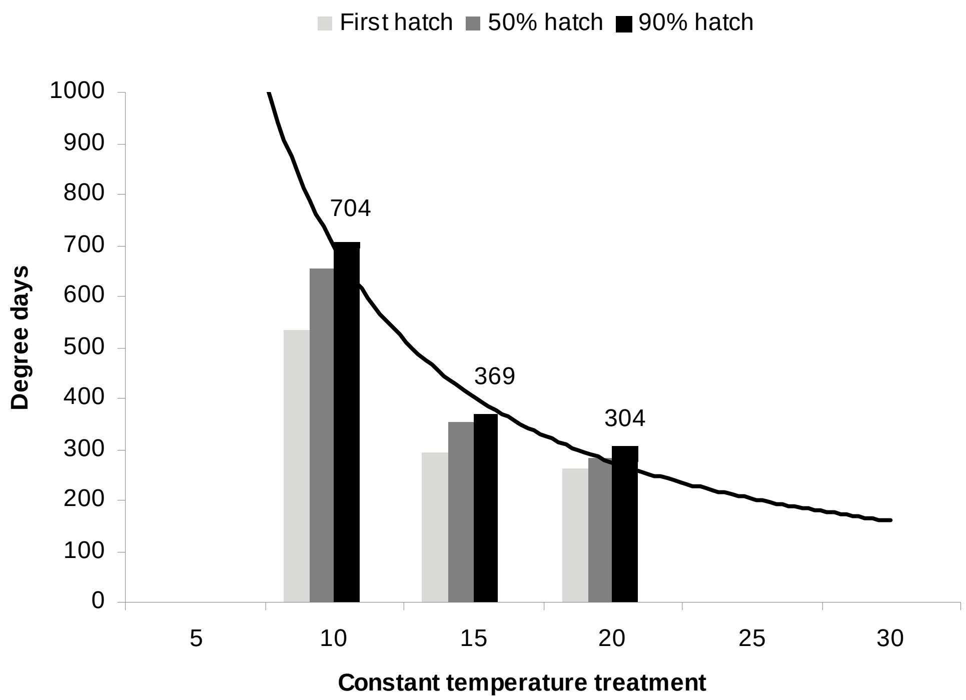


**Figure 4.4 Egg development times versus temperatures in *Lestagella penicillata* from Window Stream, Western Cape, South Africa**

## Conclusions and Recommendations

Life histories of the target species showed differing degrees of flexibility in life history responses – from subtle changes in the timing of emergence and hatching to more extreme differences involving the production of additional generations within a year (changes in voltinism) given differing environmental conditions – primarily in relation to flow and temperature. The following conclusions could be drawn in relation to the proposed research hypotheses:

***1)*** *a) "Predictability of river systems (with regard to thermal and hydrological regimes) will influence life-history parameters such as, hatching, breaking of diapause, length of life cycle and voltinism differently".*

The life-history responses observed for all target species did in fact show differences amongst the rivers in terms of timing of emergence, hatching/recruitment period, growth and mortality. Based on the findings of this study, the primary driving force behind most of these factors, however, was thought to be temperature. Rivers with different predictability values for flow and temperature but similar thermal ranges in water temperature, showed similar life history responses. As such predictability indices of flow and water temperature alone do not account for differential life-history responses. An observable effect of predictability, however, was on the mortality of the species. This is addressed under hypothesis 3.

For *C. ambulans,* univoltinism was observed in the colder rivers (Rooielskloof and Eerste), bivoltinism was recorded in the warmer rivers (Elandspad, Molenaars, Wolwekloof) and trivoltinism in the warmest river, the Wit. This suggested that voltinism in this species was largely linked to temperature as opposed to flow or thermal predictability.

Growth rates (in relation to temperature) of *L. penicillata* exhibited a higher level of fit (i.e. $R^2$ values and significance values) in rivers shown to have greater predictability values. This suggests a) growth rates are less variable in rivers exhibiting higher predictability values and *vice versa* or b) rivers with a lower degree of predictability consequently yielded reduced sample sizes for certain months (owing to high flow) which would in turn affect the fit to a logistic growth curve.

**b)** *"Rivers with unpredictable (variable from year to year) flow will be dominated by species exhibiting shorter life cycles (with possibility of multivoltinism) and staggered emergence".*

As only three target species were analysed for this study, this hypothesis could not be fully addressed. Further, each of target species showed preferences for different temperature ranges and as such their total abundances in each of the rivers are suspected to be largely governed by temperature (in combination with flow). Examples of this were *C. ambulans* occurring in higher numbers in the warmer rivers while the converse is true for *Aphanicercella* which was present in high numbers in the colder rivers.

Of the three target species analysed, *C. ambulans* showed the shortest life cycle and exhibited staggered emergence. Thus, if hypothesis 1(b) were to be true, this species should be most prevalent in the unpredictable rivers. This was indeed observed for the Wit, but it remained uncertain if this was as a result of the low predictability, or of the above-mentioned preference for warm temperatures.

***2)*** *<u>"Individual species responses will differ in predictable versus unpredictable rivers showing phenotypic plasticity in life-history parameters".</u>*

For *L. penicillata* and Aphanicercella there were no observable differences in the life-histories between predictable and unpredictable rivers to suggest a plastic phenotypic response to predictability. This does not discount the possibility that species may possess a plastic reaction norm, which in turn would allow them to grow faster in unfavourable conditions where they may need to do so (Nylin and Gotthard 1998).

Phenotypic plasticity could have been evident in *C. ambulans*, but such postulations would need to be confirmed through genetic and laboratory rearing.

***3)*** *<u>"Unpredictable rivers are disruptive to life cycles for various reasons – more extreme spates resulting in scouring, spates result in food reduction, unsuitable growth periods. As such higher numbers of individuals are present in predictable systems owing to more even recruitment and stability of resources and growth periods".</u>*

This was indeed observed for *L. penicillata.* For the pairs of rivers compared, lower total numbers were observed for unpredictable rivers (Eerste and Wit) as opposed to rivers showing higher predictability (Rooielskloof and Molenaars). The effect of unpredictability on the mortality of the other target species is however confounded, as they are suspected to avoid the winter period, over which unpredictability is manifest in high flows.

***4)*** *<u>"Species with short generation times are less likely to be influenced by predictability of thermal/hydrological regime"</u>*

This seemed evident when comparing the abundances of the longer-lived *L. penicillata* to those of shorter-lived *C. ambulans*, in that *L. penicillata* experienced higher mortality in unpredictable streams. In contrast *C. ambulans* exhibited similar total abundances in both predictable and unpredictable rivers. *Aphanicercella* seemed to also show little differences in terms of abundances between predictable and unpredictable rivers (in fact showed greatest numbers the unpredictable Wit river). This could be explained by the fact that species of *Aphanicercella* avoid unpredictable, high flow winter periods (through timing of emergence).

This study was unable to illustrate a clear link between predictability and life-history responses in the target species. This does not, however, discount the potential for such responses to exist. Since temperature was found to be the primary driving factor of life-history responses, it is difficult to isolate the effect of predictability. In order to fully assess the effect of predictability, a) more species would need to be analysed, and more importantly b) species would need to be compared from rivers exhibiting similar thermal profiles, but differing predictability indices. It remains to be determined what range of predictability values exist within rivers across the country and what would constitute a sufficiently ‘predictable’ or ‘unpredictable’ river with which to contrast life-history responses.

Each species showed varying responses (ranging from subtle differences in timing of traits - on the scale of weeks – to pronounced differences in the number of generations per year) in life-history traits across rivers. Seasonal fluctuations in water temperature, as a result of photoperiod, were found to predominantly govern voltinism, timing and duration of emergence, timing of hatching, length of the recruitment period and synchronization in growth of *L. penicillata*. Flow was observed to have greatest influence on the sample sizes and the inferred mortality estimates of the target species. In addition flow was observed to influence the timing of the spring thermal regime shift, which in turn had an impact on the timing of some life-history traits.

The life-history responses of the target species assessed in this study appear to be unique and finely tuned to the hydrological and thermal regimes of each river studied, possibly as a result of site specific adaptation and evolution. However, where the same species showed marked differences in life-history responses amongst rivers, it remains to be determined whether this is as a result of true genotypic differences or of phenotypic plasticity.

Life-history studies inform all areas of aquatic ecological research, whilst also providing information relevant for conservation, and management of river systems. When life-history studies are conducted in conjunction with laboratory studies (such as rearing or egg development experiments) they provide a wealth of fundamental baseline information, useful to aquatic ecologists. The collation of such data and the subsequent formation of an index would prove invaluable to freshwater research in the country. For this reason life-history studies should be prioritised in the field of Limnology/freshwater aquatic research.

Information regarding water temperature is vital in the interpretation of life-history studies and individual species responses, not only in the face of ongoing anthropogenic impacts on river systems, but also climate change research. Therefore it is recommended that appropriate systems be employed to measure and record long term water temperature trends at all existing DWA gauging stations across the country. Further research into aquatic insect life-histories should be conducted that investigate the life-histories of additional taxa, taking into account a wider range of thermal and hydrological gradients (perhaps even including responses of species in temporary rivers).  Life history studies should employ the correct field sampling frequency and incorporate laboratory studies – obtaining higher resolution data regarding the nymphal biology/growth and or the development of the eggs provides crucial information regarding critical thermal limits for growth and development and for interpretation of field data collected at the lower resolution of the sampling frequency.

The linking of life history to *in situ* thermal data and the development of guidelines is explored further in the section on Management (section 5.1 and 5.2).

### 4.1.2 Role of temperature variability in structuring aquatic macroinvertebrate communities

**Related publications**

Eady B. and Rivers-Moore N.A. 2011. Water temperatures and the Reserve (WRC Project: K5/1799): Role of temperature variability in structuring macroinvertebrate communities – Final Report. Report Number 1799/21 produced for the Water Research Commission. University of KwaZulu-Natal and the Institute for Water Research.

Eady B. 2011. Role of water temperature variability in structuring aquatic macroinvertebrate communities – case study on the Keurbooms and Kowie Rivers, South Africa. Master of Science, University of KwaZulu-Natal. 117pp.

#### Introduction and Aims

South Africa, classified as a semi-arid country, has a mean annual rainfall of 500 mm, where this low mean belies an uneven rainfall distribution, causing precipitation to be unpredictable and variable. Rainfall ranges from below 100 mm to greater than 1200 mm per annum, where the general trend is high in the south-east and low in the north-west of the country. This high variability contributes to the diverse range of aquatic ecosystems in South Africa, where associated species have had to adapt accordingly to either regular or irregular flows on perennial or non-perennial streams respectively. Stream ecosystems are affected and driven by a large number of biotic and abiotic aspects that create biotic patterns through their interactions, resulting in complex systems (Dallas, 2008; 2009; Dollar *et al.*, 2007).

Along with flow, water temperature is a primary abiotic driver affecting the types and quantities of macroinvertebrates in streams, and is a particularly important parameter to monitor within sensitive aquatic environments. Diel temperature ranges impact on the potential diversity of species that can coexist within freshwater ecosystems, due to every individual occurring in the zone of its most optimum temperature during part of the day (Vannote and Sweeney 1980). According to Vannote and Sweeney (1980), the key to sustaining temporal segregation within aquatic invertebrate communities is seasonal stream temperature patterns enabling resource partitioning to occur, thus preventing the competitive exclusions, while species zonation occurs partly due to water temperature differing spatially.

The aim of this research was to determine whether the degree of predictability in a stream's water temperature regime may provide an indication of the degree of structure and functional predictability of macroinvertebrate communities. The objectives of the research were:

- To establish whether aquatic macroinvertebrates typically respond in a predictable manner to changing environmental conditions, temperatures and flows;
- To test whether the temporal partitioning of macroinvertebrate species, such as diversity indices and functional feeding groups, are related to water temperature variability.

## Methods

Five paired sites were sampled along two rivers (Keurbooms and Kowie/Bloukrans Rivers) on a seasonal basis between June 2009 and April 2010 (Figure 4.5). Paired sites on each river system were comparable in their ecoregions, longitudinal zones, stream orders and profile zones, but differed in their thermal variability, based on the hydrological index for their respective quaternary catchments. The upper two sites (K1 and K2) on the perennial Keurbooms River are first order, site K3 is second order and sites K4 and K5 are third order. K1 is at 583 masl (meters above sea level) and the lowest site (K5) at 1 masl. A gauging weir, K6H019, is located approximately one kilometre downstream of K4 site.

Since the source of the Kowie River is situated in the middle of the town of Grahamstown (Figure 4.5), the upper-most site on this system was 'offset' to B1, an equivalent headwater site that was not affected by urban pollution, runoff or other anthropogenic activity that may hinder aquatic macroinvertebrate communities. The upper three sites (B1, B2 and B3) are all first order, where this segment of river is classified by as non-perennial. Sites B4 and B5 are second and third order respectively and are on a perennial river segment. However, due to a drought in this region, these segments were not flowing during summer and autumn. B1 was at 400 masl and the lowest site (B5) at 5 masl. A gauging weir, P4H001, is located approximately 800 meters downstream of the B4 site.

### *Data collection*

Twelve years of flow data were obtained from the Department of Water Affairs. Observed flow data were obtained for each of the gauging weirs present on the streams namely K6H019 (Keurbooms River @ Newlands) and P4H001 (Kowie River @ Bathurst). Simulated flow data for each of the quinary catchments along the rivers were used to generate simulated flow data for the other sites.

Water temperature, flow and select water quality data were collected to simultaneously to macroinvertebrates. Water temperature was recorded using Hobo UTB1-001 TidBit V2 data loggers programmed to record hourly water temperatures. Hourly water temperature data were converted to daily temperature values (mean, minimum and maximum). Conductivity, pH and total dissolved solids were measured using a hand-held meter (Cyberscan 200, with ±0.05% accuracy) and turbidity was determined using a Secchi disc. The depths of the rivers/pools and wetted width were measured.

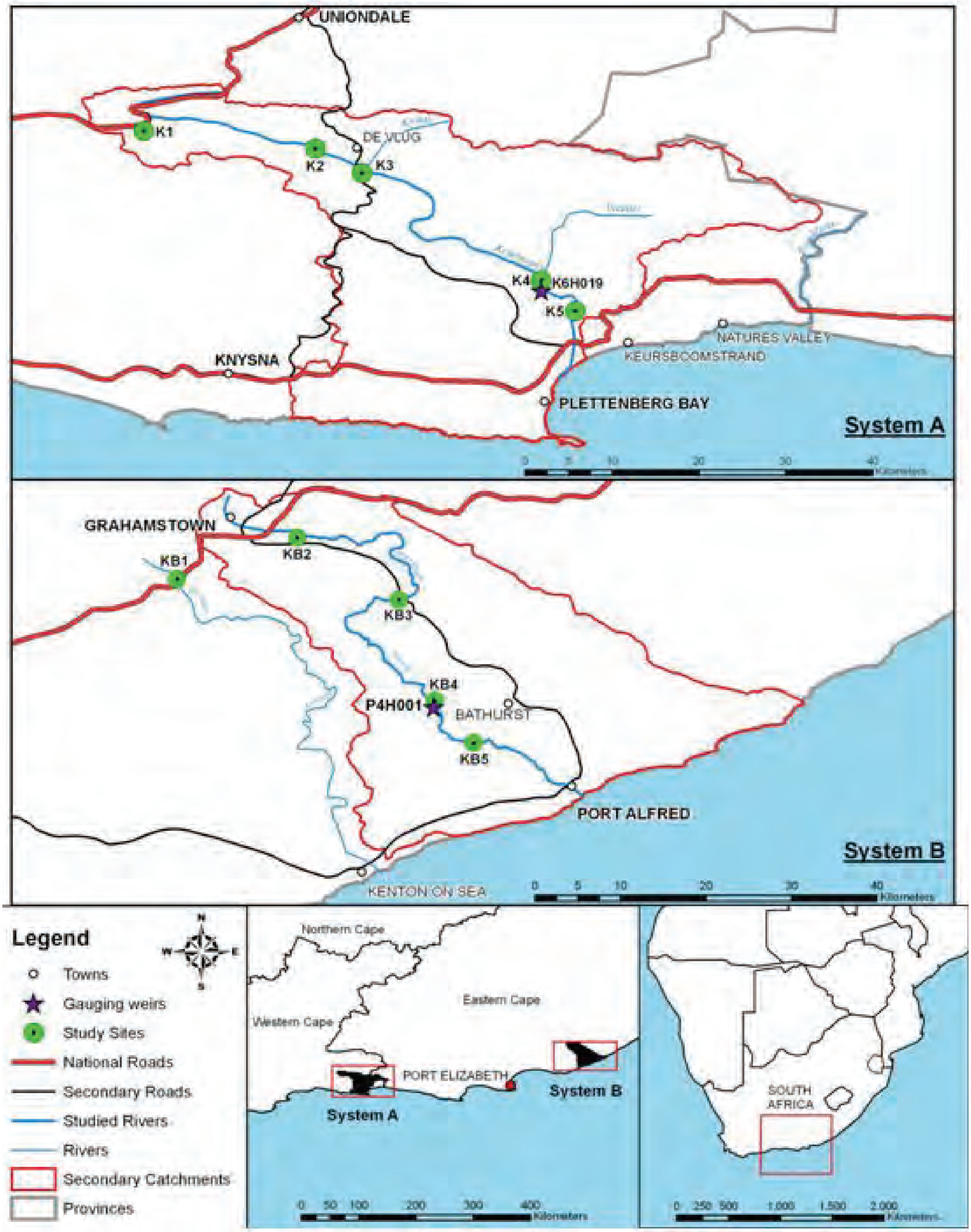


**Figure 4.5 Study area showing the paired sample sites from each river system.**

Macroinvertebrate sampling was undertaken in close proximity to where the water temperature loggers were positioned, to relate water temperature data to aquatic macroinvertebrate data. Only hydraulic biotopes close to the temperature logger were sampled. The depth at which macroinvertebrate sampling occurred varied, depending on the stream discharge. Biotopes sampled included stones-in-current, stones-out-of-current, pools, marginal vegetation and gravel, sand, mud.

***Data analyses***

Daily mean, minimum and maximum water temperature values were used to calculate Indicators of Thermal Alteration (see Table 5.1). The observed and simulated flow data were run through the Indicators of Hydrological Alteration (IHA) software.

Sørensen's similarity index was used to quantify how different macroinvertebrate communities varied seasonally. Macroinvertebrate turnover throughout the seasons was determined by using the coefficient of variability (CV) in seasonal macroinvertebrate diversity. Mean taxon richness values and standard deviation (SD, number of variables = 4 and degrees of freedom = 3) was calculated from all seasons and for each site. The coefficient of variation was determined by equation below (Equation 4.1),

$$CV = (100 \times SD)/X \qquad [4.1]$$

where SD is standard deviation and X is the mean.

Multivariate statistics were performed on the macroinvertebrate taxa and water quality data, including temperature and flow parameters, in order to identify which environmental variables affected certain macroinvertebrate groups. The multivariate analyses used included Principal Component Analyses (PCA), Canonical Correspondence Analysis (CCA), Bray-Curtis and Non-metric multidimensional scaling (NMS) ordination.

## Summary of Major Results

***Flow***

From the observed twelve years of flow data, low flow periods regularly occurred at both weirs, particularly on the Kowie/Bloukrans (Figure 4.6). Flow levels over the macroinvertebrate sampling period (June 2009 - April 2010) were significantly lower in comparison to the flow data for previous years. The abundance of the Keurbooms River peaks (perennial) was more prevalent compared to the Kowie/Bloukrans River peaks (non-perennial), experiencing no flow between January and April 2010. The Keurbooms River had more predictable (0.43) observed flows than the Kowie/Bloukrans River (0.33). Results of simulated flow data indicate similar trends to the observed flow data, with predictability values higher for all the Keurbooms River quinaries than the Kowie/Bloukrans Rivers quinaries, albeit the differences were small.

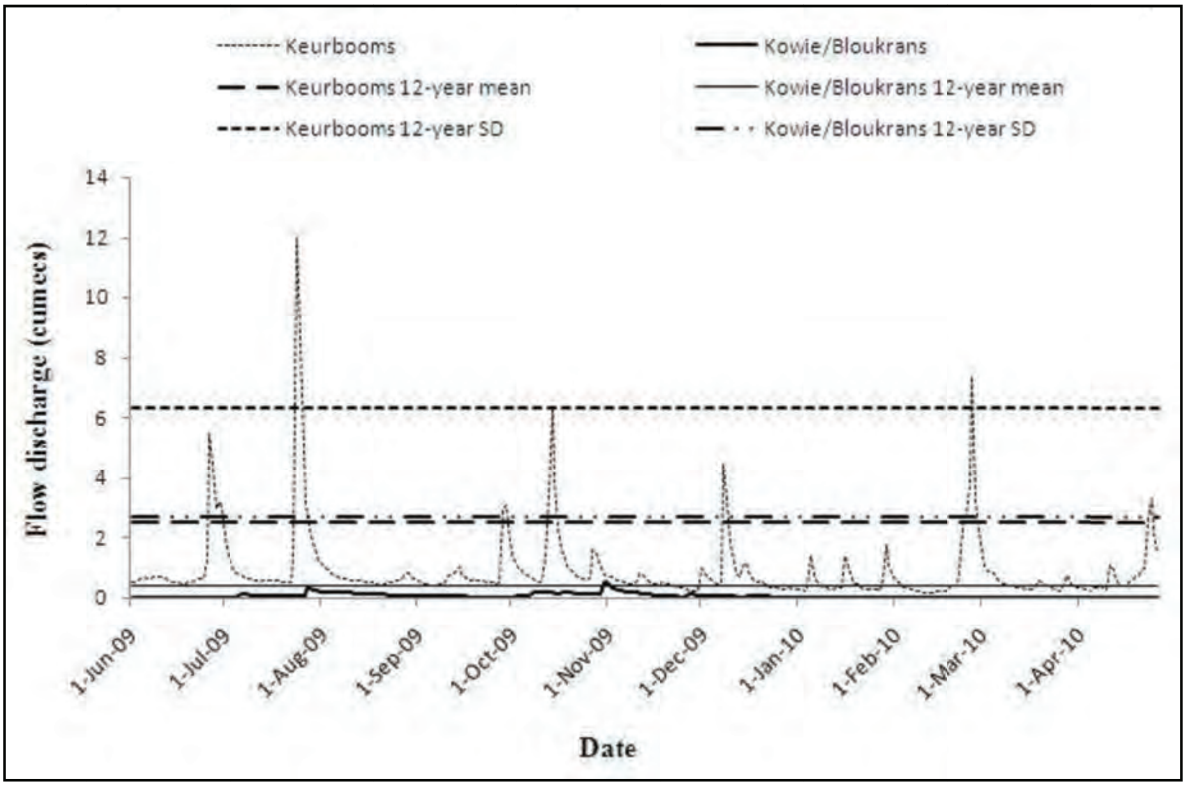


**Figure 4.6 Gauging weir flow data for both rivers from the beginning of the sampling period (June 2009) to the end (April 2010). The mean and standard deviation (SD) lines are for the 12 year timeframe common to both streams.**

***Temperature***

The cumulative mean temperature for each site over a single year yielded expected results, such that the sites at higher altitude have lower degree day values, whereas sites at lower altitudes had higher values (Figure 4.7). All sites corresponding to each other on both rivers had similar total cumulative degree day values. The predictability results infer that both river systems are predictable, with all ten sites having values between 0.6 and 0.7. The duration of the recorded data and the amount of classes that temperatures are partitioned into may have affected the calculated values. The trends for temperature were not as clear as the trends for flow, where a clear division was delineated by the first axis. From the eigenvectors, deductions can be made that the major drivers of variability on axis one are annual temperature (mean), degree days, and mean spring and summer temperatures.

Within the PCA, the temperatures increase as one moves from right to left (Figure 4.8), increasing with decreasing altitude along the longitudinal gradient. Variability on axis two was explained by annual standard deviation, coefficient of variation, predictability, annual range and the standard deviation of the range (Table 4.2).

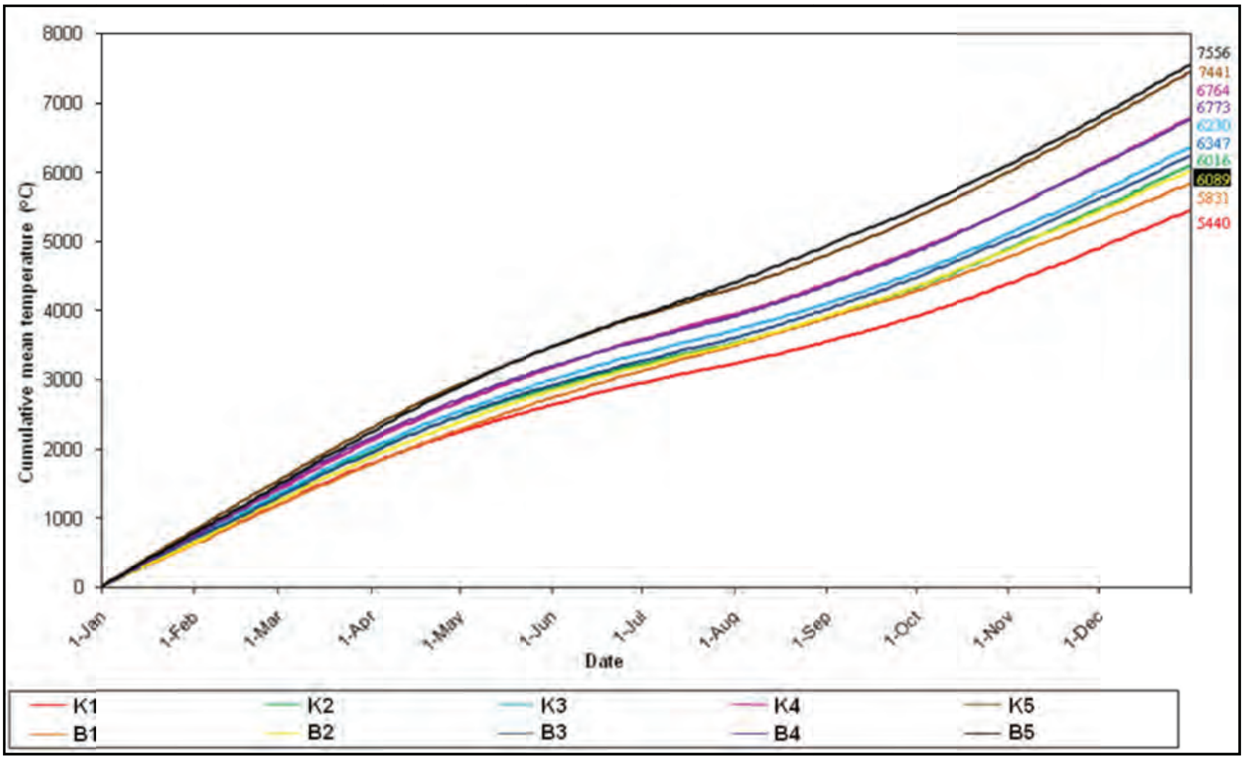


**Figure 4.7 Cumulative mean temperature degree days for the duration of a year (2009- 2010) for each site. Site names in the legend are arranged to correspond to each site on each river, i.e. K1 and B1 are the uppermost sites on the Keurbooms and Kowie/Bloukrans Rivers respectively. These corresponding sites have similar degree day values for a yearly period. Degree day values are displayed to the right of the graph, colour-coded according to the site.**

### *Water Quality*

The main trends from the water quality data revealed that for all the variables measured, there was a general increase from the headwater sites to the lower-most sites for both streams. Water quality variables did not differ greatly between seasons.

### *Macroinvertebrates*

In excess of 32 000 specimens from 67 different families were identified and counted. The Keurbooms River had more macroinvertebrate taxa than the Kowie/Bloukrans, primarily as a result of better flows for the duration of this research. A total of 73 and 61 taxa (ranging between family and species level of identification) were recorded on the Keurbooms and

Kowie/Bloukrans rivers respectively. Sites with a higher number of biotopes generally yielded more taxa compared to sites with a single biotope. The taxon richness per season with downstream distance for the Keurbooms River provided a clear trend for each season and the totals per site, where there is an increase towards the middle reaches, thereafter a tapering off of taxa towards the lower reaches of the river. Interestingly, an inverse trend was found for the Kowie/Bloukrans River, where there was greater taxon richness at the upper and lower sites and a lower diversity at the middle sites (Figure 4.9). B3 was dry for the summer and autumn seasons, thus no macroinvertebrates were present.

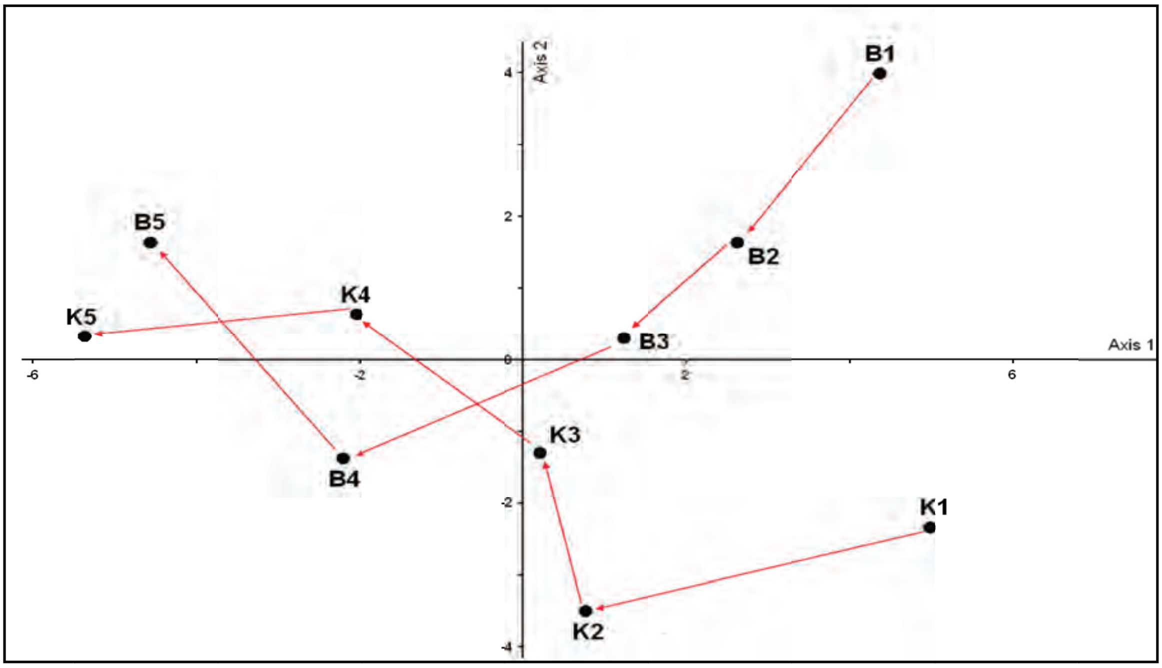


**Figure 4.8 PCA of temperature data with sites. Axis one accounts for 65.7% of the data, whereas axis two accounts for 25.4%. Arrows connect the sites as one progresses from highest to lowest altitude.**

**Table 4.2 Eigenvectors of the temperature parameters from axis one and two that contributed towards the PCA (Figure 4.8). Shaded cells contributed to the distribution of points in Figure 4.8.**

| | Axis one | Axis two |
|---|---|---|
| Annual Temp | -0.2931 | 0.0845 |
| Annual CV | 0.0708 | -0.4516 |
| Predictability | 0.0978 | 0.3834 |
| Annual Range | -0.1062 | -0.3813 |
| Degree days | -0.2928 | 0.0866 |
| Mean Spring | -0.2976 | 0.0269 |
| Mean Summer | -0.2983 | -0.0264 |

Sørensen's similarity indices between site pairs did not yield clear trends. Taxon richness values showed distinct seasonal trends, where for both of the streams, lowest taxon richness was recorded for winter (where daily water temperature range was lowest) and thereafter increasing towards autumn, where the highest taxon richness was recorded. Certain functional feeding group (FFGs) similarities were evident, both on a temporal and spatial basis, on both streams, where macroinvertebrate community structure varied spatially along the longitudinal profile.

A temporal and spatial shift in FFGs was evident at several sites over the duration of this research, particularly spatially. Similar to the River Continuum Concept (Vannote *et al*., 1980), macroinvertebrate community structure varied spatially along the longitudinal profile. Where riparian vegetation was abundant in the upper reaches, Notonemouridae shredders were present for all seasons at the highest site (K1) on the Keurbooms River, and only one season (autumn) at the highest site (B1) on the Kowie/Bloukrans. Pisuliidae shredders were found at B1 for all seasons except winter. Other shredders that were at the upper-most sites were Elmidae (especially at K1) and Leptoceridae.

The middle sites of both streams were dominated by the collector and grazer FFGs, particularly in the form of Simullidae (collector-filterers), Chironomidae (collector-gatherers and scrapers). Lower-most sites on both streams, analogous to the River Continuum Concept, largely comprised collectors. The Keurbooms River yielded more specialist taxa (i.e. taxa occurring at a single site over a single season) than the Kowie/Bloukrans River. Conversely, the total presence values over all four seasons from all five sites were greater on the Kowie/Bloukrans than the Keurbooms, suggesting a higher amount of generalists in the Kowie/Bloukrans River.

The reduced flow over the duration of this research on the Kowie/Bloukrans (and to a lesser extent, the Keurbooms) altered certain biotopes, especially from stones-in-current to stones-out-of-current. This may have excluded certain taxa and introduced others. Eutrophication was also prevalent in the middle reaches of the Kowie/Bloukrans, which would of confounded effects. The water temperature predictability values were plotted against macroinvertebrate coefficient of variation (CV) values (Figure 4.10). A trend for CV to decrease with increasing predictability values was evident for both streams, more so for the Keurbooms ($R^2$ = 0.698) than the Kowie/Bloukrans ($R^2$ = 0.296). Several environmental variables appeared to be driving the highly corresponding taxa, particularly at the lower-most sites. The environmental parameters driving the downstream sites at B5 and K5 are partially due to these sites having the warmest temperatures, highest flows, deepest and widest streams and the highest conductivity readings. Temperature variability was the main driver for the K1, K2 and K3 sites, whereas temperature predictability drives B1, B2 and B3 (Figure 4.11).

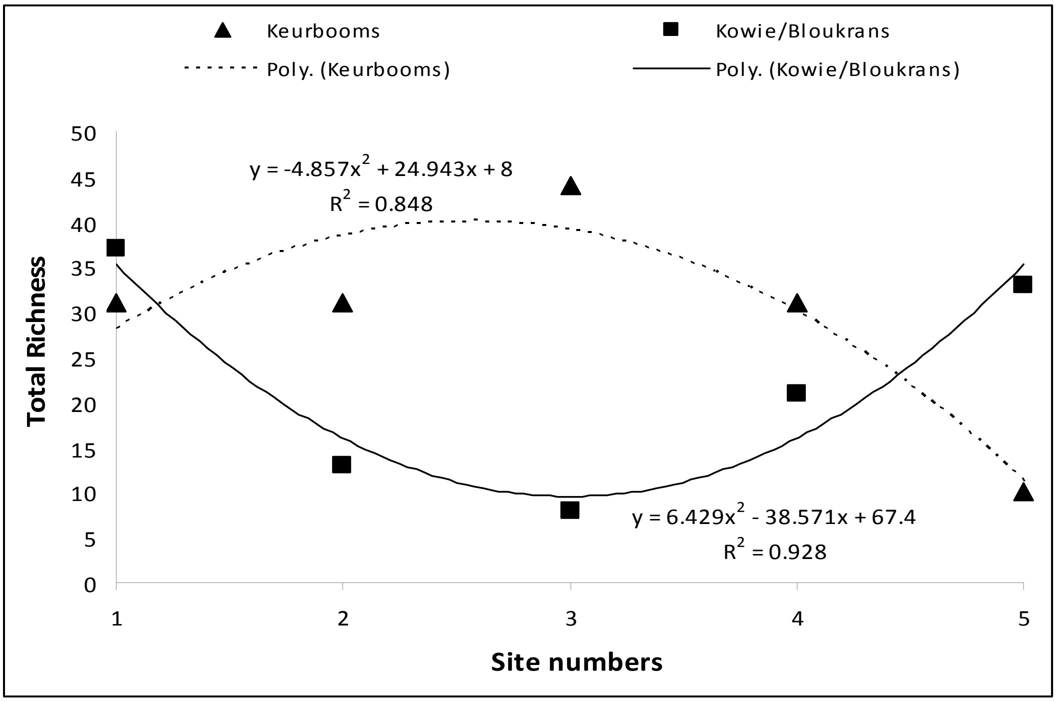


**Figure 4.9 Total macroinvertebrate richness for all seasons for both rivers (Polynomial trendlines are of the 2nd order).**

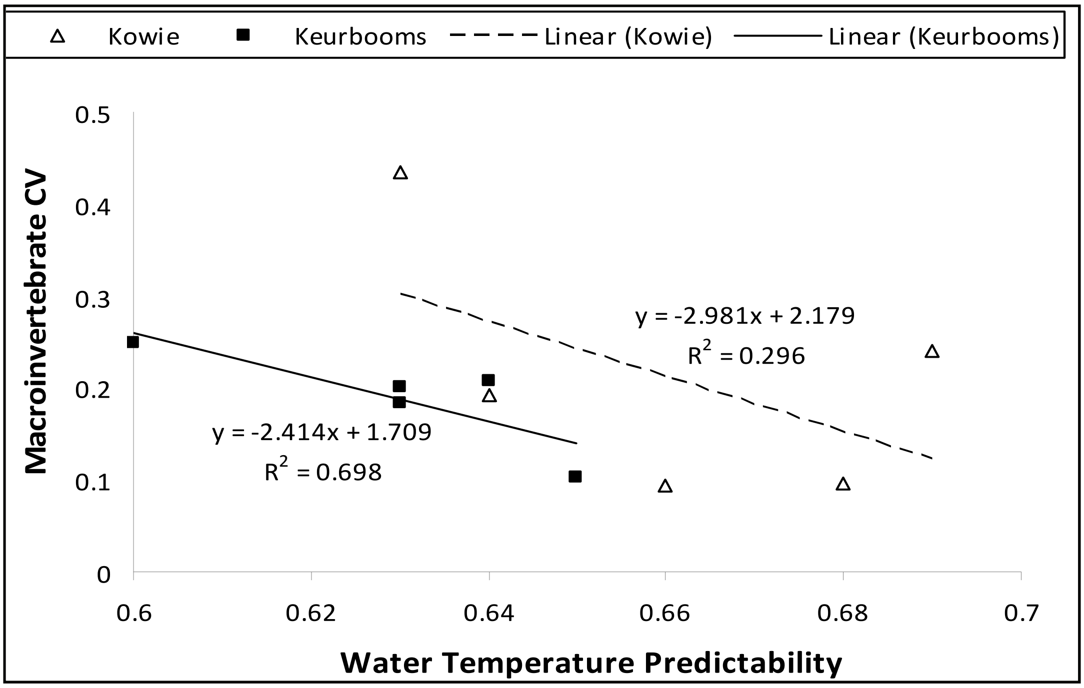


**Figure 4.10 Water temperature predictability values plotted against macroinvertebrate coefficient of variation (CV) for each stream system.**

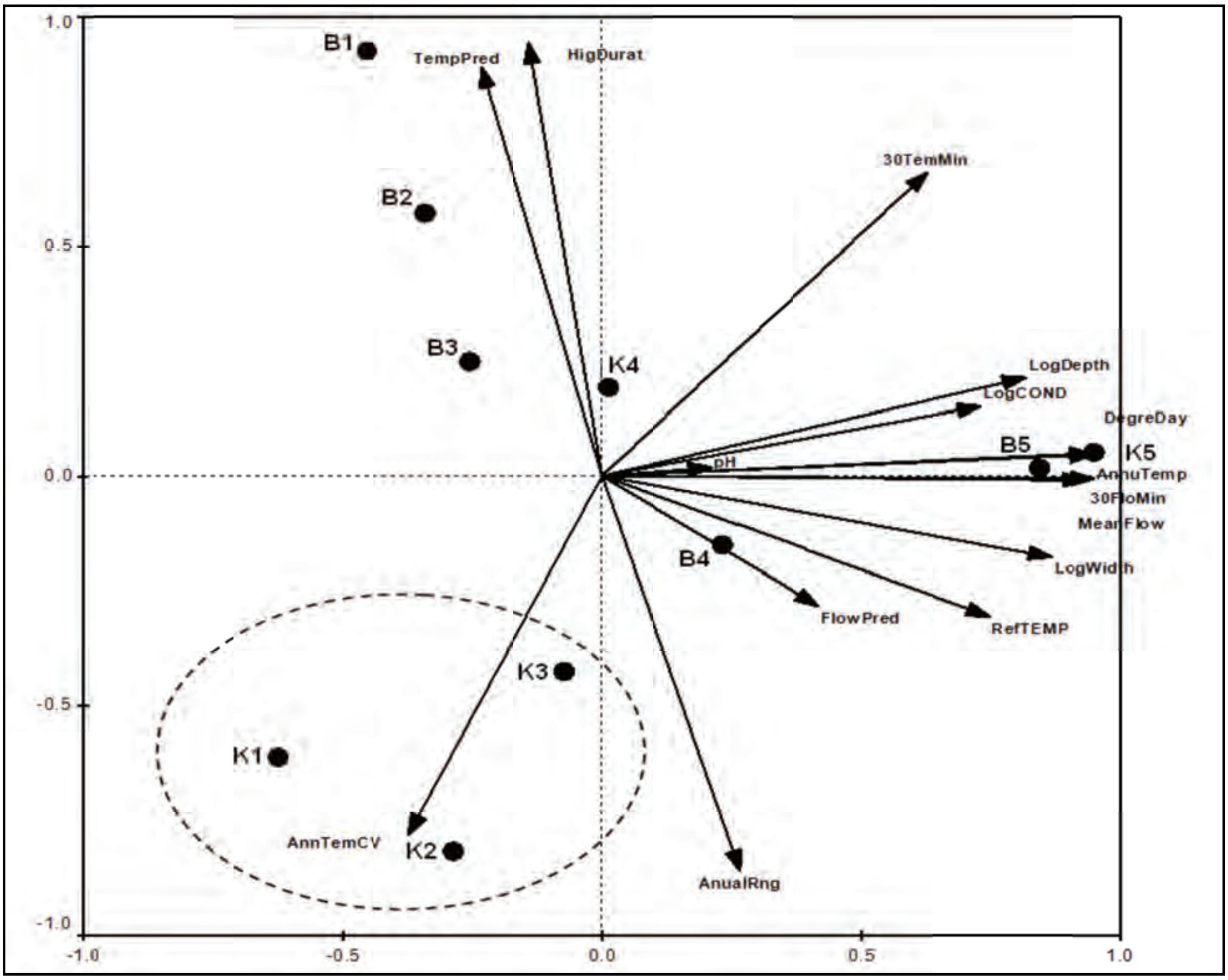


**Figure 4.11 NMS of the environmental parameters with the highest correlations, indicating which sites were driven by them. Axis one accounts for 49.3% of the data, whereas axis two accounts for 26.6%. Sites are represented by the points and the environmental parameters are represented by the arrows. Dashed oval indicates the three sites most affected by annual temperature coefficient of variation.**

***Key Findings***

There was a tendency for macroinvertebrate assemblages to differ on a temporal basis as temperature predictability values decreased (higher macroinvertebrate CV, indicating seasonal turnover). This trend was more evident on the Keurbooms River ($R^2$ = 0.698) than the Kowie/Bloukrans River ($R^2$ = 0.296). This may be attributed to the more frequent supply of water on the perennial Keurbooms than on the non-perennial Kowie/Bloukrans. Water temperature predictability values calculated from the year cycle did not provide a direct relationship with macroinvertebrate communities. The range of predictability values for all ten sites was between 0.60 and 0.69, which indicated that there were no sites that stood out having either highly or poorly predictable thermal regimes. The site having the highest predictability value was B2 (0.69), which, over the seasons, yielded 13 different taxa under flowing conditions. Of these taxa, six (46%) were present for every season, leading to the assumption that these taxa are not greatly influenced by temperature. The lowest water temperature predictability value calculated was for K2 (0.6), where 31 different taxa were collected over the seasons. Of these taxa, three (10%) were present for all the seasons.

An inference was made that the more temperature predictable sites have less macroinvertebrate taxa turnover across seasons, making it easier to predict what macroinvertebrate taxa may be present for a particular season due to more stable communities, depending on the flow regime (flowing versus non-flowing). This trend was more conclusive for the Keurbooms than the Kowie/Bloukrans.

The major environmental drivers at the K1, K2 and K3 sites were annual temperature coefficients of variation and annual temperature range. Thus, these sites have high temperature variability over the years' cycle, encouraging a range of thermal niches for macroinvertebrate communities to establish. Out of these sites, the greatest macroinvertebrate diversity is present, being K3 with 44 different taxa, where K2 had 31 taxa and K1 with 31 taxa. This could indicate that the high annual temperature fluctuations and persistent flowing conditions (K3) may well be the driver for a high temporal macroinvertebrate turnover.

From the taxa collected across all seasons for both rivers, several taxa were only found on one stream system; 33 and 21 unique taxa on the Keurbooms and Kowie/Bloukrans systems respectively. An interesting trend is that the majority of the unique taxa on the Keurbooms (mostly Simullidae, Ephemeroptera and Trichoptera) were found in the stones-in-current biotope, suggesting they have adapted to conditions of constant flow. Similarly, the taxa unique to the Kowie/Bloukrans (Mollusca, Diptera, and some Trichoptera) were predominantly found in biotopes with calm or stagnant waters, further suggesting that these taxa have adapted to conditions consisting of pools and sluggish waters due to inconsistent flow. These two trends distinguish the types of river systems from one another, i.e. perennial versus non-perennial.

Sites exhibiting greatest macroinvertebrate temporal turnover were B1, K1, K2, K3 and K4, where high variation in temperature and flow contributed to seasonal changes. This allows different macroinvertebrate taxa to become better suited to a range of ecological niches. Thus, where seasonal variation in temperature and flow are more prominent, higher macroinvertebrate turnover is likely to occur.

Variability in water temperatures and flow were the driving environmental variables that resulted in the dissimilarity of macroinvertebrate community structures across sites and seasons. This relationship was, however, more pronounced at some sites than others, with variation occurring spatially and temporally. This was evident on both the perennial and non-perennial streams.

### Conclusions and Recommendations

Macroinvertebrate species turnover through the seasons was greater for sites with lower water temperature predictability values (i.e. more variable) than sites with high predictability values.

Aquatic macroinvertebrates do respond in a predictable manner to modifications in their environment. This was particularly evident in relation to variability in flow and water temperature, depending on the sensitivity of the taxa to changes in their habitat, although this was more evident amongst certain taxa than others. Functional feeding groups did not vary temporally, however did appear to differ spatially, similar to the River Continuum Concept (Vannote *et al*., 1980).

Changing precipitation patterns and increased air temperature are the primary climatic variations that are predicted to occur in South Africa with climate change. In Africa, climate change threatens freshwater systems due to alterations in temperature and precipitation, which would have unfavourable impacts on water quantity, water quality and water temperature. It becomes an important task to discover to what extent climate change may have on amplifying existing variability in aquatic ecosystems and how this may affect the predictability of these ecosystems.

In a semi-arid country such as South Africa, macroinvertebrates are at a risk, particularly in non-perennial rivers where flow predictability is low. Anthropogenic impacts could threaten macroinvertebrate communities on perennial rivers, where the construction of dams and water abstractions for irrigation may alter the system to have a predictable flow regime, resulting in less species turnover and diversity losses. The results from this research could aid decision makers in making correct choices in the future, especially on streams with unpredictable flows, for example, the management of water releases from dams to uphold the ecological reserve. With these findings, we are potentially able to predict what the aquatic macroinvertebrate communities may consist of at various stages on streams, depending on the degree of water temperature and flow predictability.

### 4.1.3 Importance of temperature extremes on aquatic macroinvertebrates in the Western Cape

**Related publication**

Dallas H.F. 2010b. Water temperatures and the Reserve (WRC Project: K5/1799): The importance of temperature extremes on aquatic macroinvertebrates in the Western Cape. Report Number 1799/17 produced for the Water Research Commission. The Freshwater Consulting Group.

## Introduction and aims

Lotic systems in regions of seasonal climates exhibit diel (daily) and annual (seasonal) temperature periodicity patterns (Ward, 1985). Hourly temperatures are generally lowest during the night time or early morning and highest in the mid to late afternoon, although this may shift with season (e.g. Hopkins, 1971) and size of river. Small, heavily canopied streams and large, deep rivers exhibit the least diel ranges in water temperature, while shallow streams exposed to direct solar radiation and braided rivers exhibit the largest diel ranges in water temperature (Ward, 1985; Constanz et al., 1994). Seasonally, temperatures generally exhibit a sinusoidal pattern with temperatures highest in the summer and lowest in the winter. Inter-annual variation is less predictable with variation in temperature regimes reflected as ‘hot-dry’ and ‘cool-wet’ years (Poole et al., 2001).

Effects of temperature extremes on biotic communities have been shown to be important, and South African river systems seem to be defined by extremes rather than averages (Rivers-Moore et al. 2008b, c). More specifically, cumulative daily maximum water temperatures have been shown to have the greatest effect on the distribution of aquatic species (Rivers-Moore et al. 2008c). In the south-western Cape region of South Africa, highest summer temperatures coincide with the lowest flows. This combination has potential to lead to greater environmental stress for the aquatic organisms living in rivers of this region. Over time, organisms may have evolved mechanisms to cope with this harsh environment.

Global and regional climate change models predict that rainfall will decrease by up to 20% in this region, while air temperatures may increase by 2 to 3°C (Midgley et al., 2005; Dallas 2009, Dallas and Rivers-Moore 2009a, b). Less rainfall would result in a reduction in stream flow (New, 2002). Catchments with lower rainfall under the current climate were shown to be more likely to exhibit greater changes in stream flow relative to catchments with more rainfall (New, 2002). Reduced stream flow and increase air temperature is likely to result in increased water temperature particularly in rivers that are not groundwater dependent and in which water temperature is influenced by surface flow. Recognising that the south-western Cape is an area of high water stress, and that stream flow is likely to decrease further in the future in response to both climate change and increased water demand, it was considered valuable to develop a study that examines the likely effects of temperature extremes on aquatic macroinvertebrate assemblages. The specific aims of this study were:

- to determine the aquatic invertebrate assemblages present at selected sites at the start of the summer season (late November) and to compare them to assemblages present at the end of the summer season (early March), and
- to compare these assemblages for a range of sites with high or low summer temperature ranges, relative to other sites in the region, which vary in terms of their thermal regime over the summer period.

## Methods

Ten relatively unimpacted upper catchment sites were selected based on their mean and maximum temperatures, as well as their ability to be paired with another site. Five of these were at the upper end of the temperature range and exhibited high temperatures (Rondegat[1], Groot[2], Wit[3], Boesmans[4] and Palmiet rivers), while five of these were at the lower end of the temperature range and exhibited low temperatures (Dwars[1], Matjies[2], Rooielskloof[3], Duiwelsbos[4] and Bok rivers) relative to the other sites (superscripts indicate four site pairs).

Hourly water temperature data were recorded at each site and analysed to characterise each site in terms of its thermal regime. Specifically, inter-annual variation between the summer of 2008-2009 and the summer of 2009-2010 were examined. Temperature metrics for describing "Indicators of Thermal Alteration (after Richter et al. 1996) were calculated for all sites for the period 16 December 2008 to 6 March 2009 and 16 December 2009 to 6 March 2010. *In situ* water chemistry (conductivity, pH and dissolved oxygen) were taken at each site.

Aquatic macroinvertebrates were sampled in the stones biotope in late November 2009 and early March 2010, which represented the start and end of the warmest period. A kick-net (30 x 30 cm, 1-mm-mesh size) was used to sample the stones for approximately 5 minutes. Loose substratum was agitated, and dislodged organisms were collected downstream in the net. Collected material was emptied into a tray; debris was removed, and organisms were identified to the family level in the field. The full sample was then collected and preserved in 70% alcohol for laboratory identification of the organisms, which were identified to species where possible, and the number of individuals per species was counted. Multivariate and univariate analyses were undertaken on the data.

## Summary of Major Results

### Water temperature

The thermal regimes varied considerably across the spectrum of rivers included in this study. Most significant was the variability of water temperature within a site, with some sites exhibiting greater thermal stability, most likely is response to greater groundwater dependence, compared to more surface-dependent sites, which had very a high coefficient of variation. Water temperature varied substantially within each pair of sites over the summer periods (Figure 4.12).

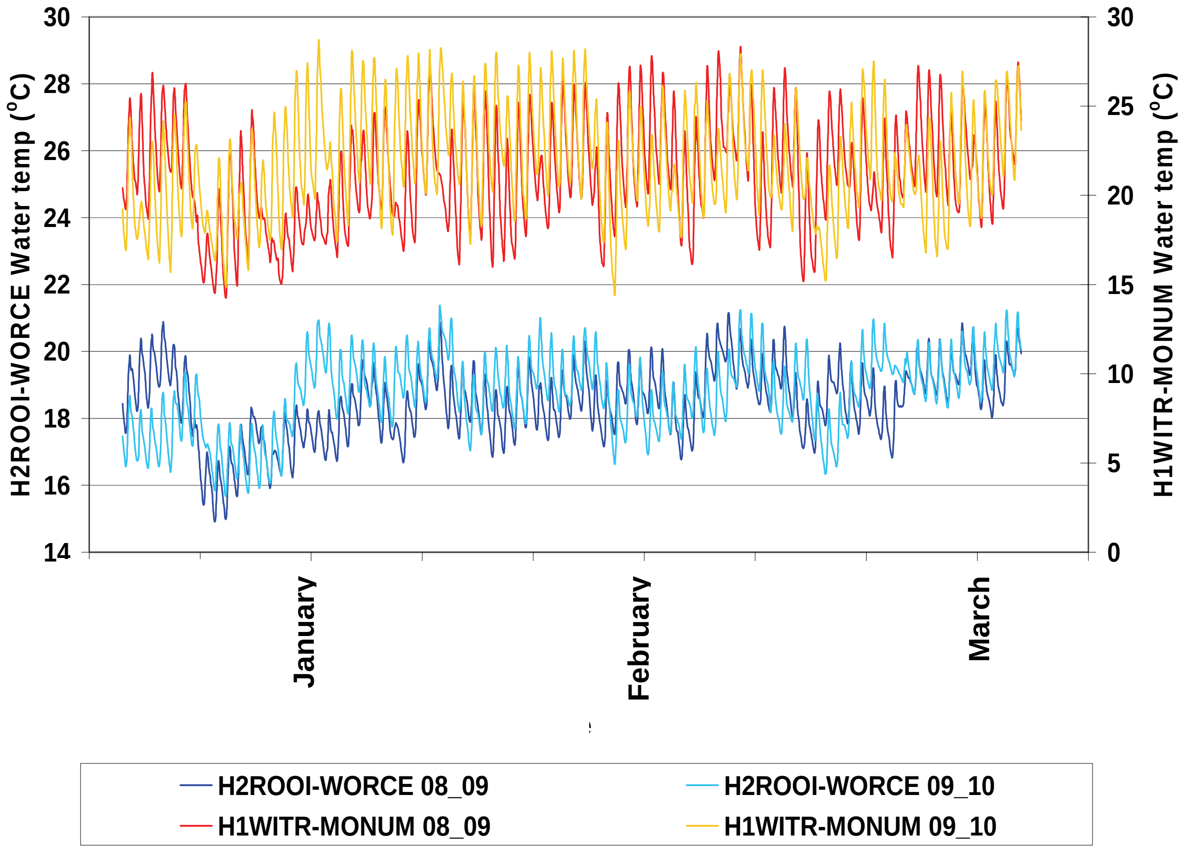


**Figure 4.12 Water temperatures recorded at hourly intervals over the summer period for the H1WITR-MONUM and H2ROOI-WORCE sites**

Cluster and ordination using temperature metrics calculated for each site and summer period, indicated that metric data grouped largely by site rather than summer period. Examination of this data using PCA analysis showed that cumulative % variance explained by the first two PCA axes was high (94%). Mean annual temperature and high temperatures contributed most to variation on the x-axis, while CV, predictability, range and minimum temperatures to the y-axis. The pairs of sites selected for comparison all showed consistent differences in temperature metrics. Relating this to these selected rivers, the Bok River was on opposite ends of the ordination x-axis to the Wit, Palmiet and Rondegat Rivers, while the Groot and Wit/Palmiet Rivers were separated on the Y-axis (Figure 4.13).

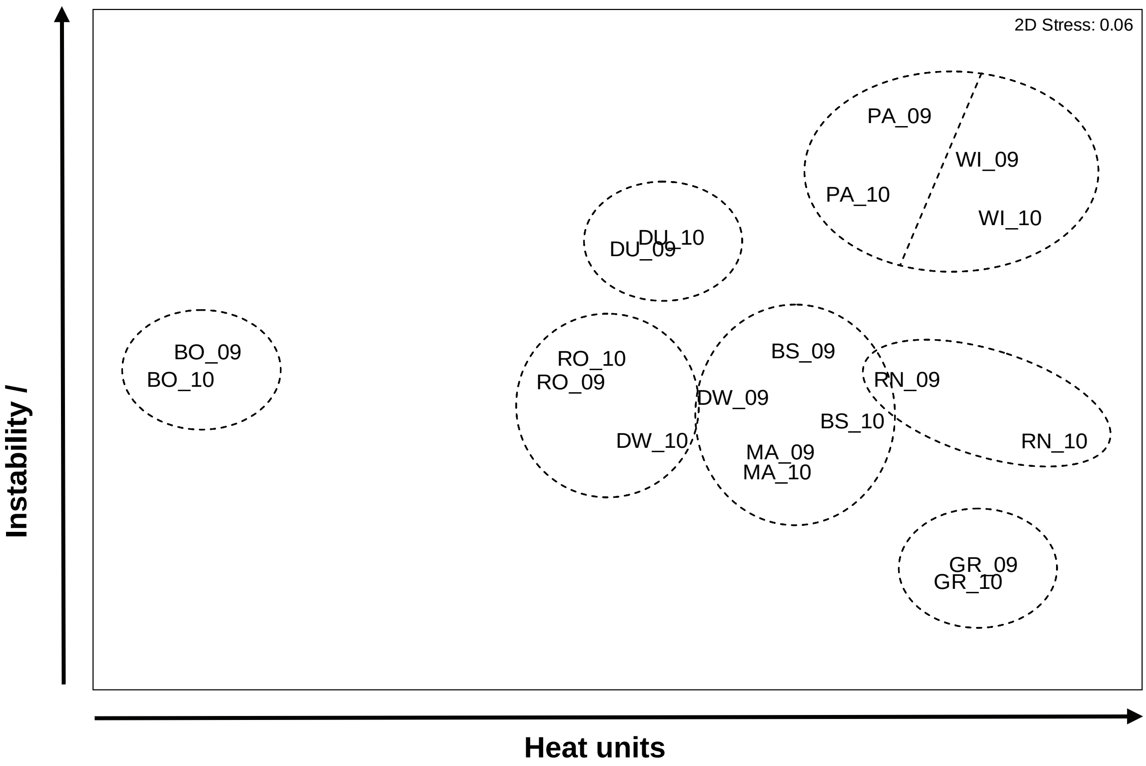


**Figure 4.13 Ordination analysis of temperature metrics for each site during two summer periods (DW=E1DWAR-KRAKA, RN=E1ROND-EWR03, GR=E2GROO-EWR06, MA=E2MATJ-MATJI, WI=H1WITR-MONUM, RO=H2ROOI-WORCE, BS=H5BOES-PIKKI, DU=H7DUIW-MARLO, BO=J1BOKR-BOKRI and PA=G4PALM-NUWEB; 09 = summer 2008-2009; 10 = summer 2009-2010)**

Absolute maximum temperatures and total degree days (both mean and maximum) also varied considerably from one site to another and within pairs of sites. Cumulative maximum daily temperatures over the 2008_2009 summer period varied from 1572 degrees at Bok river to 2251 degrees at Wit river (Figure 4.14) This represents a difference of 678 degrees over the summer period. Absolute minimum and maximum temperatures, which provide an indication of the extremes likely to be experienced by the invertebrate assemblages at each of site, revealed that the Bok River had the lowest absolute maximum temperature, while the Wit, Palmiet, Groot and Rondegat Rivers had the highest.

**Macroinvertebrate data**

Examination of the macroinvertebrate data collected over the three months representing the start and end of the summer period did not reveal any distinct patterns with respect to the grouping of macroinvertebrate assemblages, the richness or diversity, or the number of taxa within particular orders. The response of individual species showed that some species, including four mayflies and one blackfly species, were more common in early summer compared to late summer. Late summer had more chironomid species and athericids.

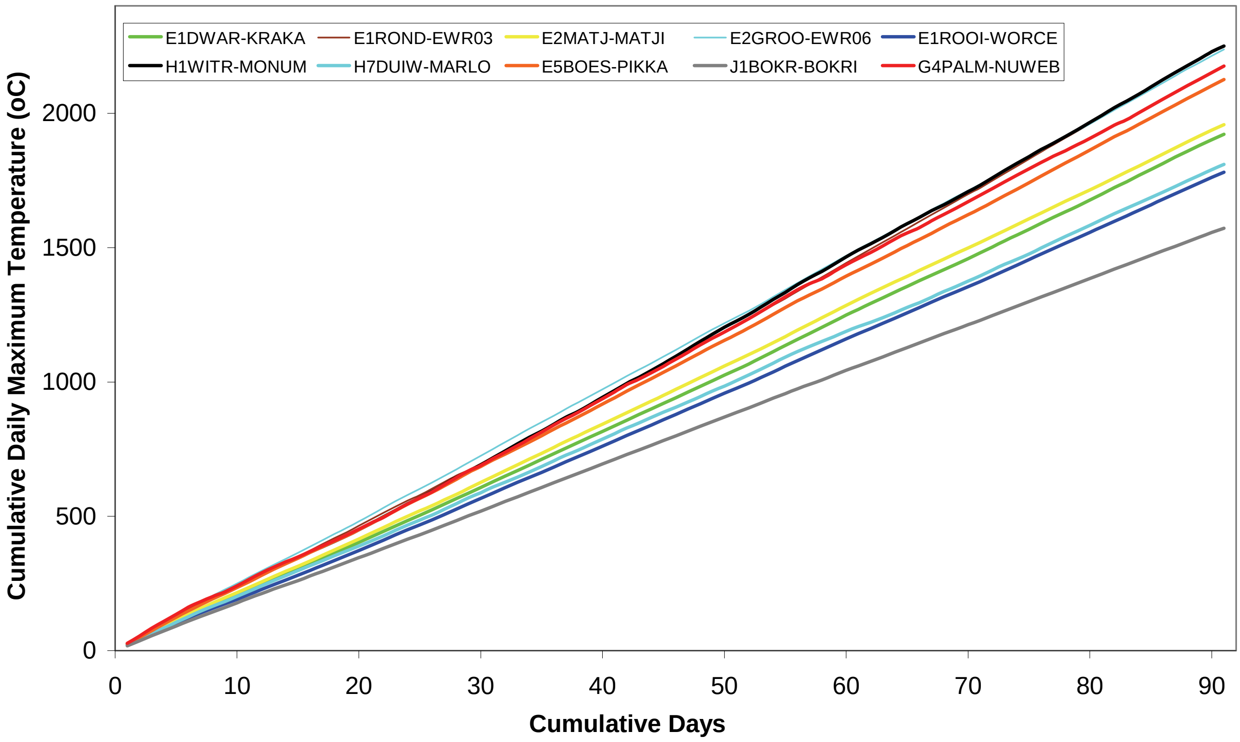


**Figure 4.14 Cumulative daily maximum temperatures for ten sites given for the summer period (16 December 2008 to 16 March 2009)**

## Conclusions and Recommendations

While the water temperature data showed distinct difference in thermal conditions across the range of sites, the faunal trends observed could not be related to these thermal conditions. Likely reasons for this are the relatively course level at which the sampling took place, in particular sampling over two time periods only. For an understanding of life history responses of individual species to abiotic drivers such as water temperature it is necessary to do sampling more frequently. In this way an understanding of the life cycle stages, including growth rate, emergence, egg laying and diapause may be attained.

A related study that focuses on six rivers including the Wit and Rooielskloof Rivers, which were used in the present study, had a sampling frequency of 1 month (see section 4.1.1). Data from this study suggest that this is an adequate sampling frequency to provide an understanding of life history trends, although a frequency of every two weeks would provide an even greater understanding. A further consideration is the relatively large mesh size of the net used for sampling (950 µm), which would bias data in favour of larger individuals.

Data of Ross-Gillespie (see section 4 1.1.) shows that the telagonodid, *Lestagella penicillata*, nymphs emerge from October to December, while egg hatching occurred earlier (December) in warmer rivers (e.g. Wit) and later (January) in colder rivers (e.g. Rooielskloof). Early instars would not have been captured using the sampling mesh in the present study.

The present study has focused on water temperature as a driver of biotic communities. It is well known that invertebrates are influenced by a host of abiotic factors including, in addition to water temperature, flow, substrate, nutrient input (allochthonous versus autothonous) and sediment input (Dallas and Day 2004). Biotic factors such as predation pressure may also contribute to observed patterns in the spatial and temporal distribution of invertebrates. While an attempt was made to restrict variation within pairs of sites in terms of these other factors, it is possible that for some pairs results were confounded by non-thermal related factors.

Further, research by Schael and King (2005) provides strong evidence of the existence of distinct assemblages of aquatic macroinvertebrates within rivers and catchments. They maintain that rivers in upper catchments have biotic fingerprints that distinguish the one from the other. The signatures are due to a unique mix of species per river and catchment derived from a regional pool of mostly common species. The sites in the present study included ones in the Olifants/Doring, Breede and Palmiet catchments, which were all shown by Schael and King (2005) to have distinct catchment signatures.

The sites identified in the present study provide an excellent option for further more detailed examination of aquatic invertebrates on a more frequent sampling programme and using a finer mesh size.

## 4.2 Thermal tolerances of selected aquatic macroinvertebrates from rivers of the Western Cape

**Related publications**

Dallas H.F. and Rivers-Moore N.A. 2011. Critical Thermal Maxima of aquatic macroinvertebrates – towards identifying bioindicators of thermal alteration. Hydrobiologia (DOI: 10.1007/s10750-011-0856-4)

Dallas H.F. and Ketley Z.A. 2011. Upper thermal limits of aquatic macroinvertebrates: Comparing critical thermal maxima with 96-$LT_{50}$ values. Journal of thermal biology 36: 322-327.

Dallas H.F., Ketley Z.A. and Gray E. 2009. Water temperatures and the Reserve (WRC Project: K5/1799): Report on available technology for laboratory testing of the thermal tolerance of aquatic macroinvertebrates and construction of appropriate laboratory setup. Report Number 1799/4 produced for the Water Research Commission. The Freshwater Consulting Group and Freshwater Research Unit, University of Cape Town.

Dallas H.F. 2010. Thermal tolerances of selected aquatic macroinvertebrates from rivers of the Western Cape. Report Number 1799/9 produced for the Water Research Commission. The Freshwater Consulting Group.

### Introduction and aims

Water temperature is an important abiotic driver of aquatic ecosystems (e.g. Caissie, 2006; Dallas, 2008, 2009; Smith, 1972; Ward, 1985; Webb et al., 2008) and temperature influences many aspects of an organism's existence such as metabolic (Eriksen, 1964), growth (e.g. Vannote and Sweeney, 1980) and feeding (Kishi et al., 2005) rates; fecundity (e.g. Brittain, 1991); emergence (McKie et al., 2004); behaviour and ultimately survival. Water temperature is modified by several anthropogenic activities (Dallas, 2008) including river regulation and water abstraction (e.g. Olden and Naiman, 2010; Ward, 1985; Webb et al., 2003); inter-basin water transfer; thermal discharge and agricultural irrigation return-flow (Wellborn and Robinson, 1996); changes in land-use such as removal of riparian vegetation (Rutherford et al., 1997); and climate change and global warming (e.g. Dallas and Rivers-Moore, 2009a, b; Hogg et al., 1995). An understanding of an organism's thermal sensitivity or tolerance is therefore of importance in determining the likely effect of changes in water temperature on aquatic ecosystems, and also in the generation of thermal criteria for the protection of aquatic ecosystems.

Northern Hemisphere studies on thermal tolerances of aquatic insects are numerous (e.g. Brittain, 1991; Elliot, 1987a, b; Hogg et al. 1995; Huryn, 1996; Nebeker, 1971a, b; Nebeker and Lemke, 1968; Ward and Stanford, 1982; Wellborn and Robinson, 1996), while fewer studies have been undertaken in the Southern Hemisphere (e.g. Campbell, 1986; Dallas, 2008; McKie et al., 2004). Upper thermal limits are typically determined experimentally within a controlled laboratory environment and are often based on dynamic, non-lethal or static, lethal methods (Lutterschmidt and Hutchison, 1997).

The dynamic method involves changing temperature at a constant rate until a predefined sub-lethal endpoint, which is used to estimate lethality, is reached. A commonly used dynamic method is the Critical Thermal Method (CTM), which is used to determine a Critical Thermal Endpoint (CTE), expressed as either critical thermal maxima (CTmax) or minima (CTmin). CTE is a behavioural stress response and is defined as the "arithmetic mean of collected thermal points at which locomotor activity becomes disorganized to the point at which the organism loses its ability to escape conditions that will promptly lead to its death" (Cox, 1974 cited by Ernst et al., 1984). In fish, it often includes loss of equilibrium and onset of muscle spasms (Beitinger et al., 2000). The analogous behavioural stress response of aquatic macroinvertebrates varies amongst taxa but generally includes either a loss of grip and inability to remain attached to a substrate or increased movement, both followed by immobility, with a lack of response when stimulated with a jet of water (Dallas, pers. observation). In aquatic insects CTM has been widely used for a range of organisms including terrestrial spiders (Hanna and Cobb, 2007) and is comparatively quick and requires a relatively small sample size of test organisms (approximately n=30). The static method, which is equivalent to the incipient lethal temperature (ILT) technique (Fry, 1947; Beitinger et al. 2000), involves holding duration constant while temperature is varied, with assessments based on survival of a proportion of a sample (Terblanche et al., 2007). The resultant lethal temperature ($LT_{50}$) is the temperature at which 50% of the sample survives in a specified time. The experimental time period is of longer duration, normally four to ten days, and requires a large number of individuals (approximately n=150).

The development of temperature criteria is important for the effective protection and management of aquatic ecosystems. Temperature criteria typically have two key elements: a threshold temperature that signals when adverse biological responses are likely to occur, and an averaging period that indexes the duration of exposure likely to trigger that response (Sullivan et al., 2000). Averaging periods range from an instantaneous maximum, to a 7 day, monthly or seasonal average (Sullivan et al, 2000). Averaging approaches typically use a 7-D moving window, for example, the average of daily maximum temperature (ODEQ, 1995) or the average of daily mean temperatures (US EPA, 1977) of seven warmest consecutive days. Risk occurs when the stress magnitude, frequency and duration exceed the species ability to deal with that stress. A seven day moving average of daily maximum water temperatures has been related to fish distributions in the Sabie Rivers (Rivers-Moore et al., 2005b) and blackfly outbreak probabilities (Rivers-Moore et al., 2008a). Sub-lethal values related to, for example, growth rates and life cycle cues for activities such as emergence or spawning; also provide useful values for developing temperature criteria. For example, King et al. (1998) have shown that temperatures between 18 and 19°C trigger spawning of several of South Africa's indigenous fish species. The primary aims of this component were:

- to examine the upper thermal tolerance of a range of aquatic macroinvertebrates using the CTM;
- to identify taxa suitable for use as test organisms;
- to identify thermally sensitive families;
- to estimate upper thermal limits of select taxa using the $LT_{50}$ method;
- to explore the relationship between the two methods; and
- to determine if the simpler CTM can be used for future thermal testing while allowing for extrapolation to longer duration thermal stress.

The secondary aims were to undertake preliminary experiments to determine potential sources of variation in CTmax, including examination of differences in CTmax within macroinvertebrate families and between sites (rivers); and to explore the relationship between CTmax and start (acclimation) temperature; and CTmax and age.

## Methods

Aquatic macroinvertebrates were collected from rivers in the vicinity of Cape Town, with the primary collection site in the Eerste River in the Hottentots-Holland Mountains in the south-western Cape, South Africa. Organisms were collected using a 950µm diameter mesh, kept cool and returned to the freshwater aquaria where they were held at 17$^{o}$C, which approximated water temperatures at capture.

For CTM experiments, organisms were placed in an experimental chamber, which was immersed in a water bath in the temperature controlled aquarium. Each experiment was approximately 90 minutes in duration, which included a 30 minute control phase, followed by a 60 minute experimental phase during which the water temperature was raised at a constant rate of 0.34$^{o}$C per minute using a circulating heater (Julabo™). The reaction to heating was assessed by observing the behavioural response of test organisms, including the pre-experimental behaviour, the Point of Thermal Reactivity (PTR) and the Critical Thermal Endpoint (as CTmax).

A Suitability Rank (SR) and Thermal Sensitivity Rank (TSR) were assigned to each family to facilitate comparison amongst families. The SR was derived by assessing the overall suitability of each family as test organisms using the scale: 1 = highly suitable; 2 = suitable but with limitations; 3 = unsuitable, where suitability was assessed based on identification of a distinct behavioural response (PTR and CTE); size of organism, ease of identification and availability. TSR was ranked based on median CTmax of each taxon, where 1 = highly thermally sensitive (Median CTmax <33$^{o}$C); 2 = moderately thermally sensitive (Median CTmax ≥33 and <39$^{o}$C); 3 = not thermally sensitive (Median CTmax ≥39$^{o}$C).

For $LT_{50}$ experiments, organisms were placed in experimental chambers immersed in glass aquarium tanks, filled with stream or dechlorinated freshwater, which was heated using water heaters to five different temperatures. Water temperature was recorded at 15 minute intervals throughout the experiment using Hobo TidbiT® v2 loggers (Onset Computer Corporation, 2008) placed in each of the tanks. A control temperature, which approximated water temperatures at capture, was included to test potential mortality factors such as influences from chambers, air, water or a lack of food. Chambers were checked for survival every 24 hours for four days (96-h) subsequent to the target temperature being reached.

## Summary of Major Results

### CTmax and behavioural responses

CTmax was determined for a total *ca*. 1,500 individuals from 37 families. However families represented by fewer than 10 individuals have been excluded resulting in detailed analysis of 27 families. The results of a Kruskal-Wallis test were significant (H = 729.56, 26 d.f., N = 861, p<0.001) and the mean ranks of CTmax were significantly different among the 27 families (Figure 4.15).

Median CTmax for organisms acclimated to 17$^{o}$C varied from 28.9$^{o}$C for the crustacean, Paramelitidae, to 42.2$^{o}$C for the odonate, Libellulidae and 42.3$^{o}$C for the coleopteran, Gyrinidae. Behavioural responses of aquatic macroinvertebrates to heating varied amongst families but generally included, either a loss of grip and inability to remain attached to a substrate or increased movement, both followed by immobility, with a lack of response when stimulated with a jet of water. Some similarities were observed amongst mayfly and stonefly nymph families. Pre-experimental behaviour, the PTR and CTE, SR and TSR were determined for all experimental taxa. Of the 27 families examined, four were both thermally sensitive and highly suitable as test organisms, including families Paramelitidae, Notonemouridae, Teloganodidae and Philopotamidae. Five families were moderately sensitive and highly suitable, including Palaemonidae, Heptageniidae, Leptophlebiidae, Corydalidae and Aeshnidae.

Preliminary experiments to determine potential sources of variation in CTmax, including differences in CTmax within selected macroinvertebrate families (Paramelitidae, Baetidae, Leptophlebiidae, Telagonodidae and Notonemouridae) and between sites, revealed that no significant differences were observed within three target rivers. Specifically, no differences were observed between CTmax for *Aphanicerca capensis* and *Aphanicercopsis tabularis* from Window Gorge, and *Adenophlebia peringueyella, Aprionyx perterseni* and *Castanophlebia calida* from the Eerste, and *Ephemerellina barnardi* and *L. penicillata* from the Rooielskloof. Further consolidation of experimental data to a higher taxonomic level (mostly order) on the basis of the median CTmax indicated that Amphipoda, Plecoptera, Trichoptera and Ephemeroptera are more thermally sensitive, in comparison to Hemiptera, Coleoptera and Gastropoda (Figure 4.16).

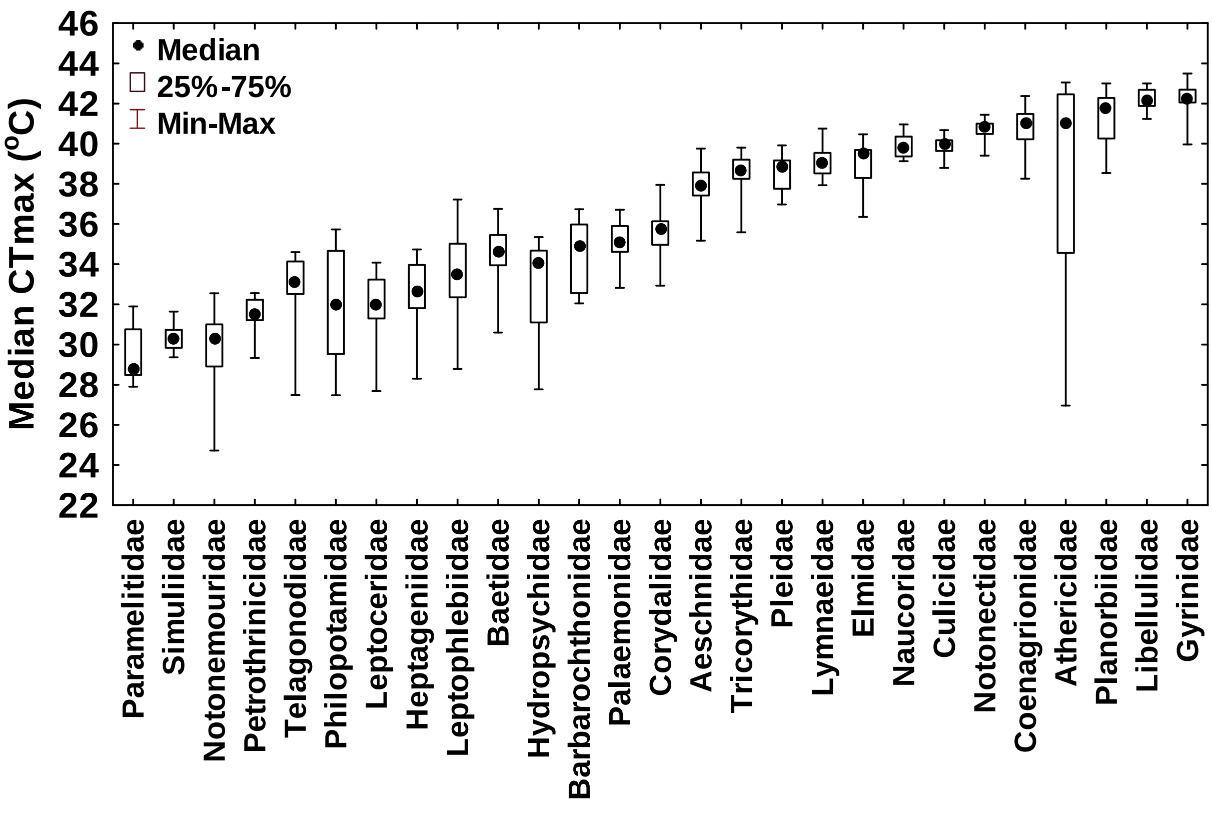


**Figure 4.15 Median CTmax, 25th and 75th percentile and minimum and maximum values (°C) for each family tested. Families are arranged from lowest to highest median CTmax. All test organisms were acclimated to 17°C**

Acclimation temperature was shown to influence CTmax, with all species acclimated to 10°C having a lower median CTmax compared to organisms acclimated to 17°C. Size (as a surrogate of age and measured as interocular distance) was shown to correlate in *L. penicillata* but not in other taxa, namely Leptophlebiidae, Corydalidae, Aeshnidae, Notonemouridae, Hydropsychidae or Philopotamidae. Possible reasons for this include the size of the collecting mesh or the fact that larger specimens were targeted.

**Comparison of Critical Thermal Maxima with 96-LT50 values**

The 24-h, 48-h, 72-h and 96-h $LT_{50}$ values for ten species were determined. The notonemourid, *A. capensis,* had the lowest 96-h $LT_{50}$ (23.4°C), followed by the amphipod, *Paramelita nigroculus* (96-h $LT_{50}$ = 24.3°C), while the odonate, *A. imperator/speratus* had the highest 96-h $LT_{50}$ (33.6°C). The 10-d $LT_{50}$ for *P. capensis* was 30.4°C. Relative thermal sensitivity was largely the same as determined by both CTmax and $LT_{50}$ experiments.

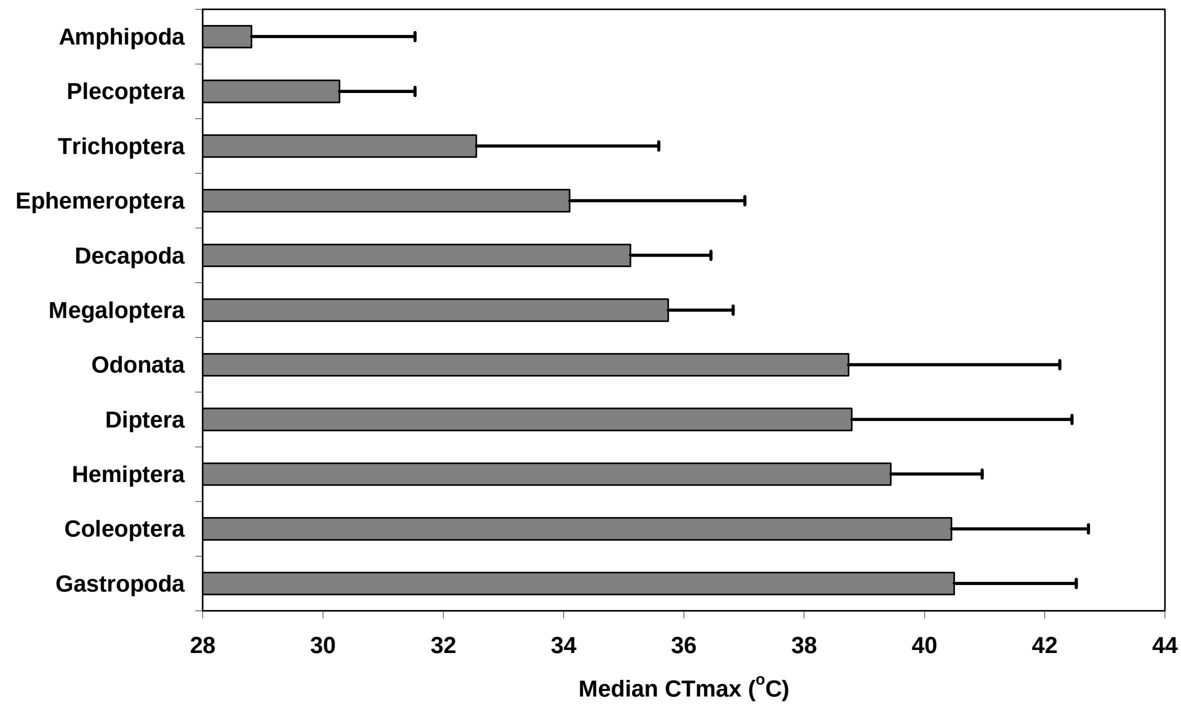


**Figure 4.16 Median CTmax (°C) with 90th percentiles for each group of aquatic macroinvertebrates acclimated to 17°C**

There was a significantly positive linear relationship between 96-h $LT_{50}$ (equivalent to Incipient Lethal Upper Temperature – ILUT) and median CTmax based on the ten species examined (Figure 4.17). The high $r^2$ value indicates that 78% of the variation in the ILUT is explained by variation in the CTmax. This confirms the efficacy of short term estimates to predict longer term measures.

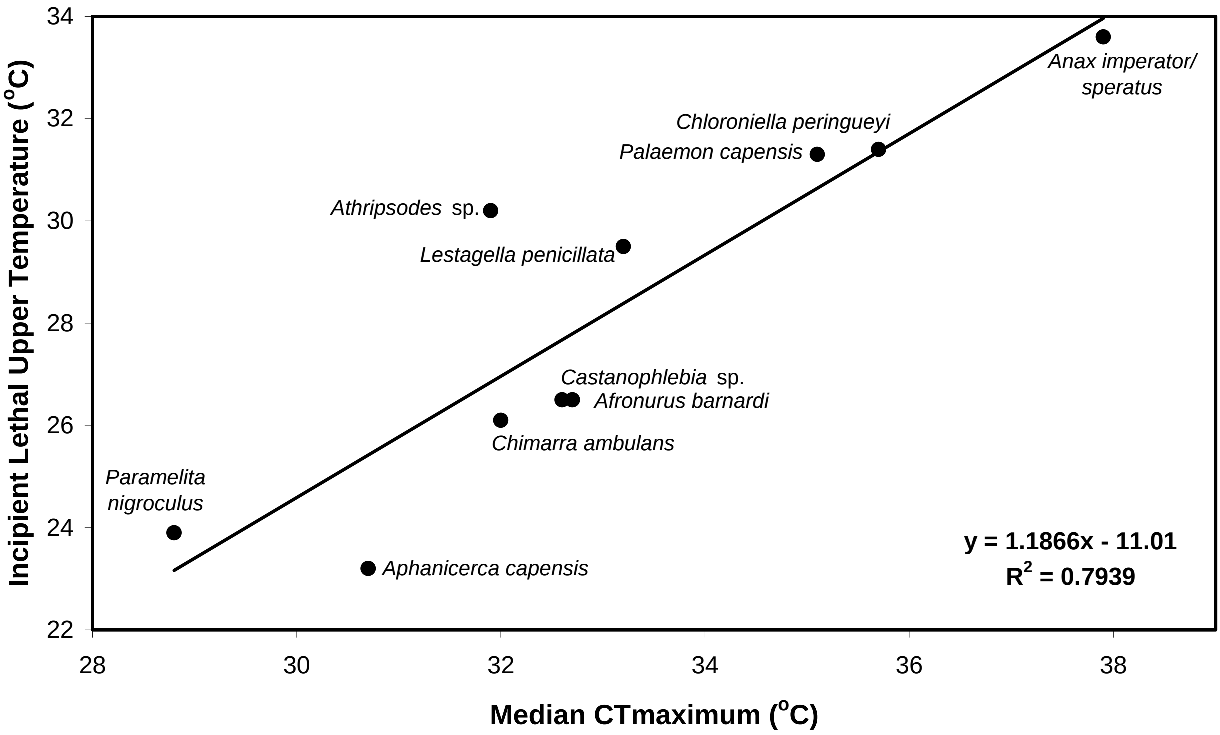


**Figure 4.17 Incipient Lethal Upper Temperature (ILUT) estimated for 96 hours expressed as a function of median CTmax temperatures**

## Conclusions and Recommendations

This study has provided some of the first data on the thermal tolerance of aquatic macroinvertebrates in South Africa. In undertaking this research it has become apparent that the issue of thermal sensitivity and the derivation of upper thermal limits for the setting of water temperature targets for river management is perhaps more complex than initially thought. Several potentially complicating factors such as thermal history, acclimation temperature and rate of temperature change may affect the resultant target water temperature thresholds. This study has identified those groups and families that are most thermally sensitive and most suited for use in thermal experiments. These include species within the families Paramelitidae, Notonemouridae, Heptageniidae, Leptophlebiidae, Telogonodidae and Philopotamidae.

The knowledge gained during the present study will form an excellent basis for future research. Several aspects that warrant further examination have been highlighted, including variation in thermal tolerance of genera and species within a family, variation of thermal tolerance within a species amongst sites with different thermal regimes and over different seasons; and the influence of acclimation temperature and rate of change of temperature on upper thermal limits.

The relationship between the short duration CTM and longer duration 96-$LT_{50}$ experiments has been established, which facilitates future experimental work based on CTM. Future $LT_{50}$ experiments should extend the time period to 10-days so that the extrapolated 10-day $LT_{50}$ data generated from existing 96-h $LT_{50}$ can be verified, and the relationship between CTmax and 10-d $LT_{50}$s established. Future evaluation of hourly *in situ* stream temperatures to identify periods of exceedance of 96-h and 10-day $LT_{50}$ temperatures will enable comparison of laboratory data with field conditions, and the setting of water quality criteria related to temperature.

When generating target thermal thresholds for South Africa, ambient stream temperatures, including daily mean, range, minimum and maximum values, which are likely to vary geographically within South Africa, will need to be considered. In this way thermal guidelines will account for intrinsic spatial variability of water temperatures. Similarly, by examining monthly variation in water temperature, temporal variability will be incorporated allowing for the generation of a temperature envelope. The challenge lies in linking the experimental data of upper thermal limits to *in situ* water temperature data in a manner that realistically reflects the likely consequences of a change in thermal regime on aquatic biota.

Ultimately biological data, including thermal tolerance data and other sub-lethal data such as temperatures for hatching or emergence, may then be applied to, for example, levels of threshold exceedance of moving averages of mean daily water temperatures (see Section 5.2-3), or used to examine the cumulative effects based on concepts such as growing degree days. In this way laboratory data on thermal tolerances may be employed for generating biological temperature thresholds, which will provide a useful management tool for maintaining stream temperatures in lotic ecosystems.

The linking of laboratory and sub-lethal data related to life history to *in situ* thermal data and the development of guidelines and scenarios is explored further in the section on Management (section 5.1 and 5.2).

# 5 MANAGEMENT

## 5.1 Temperature metrics – Guidelines to classify water temperature time series

**Related publication**

Rivers-Moore N.A., Mantel S. and Dallas H.F. 2010b. Water temperatures and the Reserve (WRC Project: K5/1799): Report on Temperature Metrics – Guidelines to classify water temperature time series into ecologically meaningful metrics. Report Number 1799/15 produced for the Water Research Commission. Institute for Water Research and The Freshwater Consulting Group.

### Introduction and aims

Understanding the predictability or cyclical constancy of water temperatures, and how this changes with downstream distance, is an important predictive tool in relating biotic responses to abiotic change. Human land use and development in stream catchments tend to homogenise and remove insulating and buffering processes (Poole et al. 2001). Among the earliest responses to stress are changes in species composition of small, rapidly reproducing species, because their rate of populaton growth and their small body size (surface area: volume ratio) translates into a small tolerance of thermal stress. It is critical to detect early signs of aquatic ecosystem degradation (Schindler 1987).

Temperature regimes can be summarized and quantified using statistics that describe distribution. The mean, median, maximum, minimum and variance can all be used to describe a temperature regime for a given length of stream over a given period of time. Measures of the time and location at which mean, maximum and minimum temperatures occur are useful as well (Poole et al. 2001). In perennial streams, summer maximum temperatures play a dominant role in the distribution of species (Dunham et al. 2003). "Factors to consider when reducing the variable summer temperature regime to simpler indices include i) the temperature threshold that reflects biological effects (usually a maximum but can be a minimum); ii) a temperature statistic within the amplitude of fluctuation (e.g. max, mean or min); iii) the averaging period that changes temperature exposure (e.g. hourly, daily or weekly)." (Sullivan et al. 2000). Temperature criteria typically have two key elements: a threshold temperature that signals when adverse biological responses are likely to occur, and an averaging period that indexes the duration of exposure likely to trigger that response (Sullivan et al. 2000).

The approach proposed is similar to the "range of variability" concept developed by Richter *et al*. (1997) and used in measuring change in flow time series (timing, duration, magnitude and frequency of events) (Richter *et al*. 1996). It also aims to include defining water temperature states or classes suitable for Colwell's (1974) predictability indices, where predictability is made up of constancy (state within a given class is the same for all seasons) and contingency (state differs between seasons but the pattern is the same for all years). This index is regarded as perhaps the most focused existing methods of describing aspects of temporal fluctuation, but the value of the indices depends in part on the class intervals used to define states of the phenomena (Archer 2000). The objective of this section was to develop guidelines for classifying water temperature time series into ecologically meaningful metrics.

## Methods

Data were obtained from 90 sites (47 loggers in the Eastern Cape and 43 loggers in the Western Cape) to illustrate the natural range of water temperature variability. Hourly data were processed as follows:

- data cleanup to identify and remove dates when the logger was outside of the water and was, therefore, recording air temperatures.
- hourly temperature data were converted to daily averages.
- daily averages were used to calculate temperature metrics based on those defined for Indicators of Hydrological Alteration by Richter et al. (1996, 1997).
- monthly metrics were determined in cases where less than half a month's data were missing (Table 5.1).
- seasonal metrics were determined in cases where fewer than one month's data were missing for the season.

The usefulness of metrics in describing thermal differences between sites was assessed using a number of approaches. Firstly, metrics were calculated for five sites along the Great Fish River in the Eastern Cape) with differing degrees of perenniality, and metrics were tabled. Next, a Principal Components Analysis (PCA) was conducted using *Statistica* (StatSoft, 2003), to demonstrate how metrics can be used to differentiate sites using ordination techniques. Here, data from 10 sites in two river systems in the Eastern Cape (Keurbooms and Kowie Rivers), for which all 37 metrics could be calculated, were used in the analysis. A worked example of a Group 5 metric is provided to demonstrate its usefulness in incorporating a spatial component into the metrics.

**Table 5.1 Temperature metrics for describing "Indicators of Thermal Alteration" (After Richter et al. 1996).**

| | | |
|---|---|---|
| **Annual descriptive statistics** | | Mean ± SD of annual temperature |
| | | Annual coefficient of variability |
| | | Predictability (Colwell 1974) |
| | | Annual range (mean ± SD) |
| | | Annual coefficient of variation of range |
| | | Degree days (annual cumulative) |
| | | Summer range (December-February) |
| | | Winter range (June-August) |
| **Group 1** | Monthly **magnitudes** (measure of central tendency) | Oct-Sept mean temperatures |
| **Group 2** | Magnitude and duration of annual extreme water temperature conditions (based on moving averages of different durations) | 1, 3, 7, 30 & 90-day minima |
| | | 1, 3, 7, 30 & 90-day maxima |
| **Group 3** | Timing – Julian date of maximum and minimum metrics (thermal triggers) | Date of onset of longest event below minimum threshold |
| | | Date of onset of longest event above maximum threshold |
| **Group 4** | Frequency and duration (successive days of event above or below a threshold) | Min. temp threshold count & duration |
| | | Max. temp threshold count & duration |
| **Group 5** | Rate and frequency of a change in conditions (i.e. the abruptness and number of intra annual cycles of environmental variation) | Rate of change in daily range with downstream distance |
| | | Rate of change in maximum temp threshold exceedance with downstream distance |

Because of the short period of data analyses, concerns regarding the inter-annual stability (or relevance) of the metrics were addressed through comparisons of metrics for corresponding periods in a successive year (1 January -31 December 2010) at twenty-two selected sites for the Eastern and Western Cape provinces. Metrics which did not show high (> 0.8) degrees of correlation with other metrics were compared for statistically significant differences using two approaches. In the first approach, we compared metrics between 2009 and 2010 using paired Student's *t*-tests for dependant samples ($p < 0.05$) (Statsoft 2003). For the second approach, we compared site pairs using a Principal Components Analysis (McCune and Mefford 1999) (centered correlation matrix). Absolute differences between co-ordinates of sites for paired years for Axis 1 and Axis 2 were calculated, and mean deviances from zero difference were tested using a Student's *t*-test.

## Summary of Major Results

***Patterns in calculated metrics***

The range of the temperature values for the calculated metrics for the 90 sites (Table 5.2) showed large variations in recorded values.

***Variation in metrics with perenniality (Great Fish River, Eastern Cape)***

Metrics of five sites along the Great Fish River were compared (Figure 5.1). The site Q2Fish-Pool is a non-perennial section in the upper reach of the river and the site Q2Fish-Saltp is further downstream along the same tributary of the Fish River. Site Q1GBrak-Saltp3 is on the main stream of the Fish River just above the junction of the tributary while Q9Fish-mort is further down the main stem of the Fish River. Q9Fish-Trum is located in a section affected by regulation that is discussed under the next case study. Temperature metrics for Q2Fish-Pool, the non-perennial site, reveal lower monthly, seasonal and annual temperatures, possibly because of its location in the uppermost reaches relative to the three perennial, unregulated sites that are further downstream (Table 5.3 and Figure 5.2). The temperature range (both summer and annual) for Q2Fish-pool is, however, 1.5 to 3 times that of the other sites (Table 5.3).

***Classification of sites using Principal Components Analysis***

The correlation values of temperature metrics with the PCA axes 1 and 2 indicated that Axis 1 was negatively correlated with all CV metrics (annual, seasonal and monthly, except annual CV of temperature range), SD of mean annual temperature, and with Julian date and absolute minimum temperature (a thermal trigger to aquatic organisms). Axis 1 was also positively correlated with mean annual temperature, Colwell's Predictability index, and with minimum and mean temperature metrics for 3 of the seasons (spring, autumn and winter),

**Table 5.2 Range of selected temperature metrics (°C) for 90 sites collected during the study in the Eastern and Western Cape.**

| | *N* | *Minimum* | *Maximum* |
|---|---|---|---|
| Annual Temperature (Mean) | 90 | 12.13 | 24.96 |
| SD of Mean Annual Temperature | 90 | 0.84 | 5.01 |
| CV of Mean Annual Temperature | 90 | 0.05 | 0.28 |
| Annual Range (mean) | 90 | 0.94 | 10.15 |
| SD of Annual Range | 90 | 0.41 | 4.19 |
| CV of Annual range | 90 | 0.21 | 0.97 |
| Summer Range | 86 | 1.26 | 12.55 |
| Winter Range | 18 | 0.48 | 5.83 |
| Spring Temp, Mean (Sep-Nov) | 70 | 14.05 | 25.41 |
| Summer Temp, Mean (Dec-Feb) | 88 | 12.48 | 26.13 |
| Autumn Temp, Mean (Mar-May) | 82 | 11.69 | 23.46 |
| Winter Temp, Mean (Jun-Aug) | 18 | 9.94 | 17.88 |
| October mean temperature | 48 | 14.42 | 20.71 |
| November mean temperature | 52 | 14.93 | 22.72 |
| December mean temperature | 87 | 14.32 | 26.13 |
| January mean temperature | 86 | 11.94 | 25.90 |
| February mean temperature | 88 | 11.63 | 26.37 |
| March mean temperature | 84 | 11.55 | 25.68 |
| April mean temperature | 82 | 11.83 | 22.24 |
| May mean temperature | 57 | 10.25 | 19.46 |
| June mean temperature | 43 | 8.20 | 18.55 |
| July mean temperature | 18 | 9.04 | 17.66 |
| August mean temperature | 18 | 10.13 | 17.44 |
| September mean temperature | 20 | 10.42 | 18.28 |

with minimum, maximum and mean metrics for the months May-Sept, with all minimum metrics for 1, 3, 7, 30 and 90 days, and with maximum temperature threshold count and duration. Axis 2 was negatively correlated with mean and SD of annual temperature range, mean temperature range for summer and winter, maximum temperature for three of the seasons (spring, summer, autumn), with mean and minimum summer temperature, with max temperature values for the months October to April, with mean temperature values for the months October to March, with maximum metrics for 3, 7, 30 and 90 days, and with minimum temperature threshold count and duration. Thus, PCA Axis 1 was primarily influenced by the metrics of the cooler months, while PCA Axis 2 is indicator of temperature highs, for the hotter summer months and for periods of time, and of temperature range metrics. The PCA produced clear site differences based on their thermal metrics, which made intuitive sense based on site knowledge. Thus for example, the five sites linked by arrows in Figure 5.3 follow a linear progression and are ranked in the same order as they follow downstream in the Kowie River. Cumulative % variance explained by the first two PCA axes was high (81%; Table 5.4) and the PCA graph with these two axes is displayed in Figure 5.3.

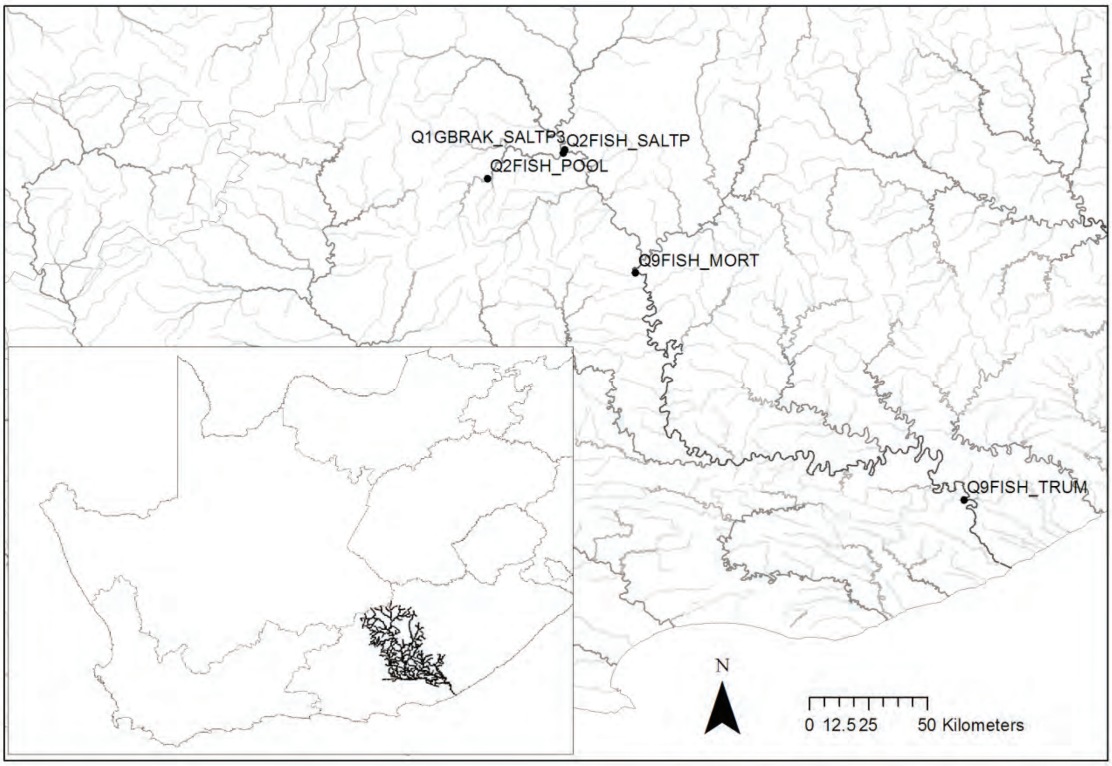


**Figure 5.1 Map showing the location of the five sites on the Great Fish River in the Eastern Cape referred to in the text.**

**Table 5.3 Selected annual and seasonal metrics for five sites located on the Great Fish River in the Eastern Cape.**

| | ***Q2Fish-Pool*** | ***Q2Fish-saltp*** | ***Q1Gbrak-saltp3*** | ***Q9Fish-Mort*** | ***Q9Fish-Trum*** |
|---|---|---|---|---|---|
| Dates of Temp for Analysis | 2 Oct 08 to 19 Jun 09 | 1 Oct 08 to 19 Jun 09 | 30 Sep 08 to 19 Jun 09 | 30 Sep 08 to 18 Jun 09 | 20 Aug 08 to 26 Jun 09 |
| Annual Temp (Mean) | 17.65 | 19.88 | 18.54 | 19.22 | 21.07 |
| SD of Mean Annual Temp | 3.65 | 3.13 | 3.81 | 3.38 | 4.08 |
| CV of Mean Annual Temp | 0.21 | 0.16 | 0.21 | 0.18 | 0.19 |
| Annual Range (mean) | 5.43 | 3.13 | 2.27 | 1.97 | 3.33 |
| SD of Annual Range | 3.49 | 2.66 | 0.84 | 0.72 | 1.66 |
| CV of Annual range | 0.64 | 0.85 | 0.37 | 0.36 | 0.50 |

| | Q2Fish-Pool | Q2Fish-saltp | Q1Gbrak-saltp3 | Q9Fish-Mort | Q9Fish-Trum |
|---|---|---|---|---|---|
| Summer Range | 7.74 | 4.96 | 2.36 | 2.26 | 4.05 |
| Spring Temp, Mean (CV) (Sep-Nov) | 15.65 (0.10) | 20.09 (0.08) | 18.24 (0.11) | 18.94 (0.10) | 20.06 (0.15) |
| Summer Temp, Mean (CV) (Dec-Feb) | 20.99 (0.06) | 22.52 (0.04) | 21.71 (0.04) | 22.18 (0.04) | 24.85 (0.06) |
| Autumn Temp, Mean (CV) (Mar-May) | 17.01 (0.20) | 18.42 (0.15) | 17.40 (0.20) | 17.83 (0.18) | 21.06 (0.17) |

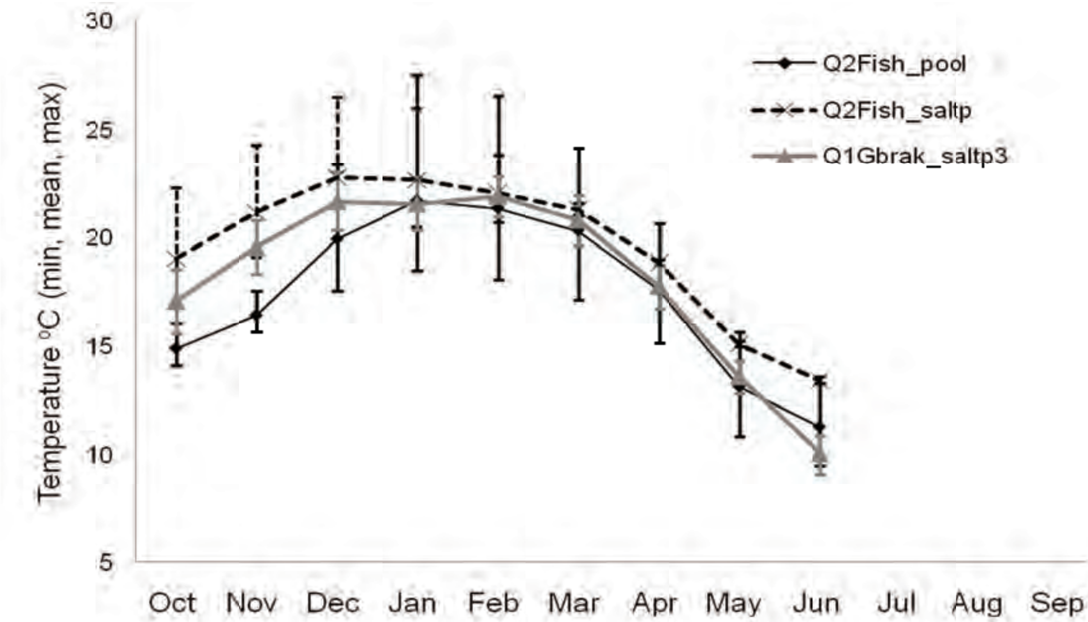


**Figure 5.2 Monthly temperature magnitudes (mean with range of min, max values recorded during the month) for three sites in the upper reaches of the Great River, Eastern Cape. The site Q2Fish-pool is a non-perennial site while the other 2 are perennial. Q9Fish-mort, a third perennial site located in the middle reaches, had mean values that were between those recorded for Q2Fish-saltp and Q1Gbrak-salpt3.**

**Table 5.4 Eigenvalues for the first four axes of the PCA using overall and Group 1 metrics**

| | Eigenvalue | % Total variance | Cumulative Eigenvalue | Cumulative % |
|---|---|---|---|---|
| 1 | 44.60 | 50.11 | 44.60 | 50.11 |
| 2 | 27.42 | 30.81 | 72.01 | 80.92 |
| 3 | 7.81 | 8.78 | 79.82 | 89.69 |
| 4 | 3.35 | 3.76 | 83.17 | 93.45 |

***Application of Group 5 Metric***

A demonstration of the applicability of one of the proposed Group 5 metrics is provided below (Figure 5.4). In this example, the number of times a 20°C threshold was exceeded by a 7-D moving average of maximum daily water temperatures was calculated for five sites on the Bloukrans/Kowie Rivers in the Eastern Cape. This example was chosen to illustrate the concept of applying the “Rate of change in maximum temperature threshold with downstream distance” metric.

It is assumed that an exceedance frequency of 120 days per annum of an MWAT threshold of 18°C (see Section 5.2) is the critical thermal threshold for species *A*. Below this value, thermal habitat quality is ideal, and above this threshold these conditions induce stress. The area of suitable thermal habitat can be estimated using integration of the quadratic equation for the curve. Using such an approach, the change in thermal habitat quality can be assessed for pre- and post-disturbance conditions, provided thermal thresholds are known for target species (Poole et al. 2001). Thus, a site 9.2 km downstream would have 100% ideal thermal habitat conditions at an MWAT exceedance of 120 days (Point A on Figure 5.4). Assuming a 10% increase in MWAT counts (Point B), ideal thermal habitat conditions at this same site would be reduced to 16.9% and this species would ultimately be forced to retreat upstream and considerably reduce its range. This is the type of approach and application for Group 5 of the proposed thermal metrics.

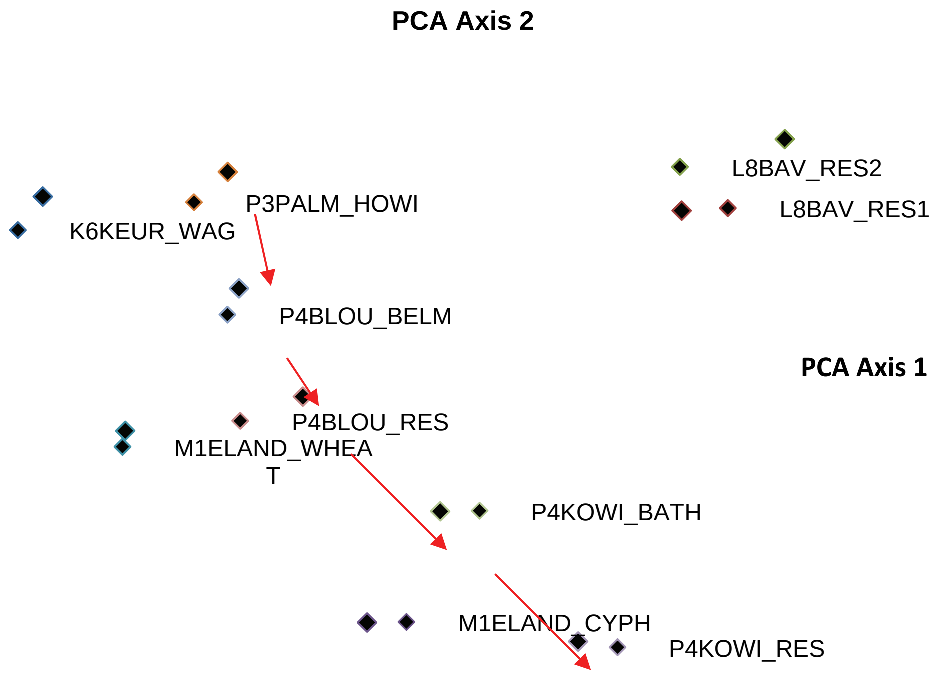


**Figure 5.3 Principal Component Analysis for all temperature metrics for 10 sites.**

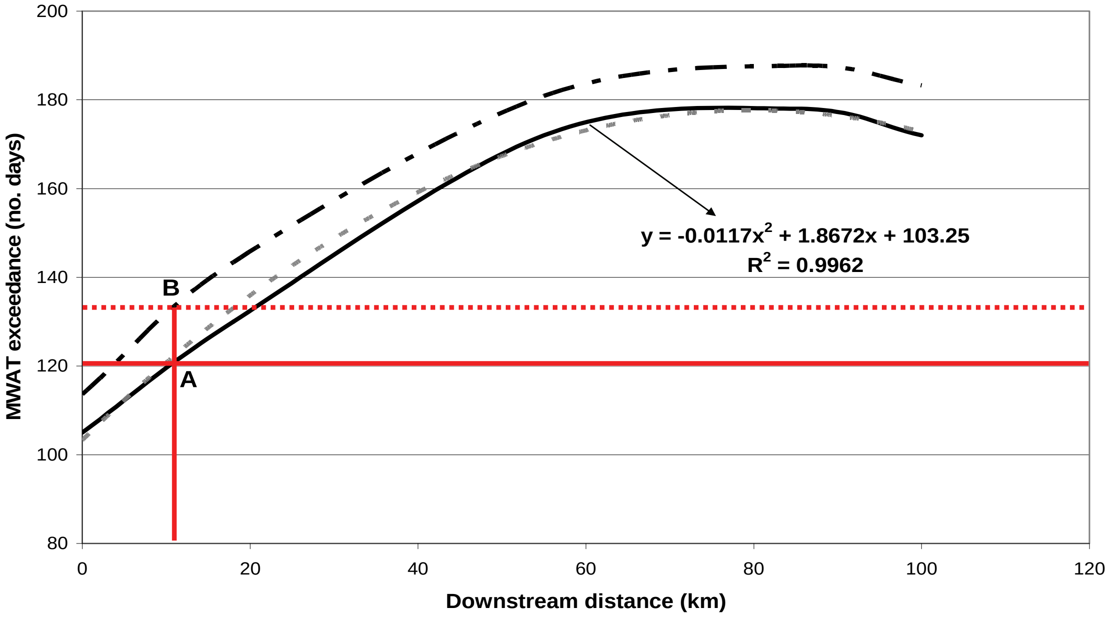


**Figure 5.4** Rate of change in maximum temperature threshold with downstream distance

### Stability of metrics between years

Ten of the fourteen metrics compared between years were not statistically different, while metrics reflecting variability (annual coefficient of variation and Colwell's predictability), together with June means and extreme maxima (maximum of the 7-D moving average of daily maxima), were significantly different between years (Table 5.5).

**Table 5.5 Sites from the Eastern and Western Cape provinces for which temperature metrics were compared from 2009 and 2010. Metrics which were significantly different between years (Student's *t*-test, 21 d.f., $p < 0.05$) are indicated by *.**

| | 2009 | | 2010 | |
|---|---|---|---|---|
| **Metric** | **Mean** | **SD** | **Mean** | **SD** |
| Annual Mean | 16.47 | 1.78 | 16.51 | 1.72 |
| Annual CV* | 23.27 | 6.30 | 24.58 | 4.92 |
| Predictability (Colwell)* | 0.65 | 0.08 | 0.61 | 0.07 |
| Min mean | 15.14 | 1.86 | 15.06 | 1.80 |
| Max mean | 18.11 | 1.96 | 18.31 | 1.94 |
| June* | 12.08 | 2.41 | 11.32 | 1.88 |
| July | 11.25 | 2.37 | 11.15 | 1.80 |
| Min_7 | 8.78 | 2.89 | 8.16 | 2.20 |
| Max_7* | 25.15 | 2.83 | 26.49 | 3.21 |
| Range (max) | 8.59 | 3.78 | 9.47 | 4.64 |
| Min T cnt | 96 | 58 | 101 | 53 |
| Max T cnt | 114 | 82 | 118 | 78 |
| Julian min | 151 | 54 | 155 | 40 |
| Julian max | 20 | 58 | 21 | 51 |

For the PCA, sites between years generally showed small positional shifts in metric space (Figure 5.5), although differences between paired years for Axes 1 and 2 were not statistically different from zero (Student's *t*-test, $p < 0.05$; 21 d.f., Table 5.6). A notable exception was group 11 in the Baviaanskloof (L8Bav-Res2, which was in direct sunlight and possibly experienced limited periods of exposure). Cumulative variance explained by PC Axes 1 and 2 was 47.81 and 72.28% (eigenvectors not shown).

## Conclusions and Recommendations

Sites can be differentiated based on their thermal characteristics, described using metrics. Metrics between years remained relatively stable. Fuller confidence in the use of these metrics and the detection of system change or stress will only be achieved through ongoing monitoring to collect long-term water temperature data. Metrics provide a structured approach to disaggregating and describing water temperature time series over annual periods, and to compare inter-annual differences in thermal data at a single site. Such metrics are first and foremost a tool which has a number of applications. For example, the usefulness of metrics in classifying sites based on thermal characteristics has applications at a number of scales, from making distinctions between hydraulic biotopes (Section 3.2.1) to defining thermal regions (Section 3.2.2) for management actions. Linking metrics to temperature thresholds for selected macroinvertebrate taxa (as has been demonstrated in Sections 5.2-3), provide the capacity to define components of the thermal ecological Reserve, and to assess flow reduction and climate change impacts on biota.

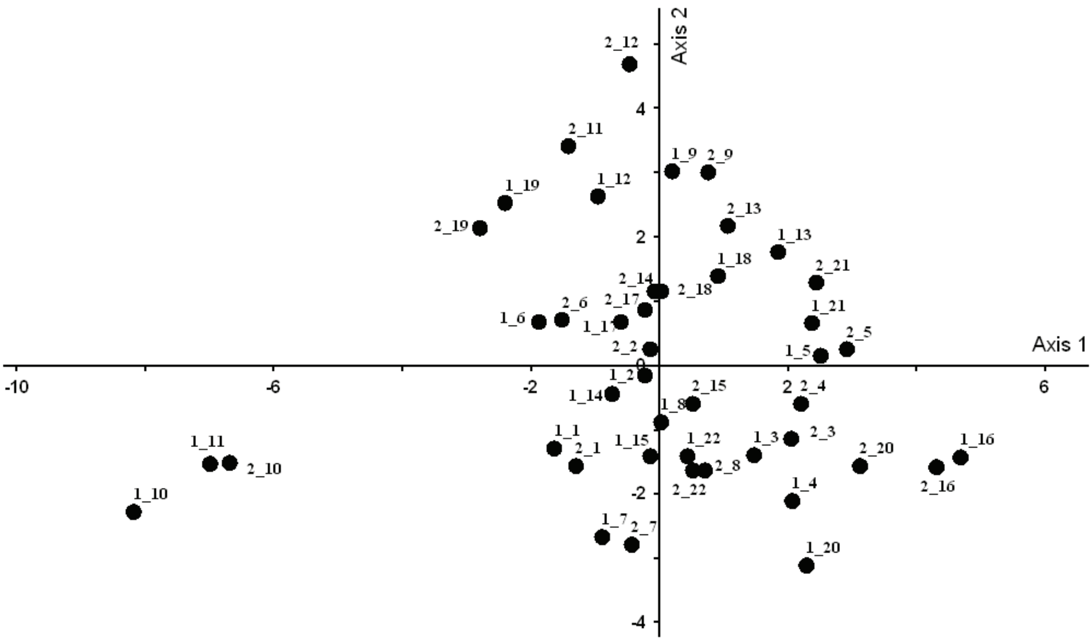


**Figure 5.5 PCA for paired years of data for twenty two sites in the Eastern and Western Cape provinces. Site codes are provided in Table 5.6 ( prefixes 1: 2009, 2: 2010).**

**Table 5.6 Differences in metric coordinate space for sites between years for axes 1 and 2 of a PCA**

| Site | Code | Diff-Axis1 | Diff-Axis2 |
|---|---|---|---|
| E1Dwar-Kraka | 1 | -0.33 | 0.26 |
| E1Rate-Beave | 2 | -0.10 | -0.41 |
| G2Eers-Jonke | 3 | -0.59 | -0.27 |
| H1Elan-Tunne | 4 | -0.14 | -1.50 |
| H1Witr-Monum | 5 | -0.41 | -0.11 |
| H4Bree-Lecha | 6 | -0.36 | -0.03 |
| K-4-Dwars | 7 | -0.45 | 0.12 |
| K3Keur-Monta | 8 | -0.69 | 0.75 |
| K6Keur-Outen2 | 9 | -0.56 | 0.01 |
| L8Bav-Res1 | 10 | -1.49 | -0.77 |
| L8Bav-Res2 | 11 | -5.58 | -4.94 |
| M1Eland-Cyph | 12 | -0.49 | -2.06 |
| N1Gats-Aasvoe | 13 | 0.78 | -0.40 |
| P4Blou-Res | 14 | -0.68 | -1.59 |
| Q1Gbrak-Dekeur | 15 | -0.68 | -0.80 |
| Q4Tark-Oak | 16 | 0.38 | 0.15 |
| Q8Kis-Cookh | 17 | -0.38 | -0.19 |
| Q9Balf-Wfall | 18 | 0.88 | 0.23 |
| Q9Fish-Trum | 19 | 0.39 | 0.40 |
| T4-RSE3 channel* | 20 | -0.83 | -1.54 |
| W7-1 Drakenstein* | 21 | -0.06 | -0.62 |
| W7-4 Kasteelskloof* | 22 | -0.08 | 0.23 |
| Mean ± SD | | -0.52 ± 1.25 | -0.59 ± 1.22 |

## 5.2 Ecological Reserve determination for water temperatures

| **<u>Related Publication</u>** |
| --- |
| Rivers-Moore NA, Dallas HF and Morris C 2011. Water temperatures and the Reserve (WRC Project: K5/1799): Ecological Reserve and Management Guidelines for River Water Temperatures for setting water temperature guidelines for the ecological Reserve in South Africa. Report Number 1799/23 produced for the Water Research Commission. Institute for Water Research and The Freshwater Consulting Group. |

### Introduction and aims

Underpinning the setting of environmental flows, which aim to maintain rivers at a targeted level of health, is the recognised need for managing water and land such that hydrologic regimes are not altered beyond agreed upon sustainability boundaries i.e. an allowable depletion or augmentation of baseline condition and variability (Richter 2009). Significant departures from these regimes results in a system change, and the Great Fish River, following on from an inter-basin transfer scheme where flows changed from non-perennial to perennial, is cited as a classic example of a permanently altered system (O'Keeffe and De Moor 1988).

In South Africa the concept of environmental flows began to be recognized in the 1980s, and this later evolved into the Ecological Reserve in the National Water Act, where water needs for maintenance of rivers was recognised (King et al. 2003). South Africa's Water Act is internationally regarded as one of the most advanced pieces of water legislation because of its framework can facilitate the integrated management of quantity and quality of water through the ecological Reserve. The South African National Water Act provides legal status to the water required to maintain ecological functioning of river systems, through the declaration of the "ecological Reserve" (Section 3), 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."*** (Republic of South Africa 1998). The determination process recognises determining the Reserve for all or part of a water resource.

Quantity and quality of water are closely linked but are typically treated separately, with water temperature typically classified under "quality". For the successful implementation of environmental flow management, both variability in discharge and temperature should be simultaneously considered (Jackson et al. 2007; Olden and Naiman 2010). 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 increased water temperatures.

While streamflow is perceived as a ‘master variable’ shaping many fundamental ecological characteristics of riverine ecosystems (Poff and Zimmerman 2010), temperature metrics have been shown to be as important as discharge metrics in explaining differences in invertebrate community structure (Jackson et a. 2007). Decreased values of ecological metrics typically mirror changes in flow and temperature metrics (Jackson et al. 2007; Poff and Zimmerman 2010), although it may be difficult to define generic/unambiguous quantitative relationships between flow alteration and ecological response, in part because alteration of flow regimes is typically confounded with other environmental factors (Poff and Zimmerman 2010).

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. Defining the temperature component for ecological flows (and in South Africa’s case, the ecological Reserve) is a two-stage process. In the first stage, a time series needs to be described using suitable metrics, while in the second stage, suitable thresholds need to be assigned to pertinent metrics, to give some measure of when conditions have not/are not being met.

A common approach to setting upper limits to temperature is based on known direct links to biological responses, which are often determined experimentally using thermal tolerance testing to generate acute or chronic temperature data (Nelitz et al. 2007). Thresholds can be either biological (cues) or statistical ($90^{th}/10^{th}$ percentiles). The physical tolerances of freshwater species to temperatures allow for the quantification of more targeted metrics, with absolute and cumulative temperatures both being important (Olden and Naiman 2010). For sub-lethal thresholds, the magnitude and duration of exposure to elevated temperatures is critical. A common index used is based on exceedance of a temperature threshold using a 7-D moving average of mean daily water temperatures (Sullivan et al. 2000; Nelitz et al. 2007; Null et al. 2010). Exceedance of a 7-D moving average of daily maximum water temperatures has been related to fish distributions in the Sabie Rivers (Rivers-Moore et al. 2005b) and blackfly outbreak probabilities (Rivers-Moore et al. 2008a). Typically, such magnitudes and durations are derived from hourly time series data which are summarized into daily statistics (mean, minimum and maximum) (Nelitz et al. 2007).

For water temperatures, the current best management practice in South Africa, where temperature time series data are available, is to calculate $10^{th}$ and $90^{th}$ percentile temperatures for each month (assume it is mean) for reference condition temperatures, and then calculate the absolute differences between reference and current temperatures for the $10^{th}$ and $90^{th}$ percentiles. Deviations from reference condition are assigned value on a six-point rating scale, which varies between no change to extreme change for absolute deviation thresholds of 2, 4, and > 5°C (DWAF 2008).

The aim of this section was to refine approaches in setting water temperature guidelines for the ecological reserve. We propose a refined approach based on the 7-D moving averages of mean, maximum and minimum daily temperatures, and exceedances.

## Methods

### *Defining the thermal ecological Reserve*

Hourly data for a full year (1 January to 31 December 2009) were converted to daily data (mean, minimum, maximum). Both disaggregating and agglomerative approaches were used in the analysis of these data. For the disaggregating approaches, daily time series for 1 January to 31 December 2009 were broken down into 37 metrics. The temperature metrics define statistics of a river's thermal regime with respect to magnitude of water temperatures, frequency, timing and duration of thermal events (Richter et al. 1996; Olden and Naiman 2010; Rivers-Moore et al. 2010b). These metrics where run for a subset of 84 sites which had reliable data.

For the agglomerative approaches, mean daily water temperatures were ranked from smallest to largest. Return intervals were calculated using cumulative frequencies based on the ranks (Equation 5.1). This was done for two sites, as a demonstration of the technique.

$$RI = \frac{1}{1 - R_{cum}} \qquad [5.1]$$

where $RI$ is return interval, and $R_{cum}$ is cumulative rank.

The thermal ecological Reserve was based on the metrics, and allows detection of change at a number of levels of sophistication. Thermal change can be detected using metrics which focus on changes in frequency, timing, duration and magnitude of thermal events. Included in this are monthly means (thus building on the current existing approach) as well as monthly ranges. Biological thresholds were incorporated into the Reserve assessment. Acute stress thresholds and exceedance thereof were related to 7-D moving averages of daily maxima and minima. In the case of a minimum threshold, the focus was on the thermal regime dropping below this, and provides scope for assessing water which is too cold, as a result of hypolimnetic releases from impoundments. A chronic thermal stress threshold was also incorporated, which was based on exceedance of a 7-D moving average of mean daily water temperatures. For these three thresholds, the Reserve can be assessed based on a number of frequency analysis criteria, including number and duration of exceedance events plus return intervals.

Our concern with these metrics as part of an ecological Reserve determination was that they are complex, and it was more desirable to provide a simpler approach where a river manager can easily answer the question "Is the temperature within acceptable limits?" For this approach, the logic behind the Sustainability Boundary Approach (Richter 2009) was incorporated. We incorporated natural variability together with a risk envelope (derived for a reference site) over an annual period of daily means plus daily ranges (i.e. daily minima and maxima) which were smoothed using 7-D moving averages to exclude thermal extremes but incorporate concepts of magnitude and duration. The confidence envelope was based on a percentage departure of the smoothed daily maxima and minima from the daily mean, from which a mean annual upper and lower departure was calculated together with a 95% confidence interval ($z = 1.96$). The 7-D moving average of daily mean water temperatures was used as the basis for calculating the upper and lower thermal envelope by multiplying these smoothed daily values with the upper 95% confidence intervals of mean departure from the daily mean.

***Setting of biological thermal thresholds***

Biological temperature thresholds for a number of taxa were derived experimentally under controlled laboratory conditions or through examination of recorded distribution of species in relation to water temperature ranges. Acute effects normally take the form of lethal upper or lower temperature, exceedance of which result in death after exposures ranging from minutes to 96 hours (Sullivan et al., 2000). The "Acute Stress envelope" is thus based on Incipient Lethal Temperatures (ILT) or $LT_{50}$ values, also referred to as Short-Term Maximum (STM, Armour, 1991) survival temperature. Temperatures causing thermal stress after longer exposures, ranging from weeks to months, are termed chronic temperature effects or "Chronic Stress envelopes". Endpoints of exposure to temperature over longer periods (chronic effects) are sub-lethal and may include growth, competitive interactions, change in behaviour, or disease (Sullivan et al., 2000). Temperature ranges defined by acute and chronic temperature effects are referred to as the zones of thermal resistance and tolerance (Jobling 1981). "Chronic Stress envelope" may be based on an averaging statistic such as 7-D moving average of mean daily water temperatures (MWAT) bounded by some measure of daily range (such as 7-D moving average of minimum and maximum, or 30-day moving average).

Brungs and Jones (1977) and Armour (1991) both describe methods for establishing an MWAT threshold using experimental temperature information (Equation 5.2).

$$MWAT = OT + \frac{(ILUT - OT)}{3} \qquad [5.2]$$

where OT is optimal temperature, and ILUT is incipient lethal upper temperature.

Since MWAT is highly dependent on OT, it was also estimated using two additional methods, thereby providing comparative estimates of OT. Both methods focused on a subset of sites where macroinvertebrate species, shown to be thermally sensitive (Dallas and Ketley 2011) were known to occur. The first method used the median water temperature calculated from hourly water temperature data at sites where the relevant taxon was recorded. The second used median water temperature calculated from spot water temperature measurements for sites in the Rivers Database (DWAF 2007) where the taxon was recorded. For this only upper catchment sites (Mountain Headwater Stream, Mountain stream, Transitional and Upper Foothill) in the Western Cape were used.

We define recurrence intervals for MWAT exceedance using Equation 5.3, and the probability of a threshold exceedance per annum using Equation 5.4 (Chow et al. 1988).

$$RI = \frac{N}{E - 1} \qquad [5.3]$$

$$p(1) = \frac{1}{RI} \qquad [5.4]$$

where *RI* is return interval, *N* is time period (years), *E* is the number of threshold exceedance events

***Selection of target taxa***

MWAT values were calculated for six species of aquatic macroinvertebrates to test the robustness of this approach, and its applicability for different taxa and rivers. Only one species was used in the paired site analysis below. Based on thermal tolerances of selected aquatic macroinvertebrates (Section 4.2), we selected the amphipod, *Paramelita nigroculus*; the notonemourid stonefly (Order Plecoptera), *Aphanicerca capensis*; three species of mayfly (Order Ephemeroptera), the heptagenid, *Afronurus barnardi*, the leptophlebid, *Castanophlebia* sp., the telagonodid, *Lestagella penicillata*; and the philopotamid caddisfly (Order Trichoptera), *Chimarra ambulans*. Amphipods are considered to be thermally sensitive (Davies et al. 2004) and several species occur in the Western Cape, with *Paramelita nigroculus* being one of the most common species occurring in mountain streams.

Plecoptera are represented by two families in South Africa, with one of them, the Notonemouridae, comprising 31 species in six genera (Stevens and Picker 2003). Plecopterans are one of the most important and dominant insect components of stream fauna (Wishart et al. 2003). Nymphs typically require cool, aerated waters, and their distribution is limited to fast-flowing reaches of high altitude streams (Wishart et al. 2003). Stoneflies may be important for detecting changes in lotic temperature regimes as they form an important part of the stream food web and have been found to be good indicators of water quality (Ernst et al. 1984). Water temperature has been shown to influence growth in stoneflies and temperature thresholds for growth in stoneflies is generally low (Brittain 1983). The family Notonemouridae is typically Gondwanan and is endemic to the Southern Hemisphere, with relatives of the same family on the other southern continents (Stevens and Picker 2003). *Aphanicerca capensis* occurs across the Western Cape, South Africa, from the Cederberg in the north, through Table Mountain and the mountains of the south-western parts of the Western Cape to the Tsitsikamma Mountains in the east (Stevens and Picker 2003). Nymphs are shredders of leaf detritus, a major source of carbon in streams (Stevens and Picker 2003).

Ephemeropterans include families and species, which vary in thermal sensitivity. Two genera and seven species of Heptageniidae have been recorded from southern Africa (Barber-James et al. 2010). Heptageniids are a characteristic family of foothill reaches of rivers and are grazers feeding on periphyton. The family Leptophlebiidae has a worldwide distribution and in the Afrotropics there are 16 genera. Nymphs generally occur in flowing water, and are found on rocks, gravel, woody debris or roots (Barber-James and Lugo-Ortiz 2003). They are variable in their feeding habits with scrapers, collector-gatherers and shredders represented. There are only two described species of *Castanophlebia*, *C. albicauda* and *C. calida*, which are endemic to southern Africa and are common in the fast-flowing waters of mountain streams throughout South Africa and Lesotho (Barber-James and Lugo-Ortiz 2003). Teloganodids are Gondwanan in origin and are represented by eight genera worldwide, of which four described genera occur in Africa, including the southern and south-western regions of the Cape (Barber-James and Lugo-Ortiz 2003; Barber-James et al. 2008). Nymphs are found on the underside of stones in cool, well-oxygenated, fast-flowing mountain streams, and feed on periphyton and fine detritus (Barber-James and Lugo-Ortiz 2003). They are sensitive to water quality impairment. There is only one species of *Lestagella* (*L. penicillata*) known from the Western Cape of South Africa (Barber-James and Lugo-Ortiz 2003).

Trichopterans are good indicators of the condition of aquatic ecosystems (De Moor and Ivanov 2008). Two genera of Philopotamid are known for southern Africa, *Chimarra* and *Thylakion* (formerly *Dolophilodes*). They are passive filter feeders, constructing elongated fine-meshed tubes and filtering-nets, suspended by the water-current, on the under-surfaces of submerged stones (De Moor and Scott 2003). Nymphs are mostly found in fast-flowing mountain streams. *C. ambulans* occurs in the Western Cape, Mpumalanga and KwaZulu-Natal.

***Modification of thermal regimes and paired site analysis***

A basic literature review was undertaken to establish some of the possible impacts on thermal regimes of different anthropogenic impacts (river regulation, water abstraction, inter-basin transfer, climate change). Where possible, impacts on specific metrics or components of the thermal regime were identified. These data were supplemented by analyses from selected sites from the Eastern and Western Cape, to determine orders of magnitudes of impacts in selected metrics.

A detailed analysis of two paired sites from the Western Cape (Wit River) and Holsloot River (downstream of the Steynskloof dam), which are both tributaries of the Breede River, was undertaken using the metrics to demonstrate application of the thermal ecological Reserve. The Wit River site was taken as the reference site. Sites were further characterised by their exceedance of the upper and MWAT thresholds using the telagonid mayfly, *Lestagella penicillata.* A ten percent deviance from any reference metric was regarded as falling outside of reference values.

## Summary of Major Results

***Biological temperature thresholds***

MWAT values obtained using ILUT data for six species using three methods for calculating OT, showed a maximum range in OT of 3.7$^{o}$C and MWAT of 2.5$^{o}$C (Table 5.7). Method 1 estimated the OT at 17.8$^{o}$C, where absolute minimum and maximum was 6.4°C and 29.2°C respectively. The resultant MWAT values varied from 18.7°C to 21.1°C. OT determined per taxon using method 2 varied from 14.1$^{o}$C to 14.3$^{o}$C, with MWAT ranging from 16.4°C to 18.8°C. OT calculated using Method 3 ranged from 15.0$^{o}$C to 16.6$^{o}$C, with MWAT ranging from16.9°C to 19.8°C. Of the three methods MWAT was lowest, and thus most conservative, using method 2, i.e. median values from actual water temperatures water temperatures, followed by method 3, i.e. median values from the Rivers database (DWAF 2007). MWAT using the midpoint was highest.

***Paired site analysis***

The relative contribution of heat inputs changes seasonally i.e. the impacts of modified drivers and insulators are not constant during the year (Webb et al. 2008). Human impacts change the amount (magnitude) or timing of the heat load delivered to the watercourse or by modifying the flow regime (Poole and Berman 2001). Additionally, streams differ in their sensitivity to modifications. Water temperatures at different times of the year may not be affected to the same extent by future climate change (Webb and Nobilis 1994 cited in Webb et al. 2008). Any reduction in the buffer efficiency leads to larger swings in cyclical response patters, resulting in higher maxima and lower minima (daily and seasonal) (Poole and Berman 2001). This is reflected in the proportion of time spent at different temperatures over a 24-h cycle at different times of the year.

**Table 5.7 Optimum Temperature (OT) calculated using three methods: Method 1 – OT Based on midway point of water temperature (WT) data recorded for 2 years at six sites; Method 2 – OT Based on WT data recorded for 2 years at six sites; and Method 3 – OT Based on Rivers Database data (DWAF 2007).**

| | | Method 1 | | Method 2 | | Method 3 | |
|---|---|---|---|---|---|---|---|
| **Species** | **ILUT** | **OT** | **MWAT** | **OT** | **MWAT** | **OT** | **MWAT** |
| *Aphanicerca capensis* | 20.6 | 17.8 | 18.7 | 14.3 | 16.4 | 15 | 16.9 |
| *Paramelita nigroculus* | 21.4 | 17.8 | 19.0 | 14.1 | 16.5 | 15.8 | 17.7 |
| *Afronurus barnardi* | 21.7 | 17.8 | 19.1 | 14.2 | 16.7 | 16.5 | 18.2 |
| *Chimarra ambulans* | 22.7 | 17.8 | 19.4 | 14.2 | 17.0 | 16.6 | 18.6 |
| *Castanophlebia* sp. | 23.4 | 17.8 | 19.7 | 14.3 | 17.3 | 16.4 | 18.7 |
| *Lestagella penicillata* | 27.8 | 17.8 | 21.1 | 14.3 | 18.8 | 15.8 | 19.8 |

It is necessary to be able to describe expected responses of river thermal regimes to anthropogenic influences (Poole and Berman 2001). Literature together with examples from the Eastern and Western Cape sites show that different anthropogenic impacts affect different components of thermal regimes. For example, low flows make rivers vulnerable to temperature fluctuations. Because of the extent to which they disrupt the river continuum, there is a need to develop a better understanding of dam-induced impacts (reset distances) and temperature metrics (e.g. impacts on 1-, 7- and 30-day minimum temperatures). Flow-temperature changes can lead to secondary temperature impacts, including de-synchronized life cycles of aquatic macroinvertebrates (Richter and Thomas 2007). More extensive information on impacts, temperature and biological responses is included in Rivers-Moore et al. (2011a).

Application of the ecological Reserve envelope method showed the approach to be both simple and effective. For the reference site, marked seasonality and variability in daily ranges was apparent. At this site, it is desirable that an MWAT threshold of 19°C is exceeded during the summer months (Figure 5.6). At the site below the impoundment, there is clear evidence of reduced seasonality and daily ranges during the summer months, and homogenisation of the thermal regime (Figure 5.7).

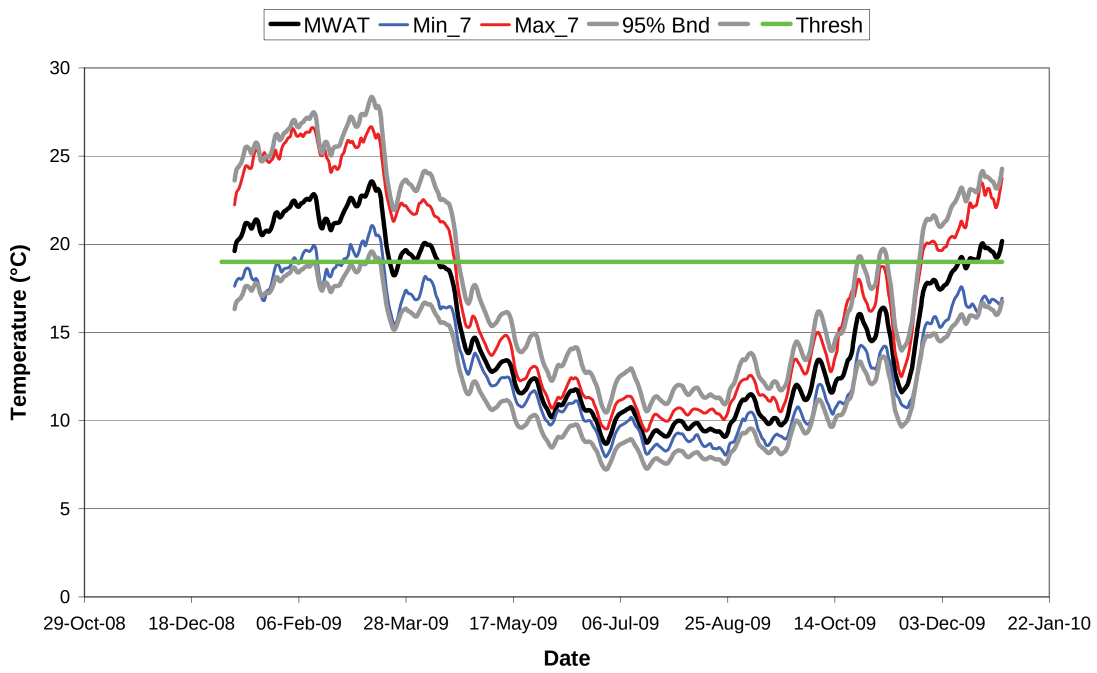


**Figure 5.6 Ecological Reserve envelope for the Wit River (Reference) site, showing 7-D moving averages for daily mean, minimum and maximum temperatures falling within 95% confidence intervals**

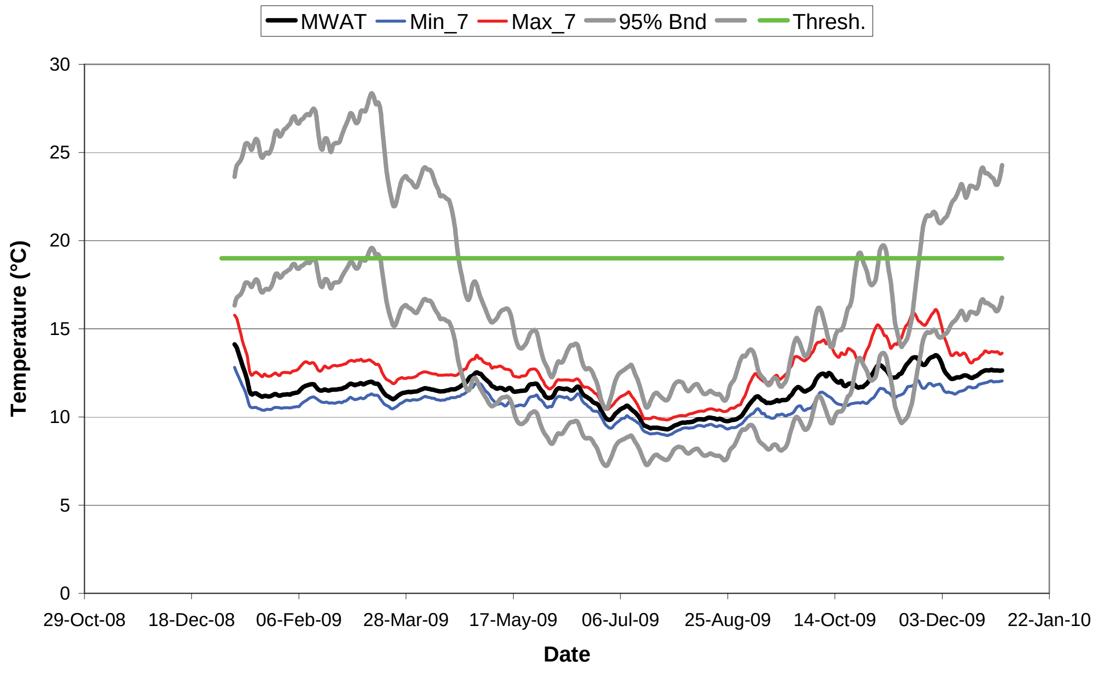


**Figure 5.7 Ecological Reserve envelope for the Holsloot River (downstream of impoundment) site, showing 7-D moving averages for daily mean, minimum and maximum temperatures falling below 95% confidence intervals for summer months.**

There were no exceedances of the MWAT threshold. Only the winter monthly temperatures (mean, maximum and minimum) values were within reference (i.e. within 10%) of the reference site values (Table 5.8 and see Text Box 1 for further explanations). Return intervals for the entire annual thermal time series were completely altered below the dam (Figure 5.8). Choosing threshold values for 7-D moving averages of mean, maximum and minimum daily values and comparing return interval statistics for exceedance events of these thresholds clearly showed that in this particular case, the water at the downstream site was too cold (Table 5.9).

**Table 5.8 Comparison of metrics for Wit (Reference site) and Holsloot River (Assessed/impacted site) sites**

| | | Assessed Site | | Reference Site | | |
|---|---|---|---|---|---|---|
| | | Mean/ Value | CV | Mean/ Value | CV | Thermal Reserve Exceedance? |
| **Annual descriptive statistics** | Annual Mean | 11.48 | | 15.13 | | Outside Reference |
| | Annual sd | 1.08 | | 4.80 | | Outside Reference |
| | Annual CV | 9.39 | | 31.71 | | Outside Reference |
| | Predictability (Colwell 1974) | 0.79 | | 0.56 | | Outside Reference |
| | Annual range (mean) | 1.87 | | 3.60 | | Outside Reference |
| | SD of annual range | 0.98 | | 2.31 | | Outside Reference |
| | CV of annual range | 52.03 | | 64.28 | | Outside Reference |
| | Max. range | 4.82 | | 11.29 | | Outside Reference |
| | Degree days | 541.42 | | 1871.88 | | Outside Reference |
| | Mean of annual minima | 10.74 | | 13.46 | | Outside Reference |
| | Mean of annual maxima | 12.62 | | 17.06 | | Outside Reference |
| **Group 1: Monthly magnitudes** | Jan | 11.94 | 10.54 | 20.90 | 6.54 | Outside Reference |
| **(Measures of central tendency)** | Feb | 11.63 | 2.40 | 21.95 | 7.61 | Outside Reference |
| | Mar | 11.55 | 3.48 | 20.83 | 10.83 | Outside Reference |
| | Apr | 11.83 | 3.77 | 17.13 | 15.60 | Outside Reference |
| | May | 11.58 | 3.93 | 12.20 | 10.24 | Within Reference |
| Mean monthly averages | June | 11.01 | 7.28 | 10.46 | 12.01 | Within Reference |
| | Jul | 9.82 | 5.83 | 9.80 | 9.59 | Within Reference |
| | Aug | 9.86 | 2.35 | 9.83 | 11.05 | Within Reference |
| | Sep | 11.13 | 3.98 | 10.67 | 10.99 | Within Reference |
| | Oct | 12.11 | 4.69 | 13.69 | 13.13 | Outside Reference |
| | Nov | 12.98 | 5.04 | 15.28 | 17.95 | Outside Reference |
| | Dec | 12.42 | 2.35 | 19.25 | 7.04 | Outside Reference |
| | Jan | 11.03 | 9.14 | 18.01 | 7.06 | Outside Reference |
| | Feb | 10.87 | 2.40 | 19.05 | 9.95 | Outside Reference |

| | | Assessed Site | | Reference Site | | |
|---|---|---|---|---|---|---|
| | | Mean/ Value | CV | Mean/ Value | CV | Thermal Reserve Exceedance? |
| | Mar | 10.93 | 3.30 | 18.27 | 12.76 | Outside Reference |
| | Apr | 11.29 | 3.81 | 15.51 | 15.54 | Outside Reference |
| | May | 10.84 | 5.19 | 11.40 | 10.30 | Within Reference |
| Mean monthly minima | June | 10.55 | 7.94 | 9.87 | 13.48 | Within Reference |
| | Jul | 9.40 | 5.19 | 9.11 | 11.48 | Within Reference |
| | Aug | 9.46 | 2.67 | 8.86 | 11.81 | Within Reference |
| | Sep | 10.22 | 4.78 | 9.54 | 11.94 | Within Reference |
| | Oct | 10.96 | 3.69 | 12.08 | 14.30 | Within Reference |
| | Nov | 11.62 | 5.02 | 13.50 | 17.07 | Outside Reference |
| | Dec | 11.74 | 2.85 | 16.68 | 7.44 | Outside Reference |
| | Jan | 13.23 | 11.44 | 24.41 | 8.28 | Outside Reference |
| | Feb | 12.91 | 3.38 | 25.56 | 6.79 | Outside Reference |
| | Mar | 12.58 | 4.73 | 23.78 | 10.22 | Outside Reference |
| | Apr | 12.71 | 4.15 | 18.98 | 16.79 | Outside Reference |
| | May | 12.47 | 4.72 | 13.11 | 11.49 | Within Reference |
| Mean monthly maxima | June | 11.55 | 6.41 | 11.12 | 10.93 | Within Reference |
| | Jul | 10.38 | 6.78 | 10.50 | 8.78 | Within Reference |
| | Aug | 10.41 | 2.91 | 10.87 | 11.28 | Within Reference |
| | Sep | 12.59 | 5.79 | 11.92 | 12.19 | Within Reference |
| | Oct | 13.93 | 6.73 | 15.68 | 14.30 | Outside Reference |
| | Nov | 15.19 | 7.05 | 17.20 | 19.15 | Outside Reference |
| | Dec | 13.51 | 4.08 | 22.10 | 9.80 | Outside Reference |
| | Jan | 2.20 | 29.49 | 6.40 | 31.79 | Outside Reference |
| | Feb | 2.04 | 20.33 | 6.51 | 24.34 | Outside Reference |
| | Mar | 1.65 | 25.54 | 5.51 | 21.29 | Outside Reference |
| | Apr | 1.43 | 23.47 | 3.47 | 44.81 | Outside Reference |

| | | Assessed Site Mean/ Value | Assessed Site CV | Reference Site Mean/ Value | Reference Site CV | Thermal Reserve Exceedance? |
|---|---|---|---|---|---|---|
| | May | 1.63 | 33.55 | 1.71 | 38.75 | Within Reference |
| Mean monthly ranges | June | 0.99 | 34.80 | 1.25 | 44.40 | Outside Reference |
| | Jul | 0.98 | 39.84 | 1.39 | 40.58 | Outside Reference |
| | Aug | 0.95 | 28.32 | 2.01 | 35.62 | Outside Reference |
| | Sep | 2.37 | 32.88 | 2.38 | 47.21 | Within Reference |
| | Oct | 2.97 | 30.76 | 3.60 | 54.75 | Outside Reference |
| | Nov | 3.57 | 27.24 | 3.71 | 40.25 | Within Reference |
| | Dec | 1.77 | 36.36 | 5.41 | 39.01 | Outside Reference |
| **Group 2: Magnitudes of extreme water temperature conditions** | Mean (7-D) | 14.12 | | 23.56 | | Outside Reference |
| | Minimum (7-D) | 8.95 | | 7.95 | | Outside Reference |
| | Minimum (30-D) | 9.19 | | 8.66 | | Within Reference |
| | Minimum (90-D) | 9.59 | | 8.96 | | Within Reference |
| | Maximum (7-D) | 16.10 | | 26.97 | | Outside Reference |
| | Maximum (30-D) | 13.25 | | 24.32 | | Outside Reference |
| | Max_90 | 14.38 | | 26.97 | | Outside Reference |
| **Group 3: Frequency and duration (successive days exceeding thresholds of extreme water temperature conditions)** | MWAT count | 0 | | 113 | | Outside Reference |
| | Min T count | 341 | | 172 | | Outside Reference |
| | Max T count | 0 | | 0 | | Within Reference |
| | MWAT duration | 0 | | 73 | | Outside Reference |
| | Min T duration | 185 | | 113 | | Outside Reference |
| | Max T duration | 0 | | 0 | | Within Reference |
| **Group 4: Timing Julian dates of onset of main exceedance of thresholds** | Julian MWAT | 0 | | 7 | | Outside Reference |
| | Julian minimum | 121 | | 135 | | Outside Reference |
| | Julian maximum | 0 | | 0 | | Within Reference |

**Text Box 1: Explanatory text of temperature metrics by Group**

**Group 2 – Magnitudes of extreme thermal conditions**

Magnitudes of extreme events at different temporal scales (weekly, monthly and seasonally) are calculated using moving averages of corresponding periods. Values indicate the maximum (for the mean and maximum daily water temperatures) or minimum (for minimum daily water temperatures) values for seven, 30 and 90 day moving averages based on annual water temperature time series.

**Group 3 – Frequency/duration of extreme thermal conditions**

This group of metrics calculates the number of exceedance events of the chronic stress threshold MWAT (based on the 7-D moving average of daily mean temperatures), and the acute stress thresholds (above a maximum value and below a minimum value) for the 7-D moving averages of minimum and maximum daily temperatures. Also calculated is the duration of the longest exceedance event of each threshold.

**Group 4 – Timing of exceedance events**

This metrics assesses timing of extreme thermal events, which has implications for biotic life cycle cues. Each metric in this group relates to the Julian date when the longest thermal threshold exceedance event begins.

**Table 5.9 Comparison of threshold exceedances (minimum, $LT_{50}$ and MWAT) of seven-day moving averages of daily minimum, maximum and mean water temperatures respectively**

| Ecological Reserve Statistics | Reference Site | Assessed Site | Thermal Reserve exceedance? |
|---|---|---|---|
| **Maximum ($LT_{50}$) threshold (acute)** | **27.80** | | |
| **Minimum threshold (acute)** | **12.00** | | |
| **MWAT threshold (chronic)** | **19.00** | | |
| **No. years data** | **1** | | |
| **Record length (days)** | **359** | | |
| **Statistics for MWAT temperature threshold** | | | |
| Total no. days above threshold | 113 | 0 | Outside Reference |
| Percent time exceedance | 31.48 | 0 | Outside Reference |
| Duration (Minimum no. days) | 3 | 0 | Outside Reference |
| Duration (Maximum no. days) | 73 | 0 | Outside Reference |
| Duration (Mean no. days) | 31.67 | 0 | Outside Reference |
| Duration (SD of Mean no. days) | 36.68 | 0 | Outside Reference |
| Duration (CV of Mean no. days) | 115.83 | 0 | Outside Reference |
| Number of exceedance events | 3 | 0 | Outside Reference |
| Recurrence interval of events | 119.67 | 0 | Outside Reference |
| p(exceedance) | 0.01 | 0 | Outside Reference |
| p(Exceedance in 5 years) | 0.04 | 0 | Outside Reference |
| **Statistics for maximum temperature ($LT_{50}$) threshold** | | | |
| Total no. days above threshold | 0 | 0 | Within Reference |
| Percent time exceedance | 0 | 0 | Within Reference |
| Duration (Minimum no. days) | 0 | 0 | Within Reference |
| Duration (Maximum no. days) | 0 | 0 | Within Reference |
| Duration (Mean no. days) | 0 | 0 | Within Reference |
| Duration (SD of Mean no. days) | 0 | 0 | Within Reference |
| Duration (CV of Mean no. days) | 0 | 0 | Within Reference |
| Number of exceedance events | 0 | 0 | Within Reference |
| Recurrence interval of events | 0 | 0 | Within Reference |
| p(exceedance) | 0 | 0 | Within Reference |
| p(Exceedance in 5 years) | 0 | 0 | Within Reference |
| **Statistics for minimum temperature threshold** | | | |
| Total no. days below threshold | 172 | 322 | Outside Reference |
| Percent time exceedance | 47.91 | 89.69 | Outside Reference |
| Duration (Minimum no. days) | 1 | 2 | Outside Reference |
| Duration (Maximum no. days) | 113 | 185 | Outside Reference |
| Duration (Mean no. days) | 13.15 | 37.67 | Outside Reference |
| Duration (SD of Mean no. days) | 30.30 | 64.78 | Outside Reference |
| Duration (CV of Mean no. days) | 230.36 | 171.98 | Outside Reference |
| Number of exceedance events | 13 | 9 | Outside Reference |
| Recurrence interval of events | 27.62 | 39.89 | Outside Reference |
| p(exceedance) | 0.04 | 0.03 | Outside Reference |
| p(Exceedance in 5 years) | 0.17 | 0.12 | Outside Reference |

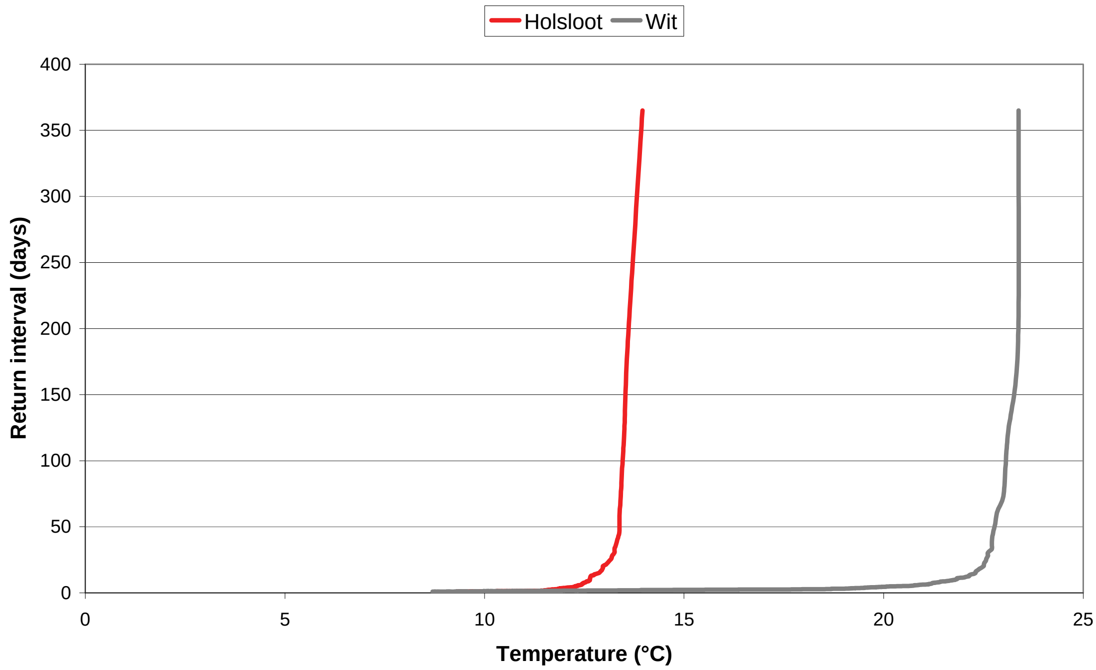


**Figure 5.8 Return intervals for mean daily water temperatures for Wit and Holsloot sites**

## Conclusions and Recommendations

Application of thermal ecological Reserve should ideally occur within a three step process – establish baseline conditions, measure departure from the baseline, and implement management action through recognising when thresholds have been exceeded (Ladson et al., 2006). A flowchart of the process of determining exceedance of the thermal Reserve is provided in Appendix 4.

***Establishing baseline conditions: Usefulness of metrics and thermal thresholds***

Reference condition describes a (statistical) distribution rather than a single absolute value i.e. temporal and spatial variability that is inherent in any measure chosen to represent the natural state of environmental systems (Stoddard et al. 2006). 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. In this study, we have used both agglomerative and disaggregating techniques to describe thermal regimes based on descriptive statistics and their "shape". We have successfully demonstrated that hourly temperature data from sites that cover a range of ecoregions and longitudinal zones can be disaggregated into metrics that describe the magnitude, frequency, timing and duration of thermal events. Such metrics were useful in deriving thermal regions (Section 3.2.2) for the Western and Eastern Cape provinces. These regions were based on thermal signatures representing baseline thermal conditions, and consequently each region can be associated with thermal metrics that describe the baseline condition.

Such metrics (Section 5.1), together with field data and laboratory studies, enabled the calculation of chronic and acute biological temperature thresholds, which, when exceeded are likely to lead to thermal stress, for six species of aquatic macroinvertebrates for the Eastern and Western Cape regions. The thresholds identified (17 to 21$^{o}$C) are not dissimilar to other thresholds previously identified for both fish and aquatic macroinvertebrate. For example, several studies, which examined the reproductive biology of fishes in South Africa, have shown that temperatures between 18 and 19°C trigger spawning of several of South Africa's indigenous fish species (Dallas 2008). Our thresholds were also close to the 15 to 19°C biological threshold range of Nelitz et al. (2007). In the absence of actual data, an 18 to 20°C is a reasonable biological threshold to use when using thermal metrics that describe magnitude and duration of thermal events.

All of these metrics and thresholds only begin to form a holistic management approach once incorporated into a spatial framework. Understanding natural patterns of flows in time and space has become fundamental to the assessment and management of environmental water allocations (Kennard et al. 2010, and citing others). It is critical to identify thermal regime types using classification to systematically arrange streams into types which are most similar in terms of their thermal regime. Environmental flow allocations, scenario analyses and risk analysis can only be undertaken after this has been achieved.

***Measuring departure from baseline conditions***

To apply this approach, the expected baseline conditions must be identified to distinguish between natural fluctuations versus changes due to human activities. There are few examples to guide analyses of changes in stream temperature variability following human activities (Jackson et al. 2007; Nelitz et al. 2007). Spot measurements of water temperatures at the time of biotic sampling are unlikely to yield satisfactory statistical relationships linking biotic response to habitat disturbance, and it is more useful to use daily time series of flows and water temperatures preceding a collection event as part of the analyses of aquatic communities. Discharge and thermal variability in a two-week period preceding sample collection have been shown to explain more aquatic invertebrate community variability than a broad suite of environmental conditions for a regulated versus an adjacent non-regulated river in Scotland (Jackson et al. 2007 p. 665).

It is often difficult to implement complicated environmental flow prescriptions that mimic natural flow variability (Richter 2009). One option is to recognise system change (departure from baseline conditions) when the “shape” of the regime curves change. The sustainability boundary approach (SBA) recognises the impact of too little and too much flow, and recognises the value of flow signatures (magnitude, duration), but fosters a precautionary approach by only focusing on magnitude, thus reducing scientific uncertainties because there is no need to know every aspect of the regime. The SBA is also robust regarding climate change, because the SBA boundaries (and allowable percentage departures) continue to apply (Richer 2009). Using a similar paradigm, our approach defines smoothed daily mean water temperatures plus daily ranges which can immediately be seen to fall within a “management envelope” of acceptable thermal conditions. Implicit in this approach is natural thermal variability.

Changes to air temperature and precipitation are shifting timing and volume of streamflows, potentially affecting aquatic communities (Wenger et al. 2010). In order to understand when and how thermal changes may be occurring, may require more detailed assessment of metrics. This could also be achieved based on combinations of metrics and return interval curves (regime “shape”). In the absence of actual data to derive metrics, it may be necessary to model them. Wenger et al. (2010) were able to successfully model several hydrological metrics. The range of success varied in correlations between predicted and observed data for various metrics, with mean annual flows beings predicted with good accuracy, and predictions for sites with strong groundwater influence were less accurate (Wenger et al. 2010). Similarly, thermal metrics were modelled with a range of success (Rivers-Moore et al. 2010d).

For the temporal component, the emphasis becomes one of detecting trend(s) over time within the different thermal types, with departure from baseline conditions and potential triggers of management intervention through exceedance of defendable thresholds. Such an approach would require monitoring at selected sites, and annual updates of the temperature metrics to check for unwanted trends.

***Management actions***

“Management of scarce water resources needs to be based on sound science that supports the development of environmental flow standards at the regional scale” (Kennard et al. 2010 p. 171). Proper management of flows has to encompass the entire catchment, such that improving the flow regime and water quality may require changes in land use patterns too (Nilsson and Renöfält 2008). It is critical that management of aquatic ecosystems aims to preserve as much system variability as possible 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).

There is no credible 'rule of thumb' for defining the amount of water that should remain in a river to satisfy environmental flow needs. While there is generally reasonable knowledge of links between ecological and societal consequences of hydrological alteration, the decisions on the acceptability of these consequences involves complex trade-offs among human values and benefits. These are typically arrived at following dialogue with stakeholders. Richter (2009) lists various reasons for bad implementation of environmental flows, including, *inter alia*, low priority given to environmental flows where other competing water requirements take precedence.

As a consequence of the general dearth 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/environmental flow studies. Null et al. (2010) call for better management of water quantity and quality using flow and water temperature simulations to evaluate potential restoration alternatives. In the absence of continuous observed data, models are important for defining baseline conditions, although there should be follow-up monitoring to verify the simulated data (Richter 2009). Models also enhance the capacity for scientists and managers to work together to "experiment" with different management actions under current and future potential climatic scenarios.

Assessing the success of management actions (for example, dam re-operation, where the release mechanisms are re-designed to release volumes of water to meet environmental flow requirements at the desired water temperature) is not a hit-and-miss process, but should occur by formulating hypotheses, and then testing these by implementing changes and tracking ecosystem responses, and continually refining through adaptive management (Richter and Thomas 2007). Given that impoundments are an easy option for achieving water needs in a catchment, and that South Africa is set to embark on a new phase of dam building, an urgent research requirement is for a better understanding of dam-induced impacts (reset distances) and temperature metrics (e.g. impacts on 1, 7 and 30-day minimum temperatures). Paradoxically, the best strategies for improving habitat (flows) may not necessarily be good for water temperatures (Olden and Naiman 2010).

## 5.3 Scenario modelling of selected species

**Related Publication**

Rivers-Moore NA, Dallas HF, and Ross-Gillespie V 2011. Water temperatures and the Reserve (WRC Project: K5/1799): Scenario modelling. Report Number 1799/25 produced for the Water Research Commission. Institute for Water Research and The Freshwater Consulting Group.

### Introduction and aims

Water temperature, together with flow volumes, is a master variable driving river ecosystems. Body weight, fecundity and water temperatures are intimately linked (Vannote and Sweeney 1980), and therefore, from both conservation and pest management perspectives, it is useful to model potential interactions between these variables under different thermal scenarios. It is clear that flow/thermal alteration is associated with environmental change, and that the risk of change increases with increasing magnitude of alteration. For example, changes in magnitude typically lead to stabilised flows (loss of extreme high and/or low flows) with impacts showing as loss of sensitive species, loss of diversity, changed community structure (Poff and Zimmerman 2010). From the perspective of pest species, homogenization of river systems can lead to ideal conditions for a pest species to dominate. In South Africa's Great Fish River, homogenisation of flow and water temperature variability, and constant flows elevated from natural conditions, have created ideal conditions for regular outbreaks of the pest blackfly, *Simulium chutteri*. These levels of outbreaks are not only a function of higher flows, but also since this species is multivoltine and generation times are a function of water temperature, outbreaks are likely to increase with global warming.

While uncertainty is inherent in modelling, such approaches can provide reasonable estimates of current and potential flows and temperatures. Simulation modelling is useful in highlighting preferred restoration alternatives (Null et al. 2010). Scenario analyses allow the cost-effectiveness of alternative temperature reduction strategies to be assessed (Seedang et al. 2008). Estimates of each potential action are required to evaluate cost-effectiveness of management options, which could be expressed as average cost per degree Celsius of water temperature of reduction (Seedang et al. 2008).

Linking biotic response to thermal regimes is well established in crop modelling and terrestrial pest modelling. Heat units (or degree days) are linked to the accumulated heat necessary for an organism's development, sometimes referred to as physiological time. An organism will develop where there is an accumulation of mean temperatures above a certain lower threshold value (below which development doesn't take place), and below an upper developmental threshold (above which growth remains static or declines). The degree-day concept assumes a linear relationship between growth and temperature, and does not take into account diurnal temperature range. One example is the orchard/apple pest species *Cyclia pomonella*, the codling moth, where 11.1 and 34.4°C are lower and upper thermal thresholds, and the total number of degree days to complete a life cycle is 603 degrees (Schulze and Maharaj 2007). Concomitantly, the probability of a species occurrence can be statistically modelled using binary regression techniques, and is based on species presence/absence data together with exceedances of an established physiological thermal threshold for this species.

The aim of this section was to examine potential impacts of temperature changes on selected aquatic organisms using scenario analyses by linking biotic response to thermal triggers using agglomerative techniques (duration curves, cumulative degree days).

**Methods**

***Selection of taxa for scenario analyses***

We selected two aquatic macroinvertebrate taxa for this study, the first being a univoltine, stenothermic ephemeropteran species of conservation importance, while the second species was a multivoltine dipteran pest species. Ephemeropterans include families and species that vary in thermal sensitivity. Teloganodids are Gondwanan in origin and are represented by eight genera worldwide, of which four described genera occur in Africa, including the southern and south-western regions of the Cape (Barber-James and Lugo-Ortiz 2003). Nymphs are found on the underside of stones in clean, fast-flowing mountain streams, and feed on periphyton and fine detritus (Barber-James and Lugo-Ortiz 2003). They are sensitive to water quality impairment and are generally found in cool, well-oxygenated upper reaches of rivers. There is only one species of *Lestagella* (*L. penicillata*) known from the Western Cape of South Africa (Barber-James and Lugo-Ortiz 2003).

*Simulium chutteri* (Diptera: Simuliidae) is a pest species found in several larger river systems in South Africa (O'Keeffe and De Moor 1988). Outbreaks of blood-sucking females of this species became pestilential and problematic to, *inter alia*, sheep farmers following construction of impoundments on rivers where this species naturally occurred historically. The key alternative life-history trait that enabled this was the adaptation of its seven-instar larval stage, to successfully exploit growth and development during cooler thermal regimes in conjunction with increased winter flow volumes following impounded flow regulation. This allowed the larvae to largely avoid predator pressure and with slower growing larger semaphoronts leads to greater fecundity and autogenous development of eggs resulting in rapid population growth in spring (De Moor, 1982a, 1989, 1992a, 1992b, 1994; Rivers-Moore et al., 2007, 2008a; Rivers-Moore and De Moor, 2008).

***Collection of thermal data***

Hourly water temperatures for a full year (1 January to 31 December 2009), collected using Hobo® Tidbit v2 data loggers, were used from sites corresponding with the geographical focus regions of this study (Figure 5.9). Data were converted to daily mean data, and modelling was based on time series of 7-D moving averages of daily means.

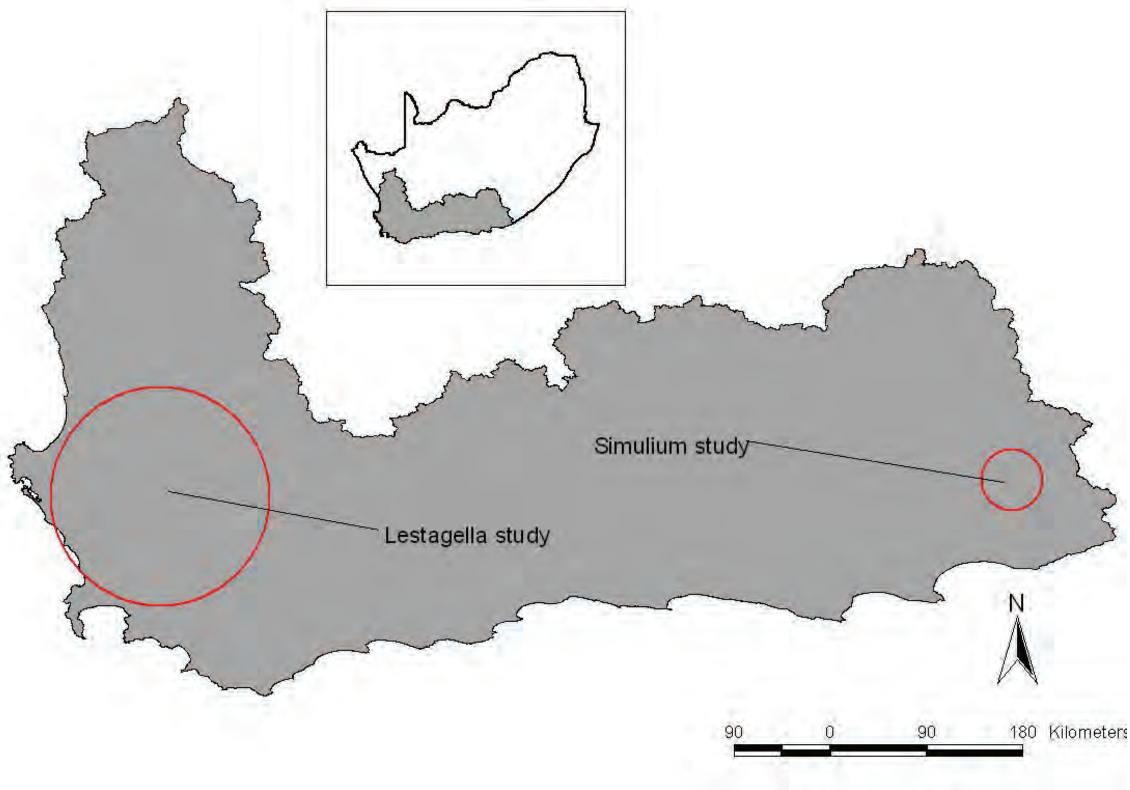


**Figure 5.9 Map of study area, showing regional focus of models from the two selected data**

***Linking thermal regimes to taxonomic response***

For *L. penicillata*, biological temperature thresholds ($LT_{50}$ and critical thermal maxima – Dallas and Rivers-Moore, 2011) were derived experimentally under controlled laboratory conditions and through examination of recorded distribution of species in relation to water temperature ranges. Presence and confirmed absence data from 31 sites in the Western Cape were collated with thermal data from these sites. Frequency and durations of exceedance of a chronic biological stress threshold (MWAT = Maximum Weekly Allowable Temperature that should not be exceeded) were calculated based on the 7-D moving average of mean daily water temperatures. A logistic regression model was used to calculate the probability of occurrence of this species, and to develop an MWAT threshold to estimate breeding success (R Development Core Team 2004).

The thermal model for *L. penicillata* was a spreadsheet model which calculated the number of days to hatching, and the probability of breeding failure. Model assumptions were:

- Model is flow-independent
- Hatching occurs above 5$^{o}$C
- Linear relationship assumed between temperature and hatching
- Breeding failure is caused by prolonged thermal stress

Model input was mean daily water temperatures, from which cumulative degree days above a critical threshold (5°C) and a 7-D moving average were calculated. The probability of egg hatching was calculated as the cumulative number of degree days succeeding a hatching cue divided by the total number of degree days required for hatching. Thermal stress of breeding adults was based on the frequency and duration of the exceedance of the 7-D moving average of mean daily water temperatures by the MWAT threshold. Successive exceedance of ≥ 30 days of the MWAT threshold constituted a breeding failure.

*Simulium chutteri* outbreaks modelled for a single site over a fifty-year period. An earlier model predicting blackfly outbreak probabilities identified the refinement of including generation times as a function of water temperature (Rivers-Moore *et al*. 2008a). The duration of the aquatic stage of *S. chutteri* is 12 to 45 days, and is inversely related to mean daily water temperature (De Moor 1982b, 1989). For example, a development time of 12 days yields two generations within a 24-day period versus one generation if water temperatures are lower and result in an aquatic stage of 24 days. The second generation will, however, be exponentially larger than the first generation, with concomitantly greater fecundity having major implications for outbreak severity. We estimated the number of life cycle completion events per annum under current (post IBT), pre-IBT (inter-basin transfer), and 2°C increased water temperatures for each flow scenario. Severity of the outbreak was qualitatively estimated using an exponential multiplier to an initial arbitrary starting population.  As with the earlier model, completion of the aquatic stage was a joint function of flows being above a lower threshold (0.05 $m^3s^{-1}$) and reaching the degree day threshold.

***Spatial mapping***

Two complementary modelling approaches were used to link thermal changes to changes in the spatial distribution of *L. penicillata*.  For the first approach, the presence of *L. penicillata* was linked to the thermal regions (Section 3.3.2).  Distributions of this species were modelled using the thermal regions, and compared with a second approach of a distribution map derived from surrogate environmental variables using Biomapper (Hirzel *et al*., 2002; 2007).  Environmental variables used were the same as those used by Rivers-Moore et al. (2011a) to define the thermal regions.  Raster images of eight environmental variables (altitude; mean annual air temperature; rainfall – May, June, November and annual; longitude and latitude; highest stream order per quaternary catchment) at a 200 m grid size for the Eastern and Western Cape provinces were used in conjunction with a presence map ($n$ = 21) of *L. penicillata*, with image processing undertaken using suitable raster-based software (Clark Labs., 2009).  We used ecological niche factor analysis, with factors selected using a broken stick model (Hirzel *et al*., 2007).  For the scenario modelling, we modelled change in distribution based on one of a number of potential future scenarios, viz. current temperatures and a 2°C increase in water temperatures using the thermal regions where *L. penicillata* was expected to occur.

**Summary of Major Results**

Critical thermal thresholds for *L. penicillata* and *S. chutteri* are provided in Table 5.10. Plotting relative abundances of *L. penicillata* against the annual frequency of exceedance of an MWAT threshold of 21.1°C showed that a likely critical thermal threshold for this species was approximately 60 (Figure 5.10).  Assuming that a relative abundance of ≤ 200 is a marginal population, a logistic regression model ($p < 0.001$) was useful in showing that the probability of occurrence of this species drops sharply from 100% to 0% for an MWAT exceedance of 55-59 (Equation 5.5).  Sites where MWAT exceedances were in excess of 60 days typically also showed exceedance of this threshold for ≥ 30 successive days.

For the model, time to hatching was estimated at a site where relative abundances of *L. penicillata* were known to be high under current thermal conditions, and for two scenarios (2°C warmer and cooler).  Days to hatching ranged from 47 to 30 under cooler and warmer conditions than under present conditions, where hatching was modelled to take 35 days based on a 350 degree day hatching threshold (Figure 5.11).  Based on the same three scenarios, the probability of breeding success was estimated at a thermally suitable and a thermally marginal site (MWAT = 0 and 54 respectively), where an MWAT exceedance of ≥ 30 successive days was assumed to lead to a breeding failure. At the optimal site, *L. penicillata* did not experience thermal stress under current conditions, and experience a brief period of thermal stress under a 2°C potential; increase. Conversely, at the thermally marginal site, *L. penicillata* experienced thermal stress twice under current thermal conditions, while a 2°C increase resulted in breeding failure (Figure 5.12).

$$p = \frac{e^{403.79-7.08MWAT}}{1+e^{403.79-7.08MWAT}} \quad [5.5]$$

**Table 5.10 Critical thermal information for *L. penicillata* and *S. chutteri.* Empty cells indicate data not relevant to the models and/or where data were unavailable but did not impact on modelling**

| Thermal threshold | *L. penicillata* | *S. chutteri* |
|---|---|---|
| Emergence period | October-December | |
| Life cycle | Univoltine; 11-13 months | Multivoltine; 12-45 days |
| Lower thermal limit for development | 5 | 10* |
| Upper thermal limit for development | 25 | |
| Hatching duration | 70 days @ 10°C | |
| Degree days | 304-350 | |
| IULT | 27.8 | |
| OT | 17.8 | |
| MWAT | 21.1 | |

*Pupation fails below this value

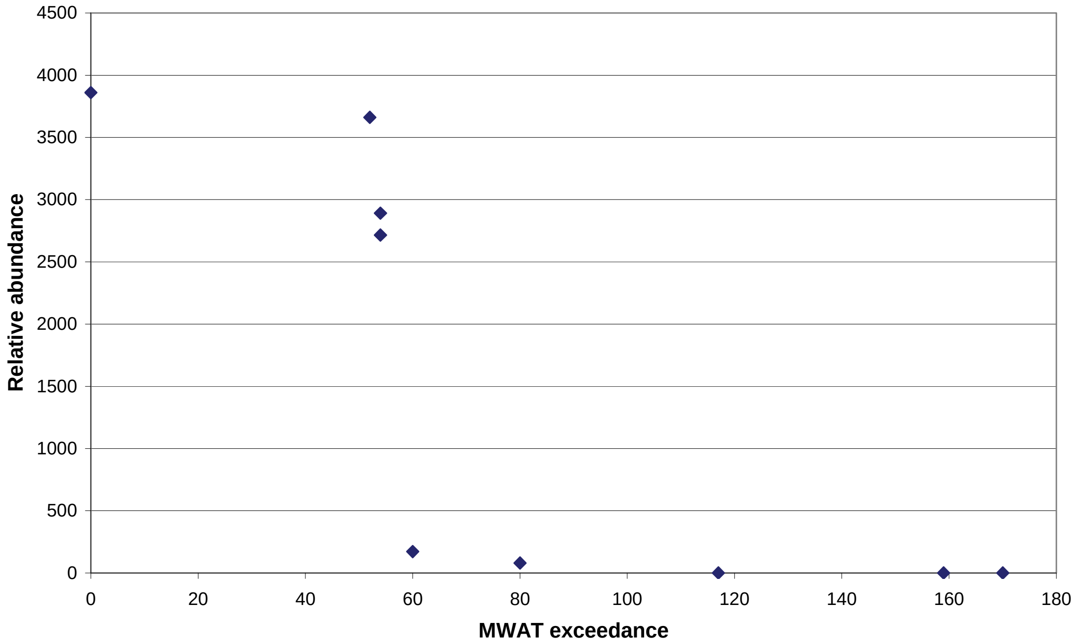


**Figure 5.10 Relative abundances of *L. penicillata* versus annual frequency of MWAT exceedance for nine sites in the Western Cape**

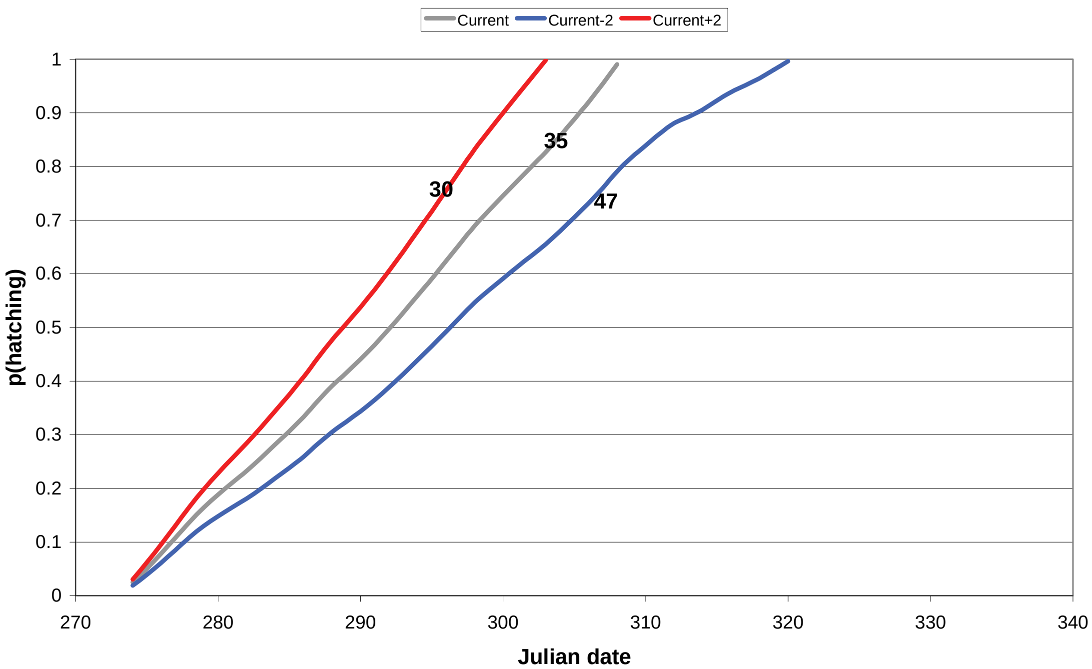


**Figure 5.11 Probability of hatching under current (i.e. observed), and 2°C increased and decreased thermal regimes (constant of 2°C added to mean daily water temperatures), of *L. penicillata* at a Western Cape site where relative abundances were high**

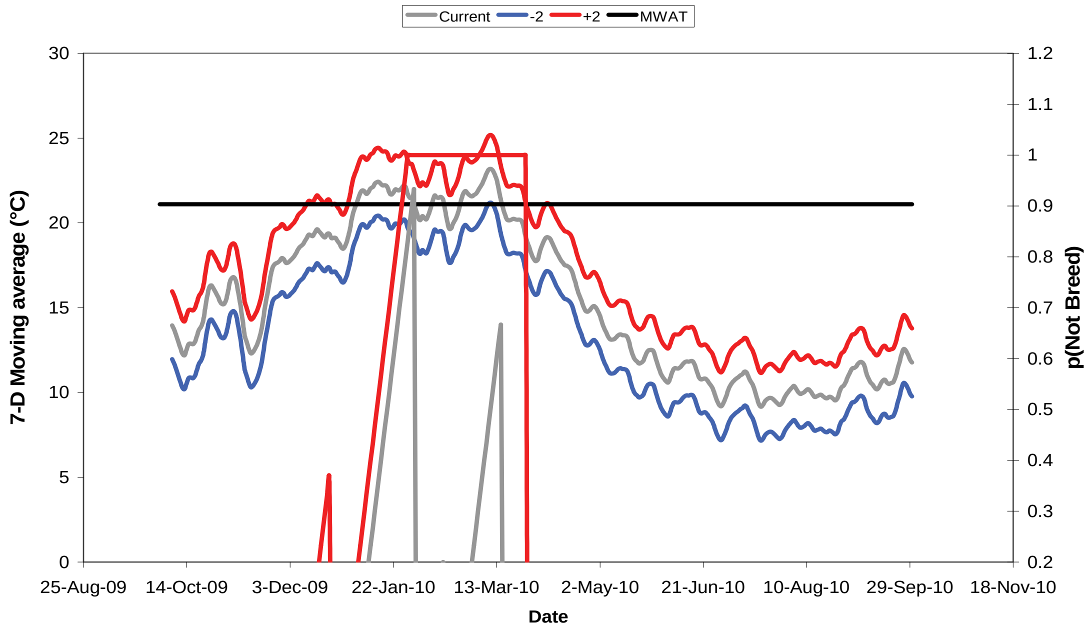


**Figure 5.12 Probability of breeding failure under current, and 2°C increased and decreased thermal regimes, of *L. penicillata* at a Western Cape site where relative abundances were low and thermal habitat was marginal**

Plotting confirmed presence and absence of *L. penicillata* with the thermal regions defined for the Eastern and Western Cape (Rivers-Moore et al. 2011a) showed a clear association of *L. penicillata* with thermal groups 3, 10 and 11 (see Section 3.3.2 for thermal regions). Because of the inaccuracies in the predictions of the classification and regression tree (CART) analyses used to derive the thermal map, absolute correlations could not be made, and rather broad patterns apply. Calculating mean annual MWAT exceedance values per thermal region showed that regions 3, 4 and 5 had exceedances of zero, while region 10 was in the range of 55 to 60. Regions 1, 2, 11, 12 and 13 were all in excess of 70 to 100 times per annum. From these values, regions 3, 4, 5 and 10 represent thermally suitable habitat for *L. penicillata* under current climatic conditions, with region 10 being the most vulnerable to increased water temperatures. Under a potential 2°C warming scenario, thermally suitable habitat was reduced by 30%, with region 10 becoming unsuitable (Figure 5.13). These results were in broad agreement with the habitat suitability map produced under current climatic conditions. This map confirmed that *L. penicillata* already as a restricted range in the Western Cape, where much of its habitat is marginal (Figure 5.13). The best predictors of occurrence (eigenvectors in brackets) were altitude (442.18), latitude (72.82), longitude (19.83) and winter (June) rainfall (12.1), with additional variables November rainfall, mean annual air temperature and stream order contributing less. The overall model explained 96.6% of the presence data using two factors. A *ca*. 20 kilometre phase shift between potential habitat using thermal regions versus environmental surrogates was to be expected given the coarseness of the data.

The number of generations of *S. chutteri* per annum showed a marked difference between pre- and post-IBT conditions. Preceding the completion of the IBT, the number of generations varied markedly between years. Contrastingly, the number of generations between years succeeding completion of the IBT was more stable (Figure 5.14; Table 5.11). With the addition of a constant daily temperature increase of 2°C, the time series of the number of generations per annum mirrored that of current thermal conditions, although the increased number of generations post-IBT was more marked than for pre-IBT conditions. Translating the number of generations into relative population sizes using an exponential multiplier showed similar trends as for the number of generations (Table 5.11). Modelled population figures were approximately double those under current water temperatures but with natural (Pre-IBT) flows. It was predicted that population numbers under current flow conditions but with a 2°C increase in water temperatures will be twenty times that of pre-IBT, current water temperature conditions. What was further illustrated in this figure was that for pre-IBT conditions, population numbers were periodically reset to very low numbers, which recovered under more favourable flow conditions.

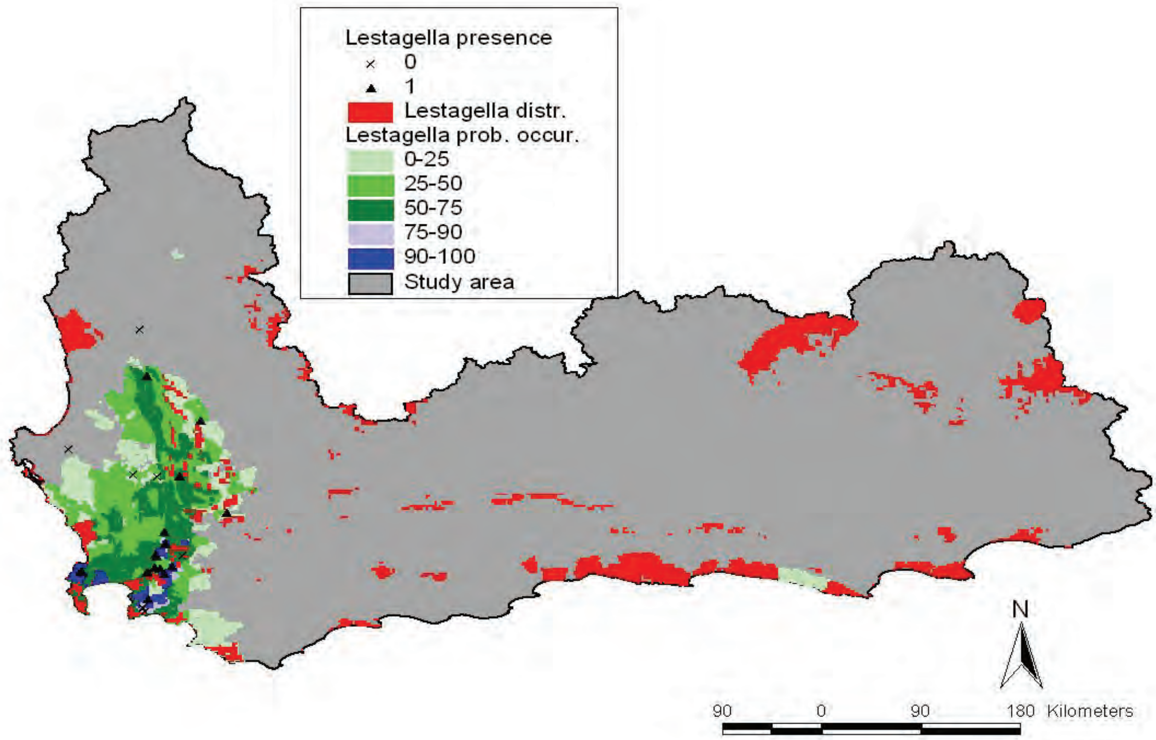


**Figure 5.13 Suitable habitat for *L. penicillata* under current climatic conditions using environmental surrogates as predictors of habitat to estimate probability of occurrence, and potentially thermally suitable regions for *L. penicillata* assuming 2°C water temperature increase of current conditions**

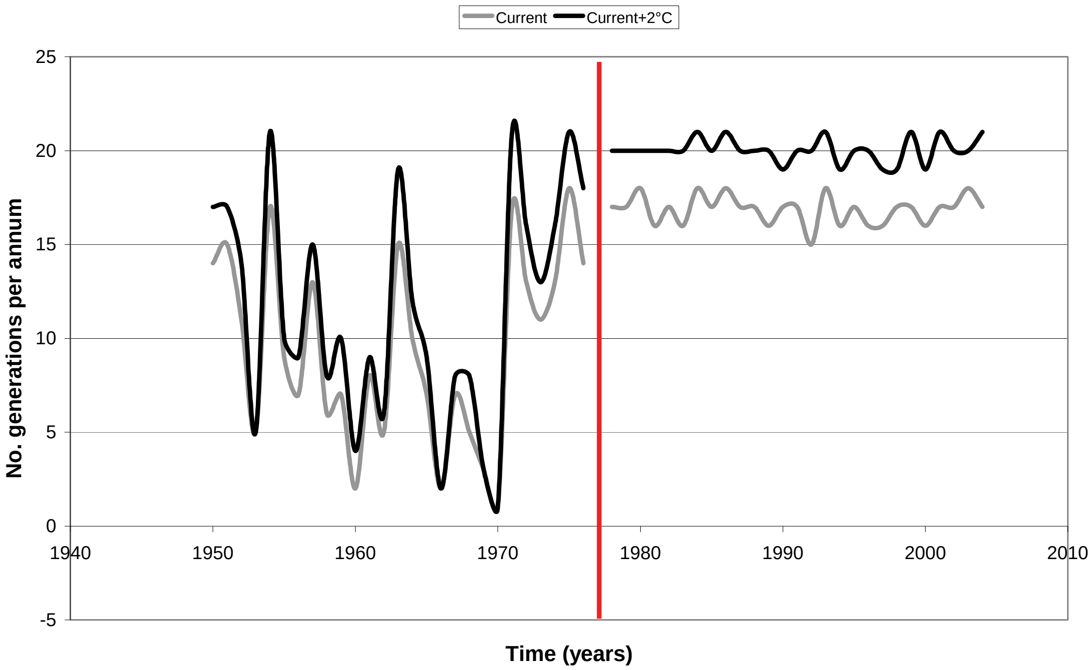


**Figure 5.14 Number of generations per annum of *S. chutteri* under different water temperature scenarios. Red line indicates the time at which an inter-basin transfer scheme became operational**

**Table 5.11 Mean (± standard deviations) monthly modelled population sizes and numbers of generations of *S. chutteri* for pre- and post- inter basin transfer scheme flow conditions under current and 2°C increased water temperature scenarios. Significant differences* between current (and natural flow conditions) are shown (one-tailed Student's *t*-test, $p < 0.05$; d.f. 323)**

| | | **Current conditions** | | **2°C increase in water temperatures** | |
|---|---|---|---|---|---|
| | | **Mean (±SD)** | **Coefficient of variation** | **Mean (±SD)** | **Coefficient of variation** |
| Population size. month$^{-1}$ | Pre-IBT | 3.74E+05± 4.67E+05 | 124.81 | 6.56E+05± 6.53E+05 | 99.51 |
| | Post-IBT | 8.12E+05± 4.93E+05 | 60.70 | 7.50E+06± 6.39E+07 | 852.01 |
| No. generations. month$^{-1}$ | Pre-IBT | 0.79±0.74 | 93.87 | 0.96±0.84 | 86.78 |
| | Post-IBT | 1.40±0.50 | 35.45 | 1.67*±0.48 | 28.60 |

## Conclusions and Recommendations

- It is possible to relate the absence of an aquatic macroinvertebrate species to thermal stress (magnitude and duration) using a suitable biological threshold value (MWAT)
- Thermal regions can be assigned MWAT exceedance values, and such spatial maps are useful in thermally suitable habitat for a chosen species
- Cold-adapted, stenothermal species have limited thermal habitat in the Eastern and Western Cape provinces. Under a 2°C increase in water temperatures, the range of these aquatic communities is likely to decrease further. Cold-adapted Gondwanaland relic species are likely to become increasingly vulnerable and range limited. A considerable portion of the habitat of *L. penicillata* occurs in thermally marginal areas, where slight increases in water temperatures could lead to non-viability of its populations.
- Reduced life cycle completion times will benefit multivoltine pest species such as *S. chutteri*, and outbreaks are likely to become more frequent and more severe under scenarios of increased water temperatures. Univoltine species are at a disadvantage under conditions of global warming. Aquatic macroinvertebrate communities could become increasingly dominated by warm water eurythermic, generalist species.

# 6 CONCLUSIONS AND RECOMMENDATIONS

The body of research in this project represents a considerable advancement in understanding thermal patterns in South African rivers, and how biota (individual species and aquatic macroinvertebrate communities) respond to thermal variability and stress. Understanding spatio-temporal thermal patterns in the Eastern and Western Cape provinces requires a multi-scale approach.

Linking biotic response to thermal drivers is naive if only mean temperature values alone are used. Spot measurements of water temperatures are at best inadequate when used in conjunction with other data. Rather, understanding biotic responses to thermal regimes not only involves fundamental research on life histories of taxa, but also in relating these to the subtler statistics of a thermal regime. The collection and/or modelling of sub-daily temperatures (mean, minimum and maximum values) is fundamental to describing thermal regimes relative to timing, frequency, duration and magnitude of thermal events.

Any ecological Reserve determination in South Africa would be incomplete unless thermographs are considered together with hydrological assessments. We provide a new framework for doing this, based on "shapes" of thermal regimes, and disaggregation of regimes into ecologically meaningful metrics. By providing a spatial framework of thermal regions in the Eastern and Western Cape provinces, it becomes possible to assign broad thermal reference conditions for each region, against which ecological Reserve determinations can be compared. The temporal and spatial complexity of river temperature signatures, and the fact that this research addressed many of these complexities while highlighting the many more yet to be explained, affirms the need for ongoing research into water temperature patterns in South Africa. The poor correlations between water temperature spatial patterns and existing spatial frameworks (such as Level I Ecoregions) reinforces the naiveté of addressing water temperatures as a lumped variable within existing paradigms.

Finally, we have demonstrated that through this spatial component, it becomes possible to undertake scenario assessments with an explicit spatial element, and linking, *inter alia*, thermal changes to biotic responses. While not all sites have been correctly classified in the thermal regions maps, our research demonstrates not only the usefulness of this approach if extended to a national spatial framework, but also that river management approaches need take cognisance of the heterogenous nature of water temperature patterns and ecological responses.

In summary, we have:

- provided the models for simulating water temperatures in the absence of water temperature data;
- automated the calculation of temperature metrics that facilitate the conversion of sub-daily temperature data into statistics that define a river's thermal regime with respect to magnitude of water temperatures, frequency, timing and duration of thermal events;
- identified thermally sensitive macroinvertebrate taxa that may be used as bio-indicators of thermal alteration;
- identified key life history cues for selected macroinvertebrate taxa in the context of water temperatures;
- demonstrated the importance of maintaining thermal variability in river systems for aquatic macroinvertebrate community structure;
- generated a preliminary map of thermal regions that can provide an initial framework within which the thermal ecological Reserve is applied;
- developed a thermograph that incorporates the natural range of variability and the concept of reference sites (and condition) with which an assessed (impacted) site can be compared, and the effect (if any) quantified; and
- provided a decision tree for determining thermal ecological Reserve exceedance.

Future areas of research should focus on the following:

- Expansion of the geographic range of this study to a national scale.
- Establishment of a water temperature monitoring network to collect long-term water temperature data. This facilitates detecting warming or cooling trends, as well as departures from current conditions.
- Development of management tools from this research for use by practitioners and managers in ecological Reserve determinations. Links between environmental flows, thermal regimes and biotic responses need to be made more explicit.
- Continuation of life history studies to expand our knowledge of the extent to which species are cued into water temperature and flow.
- Continuation of thermal tolerance studies to determine the extent to which bio-indicators are regionally applicable and the potential to utilise family level data.

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**Appendix 1 Site information for water temperature study sites. Sites marked with an * are from associated projects (Table Mountain Aquifer, Water Research Commission Periphyton – K5/1676, and Table Mountain Fund – WWF Projects). Each WRC site has been characterised using the site characterisation information collected during logger installation and which was based on Dallas (2005c). Longitudinal Zone is based Rowntree and Wadeson's classification (2000).**

| Site Name | River | Tributary of | Latitude | Longitude | Ecoregion I | Longitudinal Zone | Primary Catchment | Stream Order | Altitude | Associated Air Temperature Logger |
|---|---|---|---|---|---|---|---|---|---|---|
| E1DWAR-KRAKA | Dwars | Jan Dissels | -32.219720 | 18.988850 | Western Folded Mountains | Transitional | E | 1 | 202 | E1DWAR-KRAKA_AT |
| E1DWAR-KRAKA_AT | N/A | N/A | -32.219720 | 18.988850 | Western Folded Mountains | N/A | E | | 202 | |
| E1OLIF-ABBUL | Olfants | | -32.060910 | 18.825690 | Western Folded Mountains | Lower Foothill | E | 3 | 67 | |
| E1OLIF-ABDAM | Olfants | | -32.348010 | 18.946100 | Western Folded Mountains | Lower Foothill | E | 3 | 115 | |
| E1OLIF-BOSRI | Olfants | | -33.144220 | 19.233700 | Western Folded Mountains | Upper Foothill | E | 1 | 730 | E1OLIF-KEERO_AT |
| E1OLIF-KEERO | Olfants | | -32.862690 | 19.088970 | Western Folded Mountains | Lower Foothill | E | 2 | 230 | E1OLIF-KEERO_AT |
| E1OLIF-KEERO_AT | N/A | N/A | -32.862690 | 19.088970 | Western Folded Mountains | N/A | E | | 280 | |
| E1RATE-BEAVE | Ratel | Olifants | -32.896960 | 19.072470 | Western Folded Mountains | Mountain stream | E | 1 | 350 | E1RATE-BEAVE_AT |
| E1RATE-BEAVE_AT | N/A | N/A | -32.896960 | 19.072470 | Western Folded Mountains | N/A | E | | 460 | |
| E1ROND-EWR03 | Rondegat | Olifants | -32.370610 | 19.054160 | Western Folded Mountains | Transitional | E | 1 | 480 | E1ROND-EWR03_AT |
| E1ROND-EWR03_AT | N/A | N/A | -32.370610 | 19.054160 | Western Folded Mountains | N/A | E | | 500 | |
| E1ROND-KEURB | Rondegat | Olifants | -32.261530 | 18.966180 | Western Folded Mountains | Upper Foothill | E | 1 | 153 | E1ROND-EWR03_AT |
| E2DORI-OUDRI | Doring | Olifants | -31.856910 | 18.913170 | Western Folded Mountains | Lower Foothill | E | 5 | 80 | E2DORI-OUDRI_AT |
| E2DORI-OUDRI_AT | N/A | N/A | -31.856910 | 18.913170 | Western Folded Mountains | N/A | E | | 80 | |
| E2GROO-EWR06 | Groot | Doring | -32.657110 | 19.398850 | Western Folded Mountains | Upper Foothill | E | 3 | 491 | E2GROO-EWR06_AT |
| E2GROO-EWR06_AT | N/A | N/A | -32.657110 | 19.398850 | Western Folded Mountains | N/A | E | | 491 | |
| E2MATJ-MATJI | Matjies | Doring | -32.518840 | 19.350440 | Western Folded Mountains | Upper Foothill | E | 1 | 721 | |
| E2TWEE-MIDDL | Twee | Groot | -32.677010 | 19.233510 | Western Folded Mountains | Mountain stream | E | 1 | 701 | |
| G1BERG-BRBM1* | Berg | | -33.956970 | 19.072800 | Southern Folded Mountains | Upper Foothill | G | 1 | 250 | G1BERG-BRBM1_AT |
| G1BERG-BRBM1*_AT | N/A | N/A | -33.956970 | 19.072800 | Southern Folded Mountains | N/A | G | | 270 | |
| G1BERG-DRIEH | Berg | | -33.130900 | 18.862790 | South Western Coastal Belt | Lowland River | G | 4 | 30 | G1BERG-DRIEH_AT |
| G1BERG-DRIEH_AT | N/A | N/A | -33.130900 | 18.862790 | South Western Coastal Belt | N/A | G | | 30 | |
| G1BERG-KERSE | Berg | | -32.904400 | 18.329970 | South Western Coastal Belt | Lowland River | G | 4 | 18 | G1BERG-KERSE_AT |

| Site Name | River | Tributary of | Latitude | Longitude | Ecoregion I | Longitudinal Zone | Primary Catchment | Stream Order | Altitude | Associated Air Temperature Logger |
|---|---|---|---|---|---|---|---|---|---|---|
| G1BERG-KERSE_AT | N/A | N/A | -32.904400 | 18.329970 | South Western Coastal Belt | N/A | G | | 18 | |
| G1BERG-SKUIF* | Berg | | -33.891500 | 19.051908 | South Western Coastal Belt | Upper Foothill | G | 2 | 150 | G1BERG-BRBM1_AT |
| G1KLEI-NUWEJ | Klein Berg | Berg | -33.313820 | 19.075450 | South Western Coastal Belt | Lower Foothill | G | 3 | 79 | |
| G1TWEN-DASBO | Twenty Four | Berg | -33.140300 | 19.056860 | South Western Coastal Belt | Transitional | G | 2 | 118 | G1TWEN-DASBO_AT |
| G1TWEN-DASBO_AT | N/A | N/A | -33.140300 | 19.056860 | South Western Coastal Belt | N/A | G | | 118 | |
| G1WEMM-WEMME | Wemmershoek | Berg | -33.848820 | 19.050510 | South Western Coastal Belt | Upper Foothill | G | 3 | 200 | G1WEMM-WEMME_AT |
| G1WEMM-WEMME_AT | N/A | N/A | -33.848820 | 19.050510 | South Western Coastal Belt | N/A | G | | 200 | |
| G1WOLW-ABDAM | Wolwekloof | Berg | -33.944167 | 19.026389 | Southern Folded Mountains | Mountain stream | G | 1 | 330 | G1BERG-BRBM1_AT |
| G2DIEP-NOOIT | Diep | | -33.632090 | 18.626620 | South Western Coastal Belt | Lower Foothill | G | 2 | 44 | |
| G2EERS-JONKE | Eerste | | -33.993776 | 18.975550 | Southern Folded Mountains | Mountain Stream | G | 1 | 380 | G2EERS-JONKE_AT |
| G2EERS-JONKE_AT | N/A | N/A | -33.993776 | 18.975550 | Southern Folded Mountains | N/A | G | | 300 | |
| G2LANG-JONKE | Lang | Eerste | -33.988670 | 18.968900 | Southern Folded Mountains | Mountain Stream | G | 1 | 350 | G2EERS-JONKE_AT |
| G4BOTR-PORCU | Bot | | -34.113690 | 19.239610 | Southern Folded Mountains | Upper Foothill | G | 1 | 172 | G4BOTR-PORCU_AT |
| G4BOTR-PORCU_AT | N/A | N/A | -34.113690 | 19.239610 | Southern Folded Mountains | N/A | G | | 190 | |
| G4HERM-SAFCO | Hermanus | | -34.293800 | 19.117520 | Southern Folded Mountains | Mountain stream | G | 2 | 166 | |
| G4LUIP-HAROL | Luiperdskloof | Disa | -34.348790 | 18.927390 | Southern Folded Mountains | Upper Foothill | G | 1 | 40 | G4LUIP-HAROL_AT |
| G4LUIP-LUIPE | Luiperdskloof | Disa | -34.344830 | 18.933080 | Southern Folded Mountains | Mountain Stream | G | 1 | 110 | G4LUIP-HAROL_AT |
| G4LUIP-HAROL_AT | N/A | N/A | -34.348790 | 18.927390 | Southern Folded Mountains | N/A | G | | 40 | |
| G4PALM-KODAM | Palmiet | | -34.218340 | 18.975250 | Southern Folded Mountains | Upper Foothill | G | 2 | 200 | |
| G4PALM-KOGFR | Palmiet | | -34.320550 | 18.970400 | Southern Folded Mountains | Rejuvenated Foothill | G | 2 | 29 | G4PALM-KOGFR_AT |
| G4PALM-KOGFR_AT | N/A | N/A | -34.320550 | 18.970400 | Southern Folded Mountains | N/A | G | | 29 | |
| H1BREE-WITBR | Breede | | -33.421400 | 19.267090 | Western Folded Mountains | Upper Foothill | H | 3 | 250 | |
| H1ELAN-TUNNE | Elandspad | Molenaars | -33.736667 | 19.114722 | Western Folded Mountains | Transitional | H | 2 | 450 | H1ELAN-TUNNE_AT |
| H1ELAN-TUNNE_AT | N/A | N/A | -33.736667 | 19.114722 | Western Folded Mountains | N/A | H | | 450 | |
| H1HOLS-STETT | Holsloot | Breede | -33.837930 | 19.256310 | Western Folded Mountains | Upper Foothill | H | 1 | 430 | H1HOLS-STETT_AT |
| H1HOLS-STETT_AT | N/A | N/A | -33.837930 | 19.256310 | Western Folded Mountains | N/A | H | | 430 | |
| H1MOLE-TUNNE | Molenaars | Smalblaar | -33.731390 | 19.115000 | Western Folded Mountains | Transitional | H | 2 | 440 | H1ELAN-TUNNE_AT |
| H1WITR-MONUM | Witte | Breede | -33.637090 | 19.107890 | Western Folded Mountains | Transitional | H | 1 | 660 | H1WITR-MONUM_AT |
| H1WITR-MONUM_AT | N/A | N/A | -33.637090 | 19.107890 | Western Folded Mountains | N/A | H | | 660 | |
| H2ROOI-WORCE | Rooi-Elskloof | Hex | -33.461100 | 19.617860 | Western Folded Mountains | Mountain stream | H | 1 | 481 | H2ROOI-WORCE_AT |

| Site Name | River | Tributary of | Latitude | Longitude | Ecoregion I | Longitudinal Zone | Primary Catchment | Stream Order | Altitude | Associated Air Temperature Logger |
|---|---|---|---|---|---|---|---|---|---|---|
| H2ROOI-WORCE_AT | N/A | N/A | -33.461100 | 19.617860 | Western Folded Mountains | N/A | H | | 481 | |
| H3GROO-POTJI | Groot | Kingna | -33.821340 | 20.317140 | Southern Folded Mountains | Lower Foothill | H | 2 | 295 | H3GROO-POTJI_AT |
| H3GROO-POTJI_AT | N/A | N/A | -33.821340 | 20.317140 | Southern Folded Mountains | N/A | H | | 295 | |
| H4BREE-LECHA | Breede | | -33.818070 | 19.693550 | Southern Folded Mountains | Lowland River | H | 4 | 162 | H4BREE-LECHA_AT |
| H4BREE-LECHA_AT | N/A | N/A | -33.818070 | 19.693550 | Southern Folded Mountains | N/A | H | | 162 | |
| H5BOES-PIKKI | Boesmans | Breede | -34.041690 | 19.960890 | Southern Coastal Belt | Transitional | H | 1 | 277 | H5BOES-PIKKI_AT |
| H5BOES-PIKKI_AT | N/A | N/A | -34.041690 | 19.960890 | Southern Coastal Belt | N/A | H | | 277 | |
| H6DUTO-WEIR1 | Du Toits | Riviersonderend | -33.940480 | 19.169980 | Southern Folded Mountains | Upper Foothill | H | 2 | 352 | |
| H7BREE-SPRIN | Breede | | -34.179330 | 20.505610 | Southern Coastal Belt | Lowland River | H | 4 | 12 | H7BREE-SPRIN_AT |
| H7BREE-SPRIN_AT | N/A | N/A | -34.179330 | 20.505610 | Southern Coastal Belt | N/A | H | | 12 | |
| H7DUIW-MARLO | Duiwelsbos | Koringlands | -33.998610 | 20.457760 | Southern Coastal Belt | Mountain stream | H | 1 | 190 | H7DUIW-MARLO_AT |
| H7DUIW-MARLO_AT | N/A | N/A | -33.998610 | 20.457760 | Southern Coastal Belt | N/A | H | | 240 | |
| H7TRAD-OSKLO | Tradou | Buffelsjags-Breede | -33.947430 | 20.708540 | Southern Folded Mountains | Upper Foothill | H | 2 | 270 | |
| H8DUIW-DOORN | Duiwenshoek | | -34.008880 | 20.916300 | Southern Coastal Belt | Upper Foothill | H | 1 | 115 | |
| H8DUIW-VERMA | Duiwenshoek | | -34.251650 | 20.991860 | Southern Coastal Belt | Lower Foothill | H | 2 | 9 | |
| H9GOUK-KLPFN | Goukou | | -34.271190 | 21.299170 | Southern Coastal Belt | Lower Foothill | H | 3 | 8 | |
| J1BOKR-BOKRE | Bok | Donkies-Touws | -33.354820 | 19.718530 | Western Folded Mountains | Mountain Stream | J | 1 | 1031 | |
| H8_1* | Steenbras | | -34.183790 | 18.906020 | Southern Folded Mountains | Mountain stream | G | 1 | 415 | |
| K_2a_Oudebos* | Oudebos | Palmiet | -34.327380 | 18.961330 | Southern Folded Mountains | Transitional | G | 1 | 79 | G4PALM-KOGFR_AT |
| K_3al* | Tributary | Palmiet | -34.301430 | 18.940140 | Southern Folded Mountains | Mountain stream | G | 1 | 70 | G4PALM-KOGFR_AT |
| K_4_Dwars* | Dwars | Palmiet | -34.292470 | 18.936080 | Southern Folded Mountains | Lowland River | G | 1 | 63 | G4PALM-KOGFR_AT |
| T4_Pal1 Palmiet* | Palmiet | | -34.055750 | 19.040667 | Southern Folded Mountains | Mountain Stream | H | 1 | 620 | G4PALM-NUWEB_AT |
| T4_Pal1 Palmiet *_AT | N/A | N/A | -34.055750 | 19.040667 | Southern Folded Mountains | N/A | H | | 620 | |
| T4_Pal3* | Palmiet-Trib | Palmiet | -34.070767 | 19.039283 | Southern Folded Mountains | Mountain Stream | H | 1 | 650 | G4PALM-NUWEB_AT |
| T4_RSE2* | Tributary 2 | Riviersonderend | -34.045530 | 19.034600 | Southern Folded Mountains | Mountain Stream | H | 1 | 660 | G4PALM-NUWEB_AT |
| T4_RSE3* | Tributary 3 | Riviersonderend | -34.049400 | 19.029850 | Southern Folded Mountains | Upper Foothill | H | 1 | 760 | G4PALM-NUWEB_AT |
| T6_1a Bobbejaan* | Bobbejaan | Riviersonderend | -34.004283 | 19.098317 | Southern Folded Mountains | Mountain Stream | H | 1 | 350 | |
| T6_2a* | Tributary 1 | Bobbejaan | -33.999667 | 19.105900 | Southern Folded Mountains | Mountain Stream | H | 1 | 380 | |
| T8_2a* | Purgatory | Riviersonderend | -33.948580 | 19.178820 | Southern Folded Mountains | Mountain Stream | H | 1 | 430 | |

| Site Name | River | Tributary of | Latitude | Longitude | Ecoregion I | Longitudinal Zone | Primary Catchment | Stream Order | Altitude | Associated Air Temperature Logger |
|---|---|---|---|---|---|---|---|---|---|---|
| V3_1 (N)* | Voëlvlei Trib 1 | Sandspruit | -33.459917 | 19.111750 | Western Folded Mountains | Mountain Stream | G | 1 | 440 | |
| V3_2 (S)* | Voëlvlei Trib 2 | Sandspruit | -33.461217 | 19.119617 | Western Folded Mountains | Mountain Stream | G | 1 | 410 | |
| W7_1 Drakenstein* | Drakenstein | Wemmers | -33.802110 | 19.071120 | South Western Coastal Belt | Upper Foothill | G | 1 | 330 | G1WEMM-WEMME_AT |
| W7_4 Kasteelskloof* | Kasteelskloof | Wemmers | -33.815990 | 19.060590 | South Western Coastal Belt | Mountain Stream | G | 1 | 310 | G1WEMM-WEMME_AT |
| W7_6 Zachariashoek* | Zachariashoek | Wemmers | -33.827980 | 19.036310 | South Western Coastal Belt | Upper Foothill | G | 1 | 300 | G1WEMM-WEMME_AT |
| G1WIND-KIRST* 1 | Window Gorge | Liesbeek | -33.975278 | 18.420639 | Southern Folded Mountains | Mountain Headwater | G | 1 | 814 | |
| G1WIND-KIRST* 2 | Window Gorge | Liesbeek | -33.978139 | 18.428167 | Southern Folded Mountains | Mountain Headwater | G | 1 | 332 | |
| G1WIND-KIRST* 3 | Window Gorge | Liesbeek | -33.980722 | 18.428250 | Southern Folded Mountains | Mountain Headwater | G | 1 | 259 | |
| G1WIND-KIRST* 4 | Window Gorge | Liesbeek | -33.983939 | 18.431833 | Southern Folded Mountains | Mountain stream | G | 1 | 164 | |
| G1WIND-KIRST* 5 | Window Gorge | Liesbeek | -33.985839 | 18.433911 | Southern Folded Mountains | Mountain stream | G | 1 | 130 | |
| G1WIND-KIRST* 6 | Window Gorge | Liesbeek | -33.987778 | 18.437333 | Southern Folded Mountains | Upper Foothill | G | 1 | 105 | |
| J2DORPS-DFLOOR | Dorps | Gamka | -33.29418 | 22.05231 | Southern Folded Mountains | Mountain stream | J | 1 | 625 | |
| J3DOR-AFGUNS_AT | N/A | N/A | -33.86192 | 22.45278 | South Eastern Coastal Belt | NA | J | | 651 | |
| J3DOR-GLEND | Doring | Olifants | -33.85777 | 22.44797 | Southern Folded Mountains | Unclassified | J | 1 | 610 | J3DOR-AFGUNS_AT |
| J3GROO-PARAD | Groot | Olifants | -33.27199 | 22.38966 | Southern Folded Mountains | Upper foothills | J | 1 | 857 | |
| J3HOEK-CANGO | Grobbelaars | Olifants | -33.39476 | 22.18751 | Southern Folded Mountains | Upper foothills | J | 1 | 615 | J3HOEK-CANGO_AT |
| J3HOEK-CANGO_AT | N/A | N/A | -33.39740 | 22.18866 | Southern Folded Mountains | NA | J | | 623 | |
| J3HOL-UNION | Holdrif | Olifants | -33.69659 | 23.15142 | Southern Folded Mountains | Upper foothills | J | 1 | 822 | |
| K3KEUR-MONTA | Malgas | Gwaing | -33.90668 | 22.41861 | South Eastern Coastal Belt | Transitional | K | 1 | 329 | J3DOR-AFGUNS_AT |
| K6DIEP-PALBT | Diep | Keurbooms | -33.85938 | 23.17213 | South Eastern Coastal Belt | Unclassified | K | 1 | 335 | K6KEUR-OUTEN_AT |
| K6KEUR-KAT | Keurbooms | N/A | -33.79651 | 23.13404 | Southern Folded Mountains | Upper foothills | K | 1 | 324 | K6KEUR-OUTEN_AT |
| K6KEUR-KWAAI | Kwaai | Keurbooms | -33.82030 | 23.18126 | South Eastern Coastal Belt | Unclassified | K | 1 | 275 | K6KEUR-OUTEN_AT |
| K6KEUR-OUTEN_AT | N/A | N/A | -33.82110 | 23.18207 | South Eastern Coastal Belt | NA | K | | 272 | |
| K6KEUR-OUTEN1 | Keurbooms | N/A | -33.82188 | 23.18214 | South Eastern Coastal Belt | Lower foothills | K | 2 | 275 | K6KEUR-OUTEN_AT |
| K6KEUR-OUTEN2 | Keurbooms | N/A | -33.82084 | 23.18103 | South Eastern Coastal Belt | Upper foothills | K | 1 | 273 | K6KEUR-OUTEN_AT |
| K6KEUR-PLET | Keurbooms | N/A | -33.96748 | 23.40266 | South Eastern Coastal Belt | Lower foothills | K | 3 | 0 | |
| K6KEUR-RHINO | Keurbooms | N/A | -33.93544 | 23.36702 | South Eastern Coastal Belt | Upper foothills | K | 3 | 30 | K6KEUR-RHINO_AT |
| K6KEUR-RHINO_AT | N/A | N/A | -33.94536 | 23.35122 | South Eastern Coastal Belt | NA | K | | 233 | |
| K6KEUR-WAG | Keurbooms | N/A | -33.77761 | 22.95616 | South Eastern Coastal Belt | Transitional | K | 1 | 583 | K6KEUR-WAG_AT |
| K6KEUR-WAG_AT | N/A | N/A | -33.77518 | 22.95577 | South Eastern Coastal Belt | NA | K | | 584 | |

| Site Name | River | Tributary of | Latitude | Longitude | Ecoregion I | Longitudinal Zone | Primary Catchment | Stream Order | Altitude | Associated Air Temperature Logger |
|---|---|---|---|---|---|---|---|---|---|---|
| K8ELAN-FORES | Elandsbos | Elandsbos | -33.97497 | 23.77644 | South Eastern Coastal Belt | Upper foothills | K | 1 | 210 | |
| L8BAV-RES1 | Baviaanskloof | Gamtoos | -33.61649 | 24.24472 | Southern Folded Mountains | Lower foothills | L | 2 | 310 | L8BAV-ZAND_AT |
| L8BAV-RES2 | Baviaanskloof | Gamtoos | -33.61549 | 24.24472 | Southern Folded Mountains | Lower foothills | L | 2 | 310 | L8BAV-ZAND_AT |
| L8BAV-ZAND_AT | N/A | N/A | -33.58418 | 24.16375 | Southern Folded Mountains | NA | L | | 381 | |
| M1ELAND-CYPH | Elands | Swartkops | -33.72450 | 25.11751 | Southern Folded Mountains | Upper foothills | M | 2 | 264 | M1ELAND-CYPH_AT |
| M1ELAND-CYPH_AT | N/A | N/A | -33.72450 | 25.11751 | Southern Folded Mountains | NA | M | | 266 | |
| M1ELAND-WHEAT | Elands | Swartkops | -33.64396 | 24.94880 | Southern Folded Mountains | Upper foothills | M | 1 | 447 | M1ELAND-CYPH_AT |
| N1GATS-AASVOE | Gats | Sundays | -31.90141 | 24.56886 | Drought Corridor | Upper foothills | N | 3 | 1245 | N1WILGER-BOOY_AT |
| N1SUN-CRAD | Sundays | N/A | -31.91527 | 24.78685 | Drought Corridor | Upper foothills | N | 1 | 1369 | |
| N1WILGER-BOOY | Wilgerbos | Sundays | -31.81631 | 24.59505 | Drought Corridor | Upper foothills | N | 1 | 1425 | N1WILGER-BOOY_AT |
| N1WILGER-BOOY_AT | N/A | N/A | -31.81825 | 24.59025 | Drought Corridor | NA | N | | 1426 | |
| N4SUN-ADDO_AT | N/A | N/A | -33.21019 | 25.14254 | Great Karoo | NA | N | | 210 | |
| N4SUN-DARLI | Sundays | N/A | -33.21221 | 25.14978 | Great Karoo | Lower foothills | N | 4 | 207 | N4SUN-ADDO_AT |
| N4SUN-KIRK | Sundays | N/A | -33.38373 | 25.34882 | South Eastern Coastal Belt | Lower foothills | N | 4 | 120 | N4SUN-KIRK_AT |
| N4SUN-KIRK_AT | N/A | N/A | -33.40375 | 25.33620 | South Eastern Coastal Belt | NA | N | | 127 | |
| P3PALM-HOWI | Palmiet | Kowie | -33.36910 | 26.47455 | Southern Folded Mountains | Transitional | P | 1 | 363 | |
| P4BLOU-BELM | Bloukrans | Kowie | -33.32489 | 26.60082 | Southern Folded Mountains | Upper foothills | P | 1 | 480 | |
| P4BLOU-RES | Bloukrans | Kowie | -33.39125 | 26.70818 | Southern Folded Mountains | Upper foothills | P | 1 | 367 | |
| P4BLOU-VARNUM_AT | N/A | N/A | -33.32000 | 26.63000 | Southern Folded Mountains | NA | P | | 460 | |
| P4KOW-BATH_AT | N/A | N/A | -33.50752 | 26.82979 | South Eastern Coastal Belt | NA | P | | 210 | |
| P4KOWI-BATH | Kowie | N/A | -33.49986 | 26.74593 | South Eastern Coastal Belt | Upper foothills | P | 3 | 44 | P4KOWI-BATH_AT |
| P4KOWI-RES | Kowie | N/A | -33.54523 | 26.78718 | South Eastern Coastal Belt | Lowland river | P | 3 | 2 | P4KOWI-BATH_AT |
| Q1FISH-BRACKL | Great Fish | N/A | -31.89947 | 25.45949 | Drought Corridor | Lower foothills | Q | 4 | 982 | Q2FISH-SALTP_AT |
| Q1FISH-DEKEUR_AT1 | N/A | N/A | -31.60294 | 25.49614 | Drought Corridor | NA | Q | | 1111 | |
| Q1FISH-DEKEUR_AT2 | N/A | N/A | -31.60122 | 25.49304 | Drought Corridor | NA | Q | | 1098 | |
| Q1GBRAK-DEKEUR | Groot Brak | Great Fish | -31.60066 | 25.49287 | Drought Corridor | Lower foothills | Q | 4 | 1097 | Q1FISH-DEKEUR_AT2 |
| Q1GBRAK-SALTP1 | Groot Brak | Great Fish | -31.89474 | 25.43485 | Drought Corridor | Lower foothills | Q | 4 | 981 | Q2FISH-SALTP_AT |
| Q1GBRAK-SALTP2 | Groot Brak | Great Fish | -31.89411 | 25.43510 | Drought Corridor | Lower foothills | Q | 4 | 981 | Q2FISH-SALTP_AT |
| Q1GBRAK-SALTP3 | Unknown | Groot Brak | -31.89297 | 25.43591 | Drought Corridor | Lower foothills | Q | 1 | 987 | Q2FISH-SALTP_AT |
| Q1VANW-ZUURF | Vanwyks | Great Fish | -31.70900 | 24.74470 | Drought Corridor | Upper foothills | Q | 1 | 1560 | Q1VANW-ZUURF_AT |

| Site Name | River | Tributary of | Latitude | Longitude | Ecoregion I | Longitudinal Zone | Primary Catchment | Stream Order | Altitude | Associated Air Temperature Logger |
|---|---|---|---|---|---|---|---|---|---|---|
| Q1VANW-ZUURF_AT | N/A | N/A | -31.70944 | 24.68660 | Drought Corridor | NA | Q | | 1659 | |
| Q2FISH-POOL | Great Fish | N/A | -32.00237 | 25.13924 | Drought Corridor | Upper foothills | Q | 2 | 1192 | |
| Q2FISH-SALTP | Great Fish | N/A | -31.90562 | 25.42930 | Drought Corridor | Lower foothills | Q | 3 | 986 | Q2FISH-SALTP_AT |
| Q2FISH-SALTP_AT | N/A | N/A | -31.90002 | 25.43502 | Drought Corridor | NA | Q | | 986 | |
| Q4TARK-OAK | Tarka | Great Fish | -32.29069 | 26.29886 | Drought Corridor | Upper foothills | Q | 2 | 1410 | |
| Q7FISH-UITK_AT | N/A | N/A | -32.76309 | 25.65878 | Drought Corridor | NA | Q | | 716 | |
| Q8COET-DERUS | Coetzerskloof-spruit | Great Fish | -32.43010 | 25.36670 | Drought Corridor | Upper foothills | Q | 2 | 1141 | Q8COETZ-DERUS_AT |
| Q8COET-DERUS_AT | N/A | N/A | -32.43044 | 25.36449 | Drought Corridor | NA | Q | | 1150 | |
| Q8KVIS-COOKH | Klein Vis | Great Fish | -33.09568 | 25.82341 | Drought Corridor | Lower foothills | Q | 3 | 42 | Q8KVIS-GRASF_AT |
| Q8KVIS-DRIEFON | Klein Vis | Great Fish | -32.73214 | 25.50628 | Drought Corridor | Lower foothills | Q | 3 | 753 | Q8FISH-UITK_AT |
| Q8KVIS-GRASF_AT | N/A | N/A | -33.09247 | 25.80621 | Drought Corridor | NA | Q | | 42 | |
| Q8KVIS-R337 | Klein Vis | Great Fish | -32.50550 | 25.42614 | Drought Corridor | Upper foothills | Q | 3 | 1033 | |
| Q9BALF-WFALL | Balfour | Great Fish | -32.55785 | 26.67450 | South Eastern Uplands | Upper foothills | Q | 1 | 595 | Q9KAT-BADDA_AT |
| Q9FISH-CARLI | Great Fish | N/A | -33.08328 | 26.22550 | Drought Corridor | Lower foothills | Q | 5 | 381 | Q9FISH-CARLI_AT |
| Q9FISH-CON_AT | N/A | N/A | -33.09658 | 26.42016 | Drought Corridor | NA | Q | | 346 | |
| Q9FISH-GH012 | Great Fish | N/A | -33.08754 | 26.43636 | Drought Corridor | Lower foothills | Q | 5 | 307 | Q9FISH-CON_AT |
| Q9FISH-MORT | Great Fish | N/A | -32.36377 | 25.70432 | Drought Corridor | Lower foothills | Q | 5 | 797 | |
| Q9FISH-TRUM | Great Fish | N/A | -33.23384 | 26.95885 | Southern Folded Mountains | Lower foothills | Q | 5 | 39 | Q9FISH-TRUMP_AT |
| Q9FISH-TRUMP_AT | N/A | N/A | -33.23226 | 26.95587 | Southern Folded Mountains | NA | Q | | 64 | |
| Q9KAT-BADDA_AT | N/A | N/A | -32.73565 | 26.57621 | Drought Corridor | NA | Q | | 465 | |
| Q9KAT-SEYM | Kat | Great Fish | -32.59301 | 26.79389 | South Eastern Uplands | Unclassified | Q | 1 | 776 | Q9KAT-BADDA_AT |

## Appendix 2. Logger technology and datasheets


### Hobo UTB1-001 TidBit V2. Temperature Data Logger

### Technical Specifications

Temperature Sensor

Operation range:* -20° to 70°C (-4° to 158°F) in air; maximum sustained temperature of 30°C (86°F) in water*

Accuracy: 0.2°C over 0° to 50°C (0.36°F over 32° to 122°F)

Resolution: 0.02°C at 25°C (0.04°F at 77°F)

Response time: 5 minutes in water; 12 minutes in air moving 2 m/sec; 20 minutes in air moving 1 m/sec (typical to 90%)

Stability (drift): 0.1°C (0.18°F) per year

Logger: Real-time clock: ± 1 minute per month 0° to 50°C (32° to 122°F)

Battery: 3 Volt lithium, non-replaceable

Battery life (typical use): 5 years with 1 minute or greater logging interval

Memory (non-volatile): 64K bytes memory (approx. 42,000 12-bit temperature measurements)

Weight: 23 g (0.8 oz.)

Dimensions: 3.0 × 4.1 × 1.7 cm (1.2 × 1.6 × 0.65 in.); mounting bail 4.6 mm (3/16 in.) diameter

Wetted materials: Epoxy case

Waterproof: To 305 m (1000 ft.)

Logging interval: Fixed-rate or multiple logging intervals, with up to 8 user-defined logging intervals and durations; logging intervals from 1 second to 18 hours. Refer to HOBOware software manual.

Launch modes: Immediate start, delayed start

Offload modes: Offload while logging; stop and offload

Battery indication: Battery level can be viewed in status screen and optionally logged in datafile. Low battery indication in datafile.

## DS1923-F5 – Dallas Hygrochron Temperature & Humidity iButton

The **DS1923 temperature/humidity** logger **iButton** is a rugged, self-sufficient system that measures temperature and/or humidity and records the result in a protected memory section. The recording is done at a user-defined rate. A total of 8192 8-bit readings or 4096 16-bit readings taken at equidistant intervals ranging from 1s to 273hrs can be stored. In addition to this, there are 512 bytes of SRAM for storing application-specific information and 64 bytes for calibration data. A mission to collect data can be programmed to begin immediately, or after a user defined delay or after a temperature alarm. Access to the memory and control functions can be password protected. The DS1923 is configured and communicates with a host-computing device through the serial **1-Wire** protocol, which requires only a single data lead and a ground return. Every DS1923 is factory-lasered with a guaranteed unique 64-bit registration number that allows for absolute traceability. The durable stainless-steel package is highly resistant to environmental hazards such as dirt, moisture, and shock. Accessories permit the DS1923 to be mounted on almost any object, including containers, pallets and bags.

**Key Features**

- Digital Hygrometer Measures Humidity with 8-Bit (0.6%RH) or 12-Bit (0.04%RH) Resolution
- Operating Range: -20 to +85°C; 0 to 100%RH (see Safe Operating Range)
- Automatically Wakes Up, Measures Temperature and/or Humidity and Stores Values in 8kB of Datalog Memory in 8- or 16-Bit Format
- Digital Thermometer Measures Temperature with 8-Bit (0.5°C) or 11-Bit (0.0625°C) Resolution
  Temperature Accuracy Better than ±0.5°C from -10°C to +65°C with Software Correction
- Built-in Humidity Sensor for Simultaneous Temperature and Humidity Logging
- Capacitive Polymer Humidity-Sensing Element
- Hydrophobic Filter Protects Sensor Against Dust, Dirt, Water, and Contaminants
- Sampling Rate from 1s up to 273hrs
- Programmable Recording Start Delay After Elapsed Time or Upon a Temperature Alarm Trip Point
- Programmable High and Low Trip Points for Temperature and Humidity Alarms
- Quick Access to Alarmed Devices Through 1-Wire Conditional Search Function
- 512 Bytes of General-Purpose Memory Plus 64 Bytes of Calibration Memory
- Two-Level Password Protection of All Memory and Configuration Registers
- Communicates to Host with a Single Digital Signal at Up to 15.4kbps at Standard Speed or Up to 125kbps in Overdrive Mode Using 1-Wire Protocol
- Individually Calibrated in a NIST-Traceable Chamber
- Calibration Coefficients for Temperature and Humidity Factory Programmed into Nonvolatile (NV) Memory

## WRC Water Temperature Study: Field datasheets

| | | | |
|---|---|---|---|
| Assessor Name(s) | | | |
| Organisation | | Date | / / |

### GENERAL SITE INFORMATION

| | | | |
|---|---|---|---|
| Site information – assessed at the site | | | |
| RHP Site Code | | Project Site Number | |
| River | | Tributary of | |

Latitude and longitude co-ordinates:

| Degrees-minutes-seconds | or Decimal degrees | or Degrees & decimal minutes | | |
|---|---|---|---|---|
| S ° ' . " | S . ° | S ° . ' | Cape datum Clarke | |
| E 0 ° ' . " | E 0 . ° | E 0 ° . ' | WGS-84 datum | |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Site Description | | | | | | | |
| Map Reference (1: 50 000) | | Site Length (m) | | | Altitude (m) | | |
| Longitudinal Zone | Source zone | Mountain headwater stream | Mountain stream | Transitional | Upper foothill | Lower foothill | Lowland river |
| | Rejuvenated cascades (gorge) | | Rejuvenated foothill | | Upland floodplain | | |

| | | | |
|---|---|---|---|
| Hydrological Type: "natural" | Perennial | Seasonal | Ephemeral |
| Hydrological Type: "present-day" | Perennial | Seasonal | Ephemeral |
| Position relative to tributaries | Upstream confluence | Downstream confluence | Site not near tributaries |
| Aspect | | | |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| Desktop/spatial information – data used for classifying a site and subsequent querying of data | | | | | | | |
| Political Region | | Water Management Area | | | | | |
| Ecoregion I | | Ecoregion II | | | | | |
| Secondary Catchment | | Quaternary Catchment | | | | | |
| Water Chemistry Management Region | | | | | | | |
| Vegetation Type | | Geological Type | | | | | |
| Source Distance (km) | | Stream Order | | | | | |
| Rainfall Region | Summer | Winter | Aseasonal | Other: | | | |
| DWAF Gauging Station | Yes | No | Code: | | Distance Upstream | | Or Downstream |
| Flow Volume | | Groundwater-fed | | | | | |

Photographs

| |
|---|
| Numbers: |

**LOCATION DETAILS**

Sketch a map of the site showing the following details: scale, north, access to site, roads, bridges/crossings, gauges/instream barriers, buildings, flow direction. Record the following:

| Location and Landowner Detail: | | | | Contact No.: | | |
|---|---|---|---|---|---|---|
| | | | | Notify Owner? | yes | no |
| Permit Required? | yes | no | Details: | | | |
| Key Needed? | yes | no | Details: | | | |
| Farm Name: | | | | Farm Reg. Code: | | |

**CHANNEL CONDITION (In-channel and bank modifications)** – **Rate impacts on a scale of 0 to 4: 0 – none; 1 – limited; 2 – moderate; 3 – extensive; 4 –entire**

| In-channel and bank modifications | Upstream | | Downstream | | Comments |
|---|---|---|---|---|---|
| | Impact score | Distance | Impact score | Distance | |
| Bridge – elevated; in channel supports | | | | | |
| Bridge – elevated; side channel supports | | | | | |
| Causeways/low-flow bridges | | | | | |
| Bulldozing | | | | | |
| Canalisation – concrete/gabion | | | | | |
| Canalisation – earth/natural | | | | | |
| Gabions/reinforced bank | | | | | |
| Fences – in channel | | | | | |
| Gravel, cobble and/or sand extraction | | | | | |
| Roads in riparian zone – tar | | | | | |
| Roads in riparian zone – gravel | | | | | |
| Dams (large) | | | | | |
| Dams (small)/weir | | | | | |
| Other: | | | | | |

**CONDITION OF LOCAL CATCHMENT** – **Rate extent (land-use) or impact on a scale of 0 to 4: 0 – none; 1 – limited; 2 – moderate; 3 – extensive; 4 – entire. Indicate level of confidence: High (H), medium (M) or low (L).**

| Land-use | Within riparian zone | Beyond riparian zone | Potential impact on River Health | Level of confidence (H,M,L) | Comments (e.g. distance upstream/downstream, time since disturbance, etc.) |
|---|---|---|---|---|---|
| Afforestation – general | | | | | |
| Afforestation – felled area | | | | | |
| Agriculture – crops | | | | | |
| Agriculture – livestock | | | | | |
| Agriculture – irrigation | | | | | |
| Alien vegetation infestation | | | | | |
| Aquaculture | | | | | |
| Construction | | | | | |
| Roads | | | | | |
| Impoundment (weir/dam) | | | | | |
| Industrial Development | | | | | |
| Urban Development | | | | | |
| Rural Development | | | | | |
| Informal settlement | | | | | |
| Recreational | | | | | |
| Sewage Treatment Works | | | | | |
| Nature Conservation | | | | N/A | |
| Wilderness Area | | | | N/A | |
| Litter/debris | | | | | |
| Disturbance by wildlife | | | | | |
| Other: | | | | | |

**CHANNEL MORPHOLOGY**

| Lateral Mobility: tick one | |
|---|---|
| Confined | Channel laterally confined by valley side walls |
| Moderately confined | Channel course determined by macro-scale features, but some lateral migration possible |
| Non-confined | Channel free to migrate laterally over the valley floor |
| Entrenched | Active channel confined by steep banks and/or terraces |

| Channel pattern: tick one | |
|---|---|
| Single thread – low sinuosity | Single channel, laterally inactive |
| Single thread – high sinuosity – stable sinuous | Single channel, moderately, laterally active |
| Single thread – high sinuosity – laterally mobile | Meandering, laterally active, often with S-bends and oxbows |
| Multiple thread – braided | Multi-thread channel, laterally active, one or two channels divided by alluvial bars or islands (gravel or sand) |
| Multiple thread – anatomosing/anabranching | Multi-thread channel separated by vegetated or otherwise stable alluvial islands or bedrock divided by alluvial bars or islands (gravel or sand) |

| Channel type: tick channel type indicating dominant type(s) | | | | |
|---|---|---|---|---|
| Bedrock | | | | |
| Mixed bedrock and alluvial – dominant type(s) | sand | gravel | cobble | boulder |
| Alluvial with dominant type(s) | sand | gravel | cobble | boulder |

Water level at time of sampling -tick appropriate category

| | | | | | |
|---|---|---|---|---|---|
| Dry | Isolated pools | Low flow | Moderate flow | High flow | Flood |

Significant rainfall in the last week? – i.e. likely to have raised the water level

| | | |
|---|---|---|
| Yes | No | Comment: |

Canopy Cover -tick appropriate category

| | | | |
|---|---|---|---|
| Open | Partially Open | Closed | Comment: |

Water chemistry data

| Variable | Value | Units |
|---|---|---|
| pH | | |
| Conductivity | | |
| Temperature | | |
| Dissolved Oxygen (mgLl-1) | | |
| Percentage O2 Saturation | | |

Water turbidity – tick appropriate category

| | | | | |
|---|---|---|---|---|
| Clear | Discoloured | Opaque | Silty | Comment: |
| Turbidity (if measured (NTUs) | | | | |
| Secchi Depth (m) | | | | |

Stream dimensions – estimate widths and heights by ticking the appropriate categories; estimate average depth of dominant deep and shallow water biotopes.

| (m) | < 1 | 1-2 | 2-5 | 5-10 | 10-20 | 20-50 | 50-100 | >100 |
|---|---|---|---|---|---|---|---|---|
| Macro-channel width | | | | | | | | |
| Active-channel width | | | | | | | | |
| Water surface width | | | | | | | | |

| Bank height – Active channel | | | |
|---|---|---|---|
| (m) | < 1 | 1-3 | >3 |
| Left Bank | | | |
| Right Bank | | | |

| Dominant physical biotope | Average Depth (m) | Specify physical biotope type |
|---|---|---|
| Deep-water (>0.5 m) physical biotope (e.g. pool) | | |
| Shallow-water (<0.5 m) physical biotope (e.g. riffle) | | |

Substratum composition – Estimate abundance of each material using the scale: 0 – absent; 1 – rare; 2 – sparse; 3 – common; 4 – abundant; 5 – entire

| Material | Size class (mm) | Bed | Bank |
|---|---|---|---|
| Bedrock | | | |
| Boulder | > 256 | | |
| Cobble | 100-256 | | |
| Pebble | 16-100 | | |
| Gravel | 2-16 | | |
| Sand | 0.06-2 | | |
| Silt/mud/clay | < 0.06 | | |

| Degree of embeddedness of substratum (%) |
|---|
| 0-25 |
| 26-50 |
| 51-75 |
| 76-100 |

**INVERTEBRATE BIOTOPES** (present at a site compared to those actually sampled)

| Summarised river make up: ('pool'=pool only; 'run' only; 'riffle/rapid' only; '2 mix'=2 types, '3 mix'=3 types) | | | | |
|---|---|---|---|---|
| pool | run | Riffle/rapid | 2 mix | 3 mix |

OPTIONAL:
Rate abundance of each SASS and specific biotope present at a site using the scale: 0 – absent; 1 – rare; 2 – sparse; 3 – common; 4 – abundant; 5 – entire. Add additional specific biotopes if necessary.

| | | Specific Biotope | | | | | |
|---|---|---|---|---|---|---|---|
| SASS Biotope | Rating | | Rating | | Rating | | Rating |
| Stones in current | | Riffle | | Run | | Boulder rapid | |
| | | Chute | | Cascade | | Bedrock | |
| Stones out of current | | Backwater | | Slackwater | | Pool | |
| | | Bedrock | | | | | |
| Marginal vegetation in current | | Grasses | | Reeds | | Shrubs | |
| | | Sedges | | | | | |
| Marginal vegetation out of current | | Grasses | | Reeds | | Shrubs | |
| | | Sedges | | | | | |
| Aquatic vegetation | | Sedges | | Moss | | Filamentous algae | |
| Gravel | | Backwater | | Slackwater | | In channel | |
| Sand | | Backwater | | Slackwater | | In channel | |
| Silt/mud/clay | | Backwater | | Slackwater | | In channel | |

# Appendix 3: User Manual for the application of the Water Temperature Modelling system

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## Introduction

A modelling system consists of a number of (smaller) models which are linked together in such a manner that each model can be used as a stand-alone entity, or to derive outputs which are in turn used as inputs into a subsequent model. In the case of this modelling system, the aim is to simulate time series of sub-daily (mean, minimum and maximum) water temperatures. The choice of which models to use, and which sub-daily temperature components to simulate, are dictated by the overall study objectives – is the purpose of the study to relate biotic response to temperatures (in which case a maximum temperature model should be used); to study general trends down a river's axis (mean temperature model) or relative groundwater trends (minimum and maximum models). Additional considerations are:

- the turbidity of the water, since differences in maximum daily temperatures between a pool and a riffle will be more pronounced in a turbid river than in a clear river
- residency time of water in the system, which is a function of stream velocity. The ratio of pool to riffle should inform the choice of model i.e. a model for maximum water temperatures might be more appropriate for a stream with a large amount of riffle.

Choosing whether to use a simple statistical model, or to use this as input into a process-based model to undertake more complex analyses, is governed by the purposes the data are to be used for (Figure A1). Statistical models are less complex, while process based models have the capability to include water velocity and groundwater contributions, and be able to simulate "average" reach conditions or temperatures specific to hydraulic biotopes. Finally, temperature time series for a pool versus a riffle might have different residuals in the data, ultimately affecting model accuracy. A statistical model might not have the capability to reflect the nuances in the data as accurately as a process-based model. Again, the study objectives should inform this decision.

## Modelling system

The aim of this modelling system is to act as a support tool for aquatic scientists and managers working in South African rivers, who require daily time step water temperature data (mean, minimum and maximum) to undertake ecological studies and/or ecological Reserve assessments. In the absence of observed data, this modelling system allows users to simulate three daily water temperature values (daily mean, minimum and maximum) to meet one or more of the following objectives:

- Characterisation of a site using thermal metrics;

- Thermal Ecological Reserve Assessments, by providing temperature time series as either reference or comparative site data;
- Scenario assessments (climate change, turbidity, impact of upstream impoundment);
- Biological response modelling.

### Software requirements

- Spreadsheet
- Multiple linear regression water temperature models (Rivers-Moore and Mantel 2011)
- Stream Segment Temperature model (SSTEMP) (Bartholow 1989, 2002)

### Statistical models

If short-term water temperature data are available, and long-term air temperature data are also available which can be matched to the site, it is best to derive a site-specific linear regression model. Alternatively, if water temperature data are not available, models of the form of Equations A1-3 are suitable for simulating water temperature time series.

$$WT_{mean} = 0.46 + 0.70(AT) + 0.07(RH) \quad \text{[A1]}$$

$$WT_{\min} = -2.34 + 0.72(AT) + 0.08(RH) \quad \text{[A2]}$$

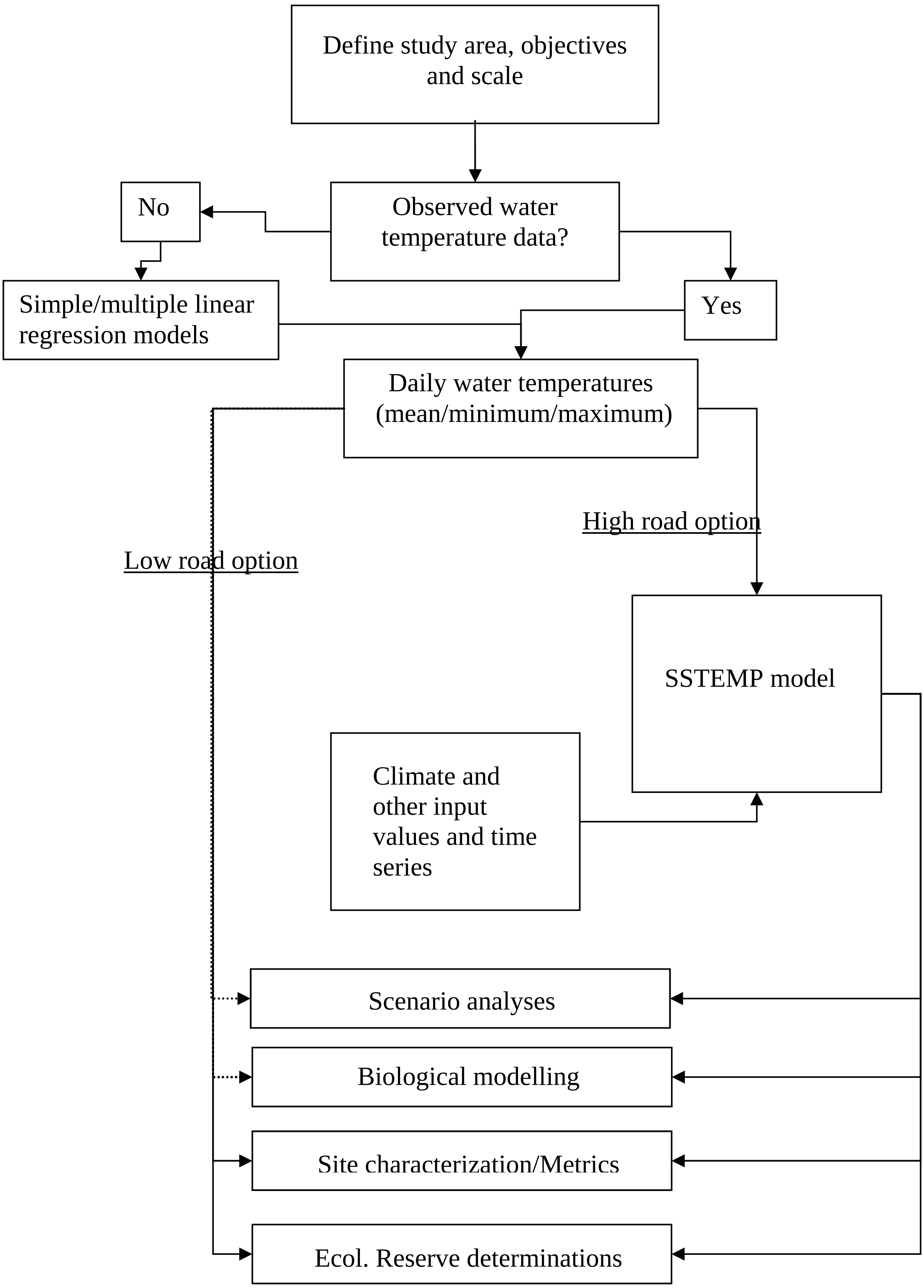
$$WT_{\max} = 3.97 + 0.69(AT) + 0.06(RH) \quad \text{[A3]}$$

where *AT* is mean daily air temperature, *RH* is relative humidity (%), and the subscripts for water temperature (*WT*) refer to daily mean, minimum and maximum temperatures.

### Populating SSTEMP

SSTEMP is a widely used daily time-step process-based temperature model, and is an adaptation of the stream network temperature model (SNTEMP) by Theurer et al. (1984), whose algorithms continue to be used in thermal studies (Webb and Zhang 1999). The model predicts maximum, minimum and mean temperatures at user-specified points and relies on measurement of a number of site and basin-specific variables, typically applied as data per ecoregion. SSTEMP uses 24-hour averages of input values to predict daily temperatures. "To predict the daily maximum, the model begins with the 24hour mean value at solar noon, and models the stream's response up to solar sunset, predicting the maximum. To estimate the minimum, the model makes a mirror image of the curve between the mean and the maximum by subtracting their difference from the mean (Sullivan et al. 1990, p. 110).

**Figure A1 Decision support flowchart for choosing modelling approach**



SSTEMP is freely downloadable software which is supplied with a user manual (Bartholow 2002). Inputs can be entered either directly using the user interface (Figure A2) to simulate water temperatures on a single day, or a time series of water temperatures of a desired length can be simulated for a user-input text file (Figure A3). Model variables can be populated according to Table A1. Good quality daily air temperatures are the most important variable for simulating water temperatures.

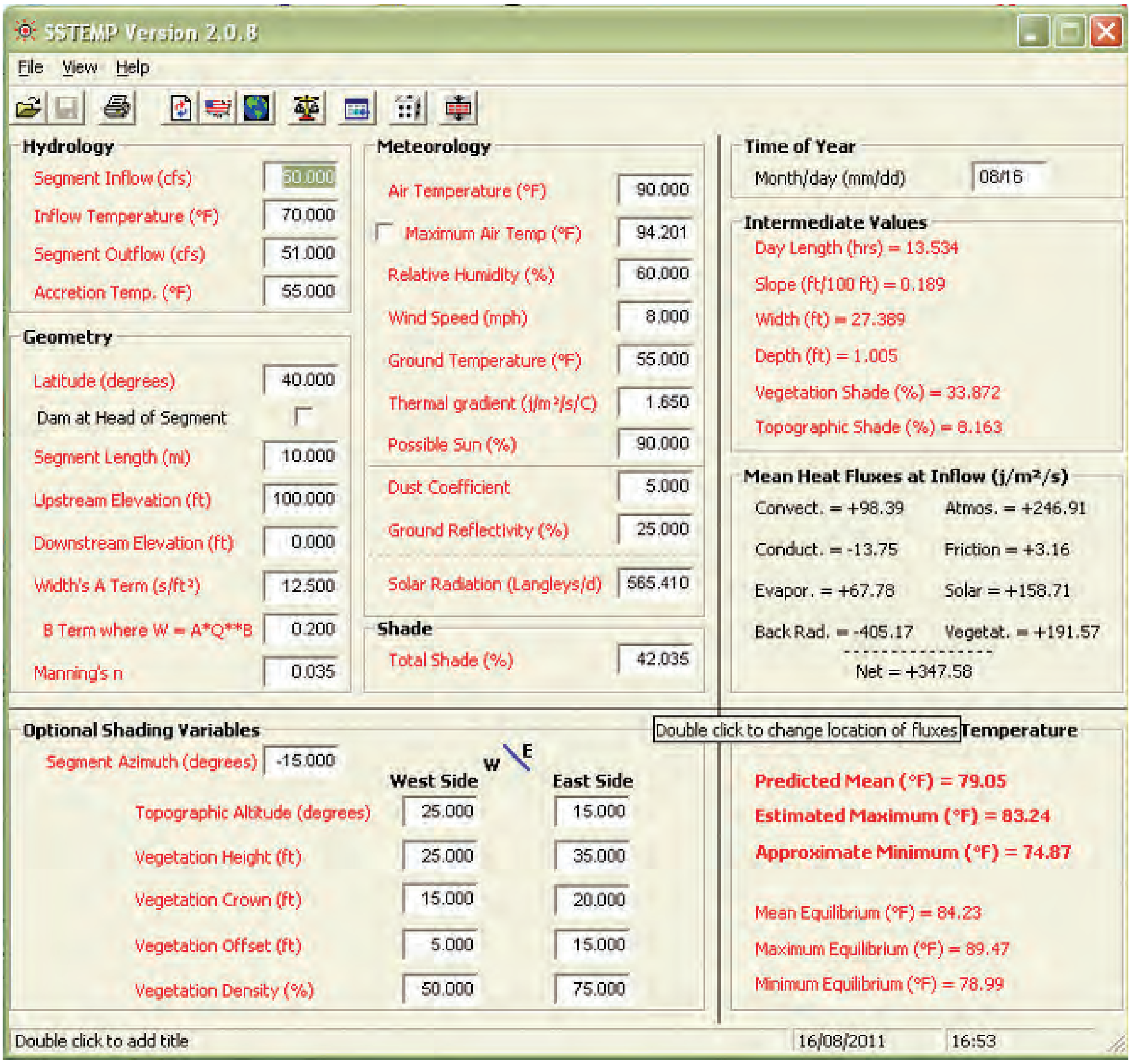
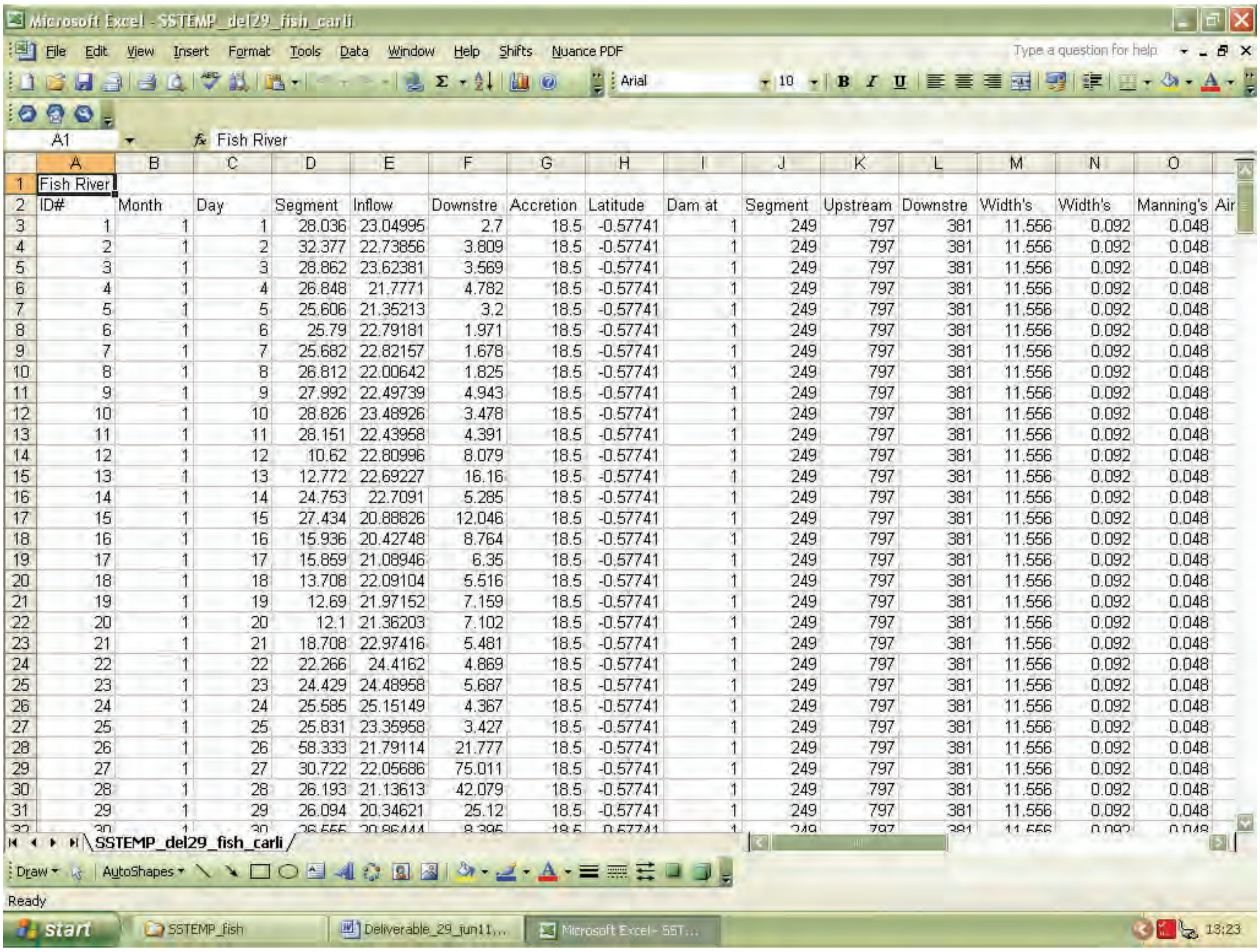



**Figure A2 SSTEMP user interface**



| | A | B | C | D | E | F | G | H | I | J | K | L | M | N | O |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Fish River | | | | | | | | | | | | | | |
| 2 | ID# | Month | Day | Segment | Inflow | Downstre | Accretion | Latitude | Dam at | Segment | Upstream | Downstre | Width's | Width's | Manning's |
| 3 | 1 | 1 | 1 | 28.036 | 23.04995 | 2.7 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 4 | 2 | 1 | 2 | 32.377 | 22.73856 | 3.809 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 5 | 3 | 1 | 3 | 28.862 | 23.62381 | 3.569 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 6 | 4 | 1 | 4 | 26.848 | 21.7771 | 4.782 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 7 | 5 | 1 | 5 | 25.606 | 21.35213 | 3.2 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 8 | 6 | 1 | 6 | 25.79 | 22.79181 | 1.971 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 9 | 7 | 1 | 7 | 25.682 | 22.82157 | 1.678 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 10 | 8 | 1 | 8 | 26.812 | 22.00642 | 1.825 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 11 | 9 | 1 | 9 | 27.992 | 22.49739 | 4.943 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 12 | 10 | 1 | 10 | 28.826 | 23.48926 | 3.478 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 13 | 11 | 1 | 11 | 28.151 | 22.43958 | 4.391 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 14 | 12 | 1 | 12 | 10.62 | 22.80996 | 8.079 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 15 | 13 | 1 | 13 | 12.772 | 22.69227 | 16.16 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 16 | 14 | 1 | 14 | 24.753 | 22.7091 | 5.285 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 17 | 15 | 1 | 15 | 27.434 | 20.88826 | 12.046 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 18 | 16 | 1 | 16 | 15.936 | 20.42748 | 8.764 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 19 | 17 | 1 | 17 | 15.859 | 21.08946 | 6.35 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 20 | 18 | 1 | 18 | 13.708 | 22.09104 | 5.516 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 21 | 19 | 1 | 19 | 12.69 | 21.97152 | 7.159 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 22 | 20 | 1 | 20 | 12.1 | 21.36203 | 7.102 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 23 | 21 | 1 | 21 | 18.708 | 22.97416 | 5.481 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 24 | 22 | 1 | 22 | 22.266 | 24.4162 | 4.869 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 25 | 23 | 1 | 23 | 24.429 | 24.48958 | 5.687 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 26 | 24 | 1 | 24 | 25.585 | 25.15149 | 4.367 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 27 | 25 | 1 | 25 | 25.831 | 23.35958 | 3.427 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 28 | 26 | 1 | 26 | 58.333 | 21.79114 | 21.777 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 29 | 27 | 1 | 27 | 30.722 | 22.05686 | 75.011 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 30 | 28 | 1 | 28 | 26.193 | 21.13613 | 42.079 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |
| 31 | 29 | 1 | 29 | 26.094 | 20.34621 | 25.12 | 18.5 | -0.57741 | 1 | 249 | 797 | 381 | 11.556 | 0.092 | 0.048 |

**Figure A3** **Example of spreadsheet file (saved as *.csv) for setting up inputs to run SSTEMP**

**Table A1 SSTEMP inputs**

| | | Example Data | | |
|---|---|---|---|---|
| Input | Units | 1 | 2 | Source |
| ID# | N/A | 1 | 1 | N/A |
| Month | N/A | 6 | 6 | N/A |
| Day | N/A | 24 | 25 | N/A |
| Segment Inflow | $m^3s^{-1}$ | 7.77 | 0.04 | Upstream weir or quinary |
| Inflow Temperature | °C | 10.98 | 8.40 | Upstream logger/modelled |
| Downstream Flow | $m^3s^{-1}$ | 6.74 | 0.12 | Upstream weir or quinary |
| Accretion Temperature | °C | 18.30 | 15.60 | ca. mean annual air temp. |
| Latitude | Radians | -0.58 | -0.59 | GPS/maps |
| Dam: Head of Segment | N/A | 1 | 0 | Yes/No |
| Segment Length | km | 56 | 6.3 | Measured or by velocity |
| Upstream Elevation | m amsl | 381 | 275 | GPS/maps |
| Downstream Elevation | m amsl | 307 | 273 | GPS/maps |
| Width's A Term | sm-2 | 11.56 | 9.28 | See text below |
| Width's B Term | N/A | 0.09 | 0.33 | See text below |
| Manning's n | N/A | 0.05 | 0.04 | See text below |
| Air Temperature | °C | 10.67 | 8.03 | Data logger – Mean daily |
| Maximum Air Temp | °C | 16.18 | 13.18 | Data logger |
| Relative Humidity | % | 55.52 | 86.22 | Data logger |
| Wind Speed | $ms^{-1}$ | 1.60 | 1.60 | Default ACRU value used |
| Ground Temperature | °C | 18.30 | 15.60 | ca. mean annual air temp. |
| Thermal Gradient | Jm-2s$^{-1}$°C | 1.65 | 1.65 | Default value |
| Percent Possible Sun | % | 90 | 90 | Default value |
| Dust Coefficient | N/A | -99 | -99 | Not used if radtn. used |
| Ground Reflectivity | % | -99 | -99 | Not used if radtn. used |
| Solar Radiation | Jm-2s$^{-1}$ | 112.07 | 79.63 | ACRU – convert |
| Total Shade | % | 5 | 5 | N/A |

***Sources of input data***

General climatological data – Schulze (2007)

Historical daily min/max air temperatures – Schulze and Maharaj (2004).

## Additional information for populating SSTEMP inputs

Certain of the variables listed in Table A1 require prior processing and calculations prior to being able to be used. Supplementary information, equations and techniques are presented in this section. Notes are provided on:

- estimating Width's A and B terms, segment length;
- locating mean daily flow data;
- solar radiation and incorporating increased turbidity effects on water temperatures;
- and whether to include impacts of upstream impoundments

Additional information relating to water temperature modelling is provided in Table A2.

**Table A2 Data, equations and constants relevant to water temperature modelling**

| Model constants | Value | Units | Source |
|---|---|---|---|
| Heat capacity of water | 4186 | J (kg °C) | Adams and Sullivan 1989 |
| Latent heat of vapourization of water | 2440x10 3 | J/kg | Adams and Sullivan 1989 |
| Water density1 | 1000 | kg/m$^3$ | Adams and Sullivan 1989 |
| Stefan-Boltzmann's constant2 | 5.67x108 | W/m$^2$.K4 | Chow et al. 1988 |
| Water emissivity | 0.97 | None | Chow et al.1988 |
| Albedo of water (clear)3 | 0.14 | None | Paaijmans et al. 2008 |
| Albedo of water (turbid)3 | 0.08 | None | Paaijmans et al. 2008 |

[1] Ranges from 1000 kg/m$^3$ at 4°C to 992.2 kg/m$^3$ at 40°C (Gordon et al. 1992 p. 7)

[2] Relates to calculation of evaporative flux

[3] This term was used to modify incident radiation as derived from ACRU (Schulze 2007), based on whether water was clear or turbid according to Equation A4 (Chow et al. 1988 p. 47). To estimate solar radiation at a particular time of day, should this be required, a "time-of-day" factor using a cosine multiplier on daily solar radiation may be used (Equation A5).

$$R_n = R_i(1-\alpha) - R_e \qquad [A4]$$

where the subscripts *n*, *i* and *e* refer to net, incident and emitted radiation (*R*) respectively, and *α* is albedo. In this equation, $R_e$ was regarded as negligible, as $R_i$ accounts for ca. 90% of radiation (Brown 1969)

Time of day factor: $TODF = \cos(\frac{\pi}{4}t + \pi)$ [A5]

Conversion of degrees to radians: $Radians = \deg rees * \frac{\pi}{180}$ [A6]

Manning's *n*, as a measure of flow resistance: SSTEMP uses a default value of 0.035 (Bartholow 2002), while values for rivers range from 0.030-0.050, depending on channel structure (Chow et al. 1988).

*Converting radiation units (Note Wm-2 ≡ Jm-2s$^{-1}$)*

MJm$^{-2}$day$^{-1}$ (ACRU format) → Jm$^{-2}$s$^{-1}$: multiply by 11.57 (1,000,000/86,400)

*Flow data* – Use observed flows from adjacent and upstream gauging weirs, or alternatively use simulated flow data using a suitable hydrological model.

*Width's A and B terms* – the relationship between width and discharge for any river system can be expressed as a constant exponential relationship (Equation A7), where the coefficients A and B may be calculated from known widths and discharges. Example data and relationships are provided (Table A3 and Figure A4).

$$W = AQ^B \qquad [A7]$$

where width's *A* term (sm$^{-2}$) is the untransformed y-intercept; *Q* is discharge, *W* is known width, and *B* is the power relationship (Bartholow 2002)

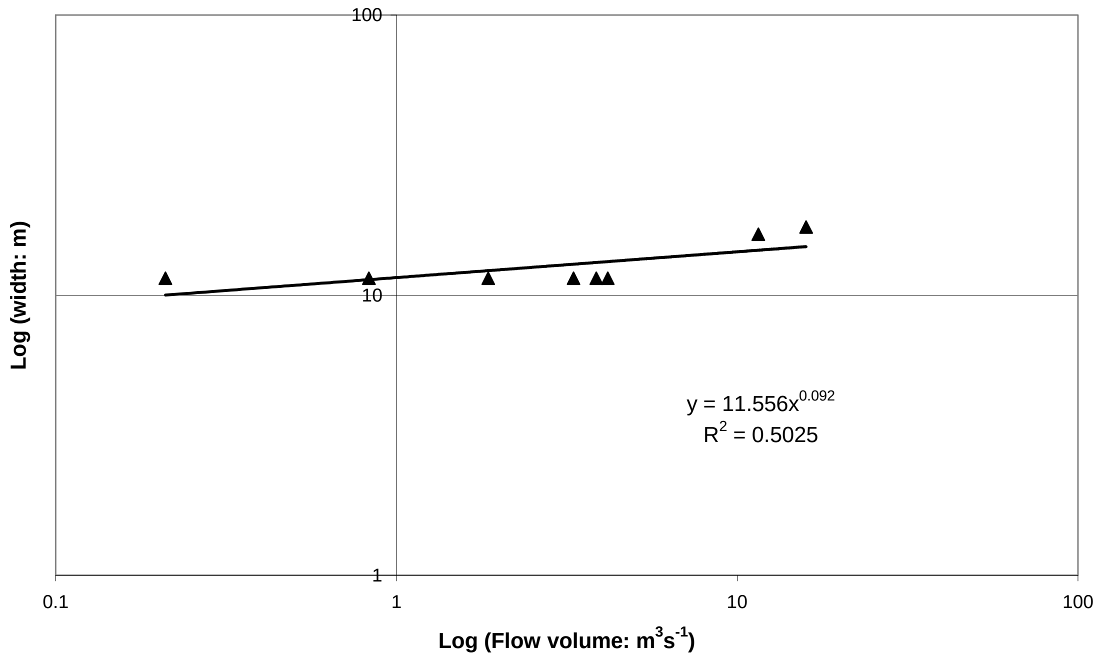



**Figure A4 Width-discharge relationship for the lower Great Fish River, from which the A and B terms of Equation A3iv could be calculated.**

**Table A3 Examples of Width's A and B terms**

| River system | A | B | $R^2$ |
|---|---|---|---|
| Site 2: Great Fish[1] | 11.56 | 0.092 | 0.50 |
| Site 1: Keurbooms[2] | 9.28 | 0.33 | 0.85 |
| Sites 3-4: Default[3] | 7.772 | 0.2 | N/A |

1 Rivers-Moore et al. 2008a; [2] Rivers-Moore 2010, unpub. data; [3] Bartholow 2002

*Segment length* – Sullivan et al. (1990) used a reach length equal to the 24-hour travel time computed from the water velocity. For the Fish River, a similar approach was used, where a typical velocity of 1 $ms^{-1}$ was calculated from known widths and mean discharges (Rivers-Moore et al. 2008a), and used to calculated the segment length over a 24-hour period (86 km). The closest upstream temperature data was 56 km upstream, which was the segment length used, and fell within the calculated segment length.

*Solar radiation* – Schulze and Chapman (2007) highlight the spatial and temporal variability of solar radiation in South Africa. Fifty year time series (1950-1999) of mean daily solar radiation from Schulze (2007) were used to calculate the average solar radiation for each day of the year, using pivot tables in a spreadsheet. Radiation units were converted from MJm-2day-1 to Jm-2s-1, and three estimates of radiation were used in the model, viz. net radiation, and radiation modified by an albedo coefficient for clear and turbid water (after Paaijmans et al. 2008).

*Dam at head of segment* – Activation of this variable is governed by the logic that a dam at the head of a segment releases water with constant diel temperatures, which affects daily maximum temperature prediction (Bartholow 2002)

## Model testing and evaluation

Simulated water temperature data should ideally only be used after some form of data verification has been undertaken. The limiting factor in data verification is having available observed water temperature time series data which overlaps, to some degree, with the simulated data. Verification can either be qualitative, where time series of observed and simulated data are plotted together and interpreted based on site knowledge, or quantitative.

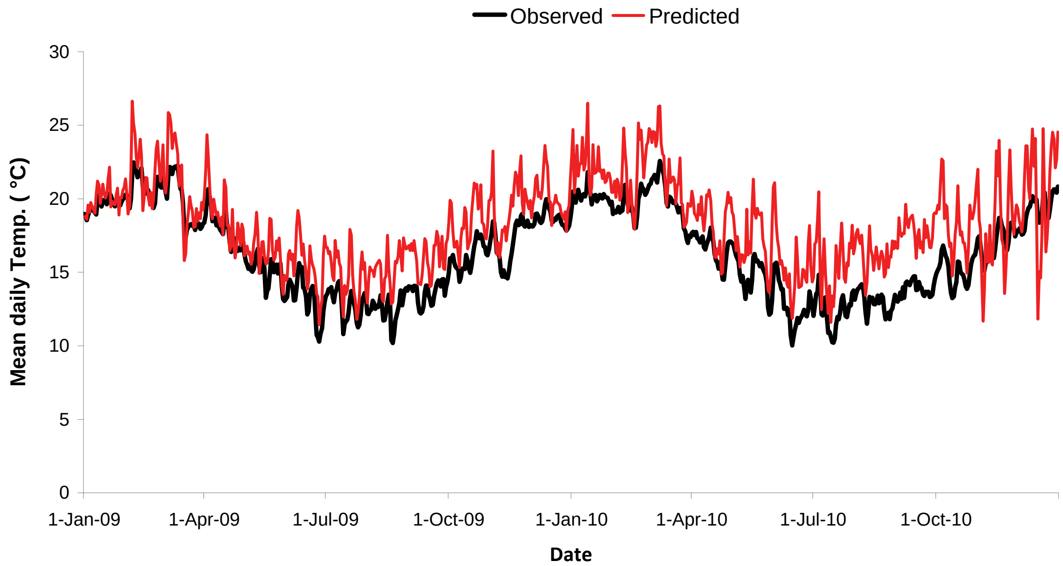



**Figure A5 Time series plot of observed and simulated mean daily water temperatures for qualitative model verification**

For quantitative model verification, various techniques are available to assess model accuracy between observed and predicted daily water temperatures. The first approach is to measure correlation between observed and predicted data, where the regression slope and $R^2$ values provide a measure of deviation of the residuals from the mean. The remaining three techniques all provide a measure of the mean value of the residuals, and have been used previously to evaluate water temperature models. Rivers-Moore et al. (2008b) calculated the mean percentage difference between observed and predicted water temperatures (Equation A8). A third approach is WSTAT (W-Statistic; Sullivan et al. 1990) (Equation A9), where a negative sign shows under-simulation, while a positive sign shows over-simulation. The fourth approach, similar to the WSTAT test, is to calculate the RMSE (root mean square error) (Equation A10) (also see, for example, Benyahya et al. 2009 citing Janssen and Heuberger (1995); Rivers-Moore et al. 2005).

$$\frac{\overline{P_i} - \overline{O_i}}{\overline{O_i}} * 100 \qquad [A8]$$

$$WSTAT = \frac{\sum_{i=1}^{N}(P_i - O_i)}{N} \qquad [A9]$$

$$RMSE = \sqrt{\frac{\sum_{i=1}^{N}(O_i - P_i)^2}{N}} \qquad [A10]$$

where N = number of daily water temperature observations, $O_i$ is observed and $P_i$ is predicted water temperature

### Techniques to check quality of input data

The use of continuous temperature time series in site comparisons, and as a tool in detecting thermal change over time, requires that the data be of good quality, and that the time series be analysed according to ecologically meaningful metrics.

### Note on nature of thermal regimes vs. flow regimes

A basic requirement for data used in, for example, Colwell's (1974) indices of predictability, particularly involving phenomena with fixed lower bounds, is that the standard deviation and mean are uncorrelated (Colwell 1974). Data that have a fixed lower bound (0), such as hydrological data, often show a high correlation between mean and standard deviation. Water temperatures do not generally have a fixed lower bound, but should still be tested for correlations between annual mean and standard deviation. The outcomes of these correlations determined the approach used in defining state classes. Data with non-significant correlations between the mean and standard deviation do not require transformation (Rivers-Moore et al. 2004).

### Data screening procedures

Continuous water quality data should not be used to calculate metrics before such a time series has been screened, and the limitations of its length and temporal resolution on the analyses understood. A time series of fixed length is a record of data representing a sample of the total population of values, and is therefore subject to statistical errors. For data to be useful in providing projections into the future based on past properties, data must have tolerable measurement errors (equipment failure), be of sufficient length to be representative, and be homogeneous (i.e. drawn from the same population) (Gordon et al. 1992).

The data screening procedure follows the following sequential steps:

- **Check for measurement errors** – this includes maintenance of equipment, and calibration of loggers
- Representativeness – The type of changes over time which can be detected by a programme of monthly samples for a ten year period may be very different from that with bi-weekly sampling for 25 years or daily sampling for five years (Esterby 1996, p.

127). Records of only a few years on length are not likely to be representative of the long-term variability at a site (Gordon et al. 1992). Records of 10-20 years are recommended as necessary to model short-term trends and seasonal components within the year, and encompassing at least one high and one low sequence (Gordon et al. 1992; Esterby 1996; Taylor et al. 2003). Sample size can also be calculated empirically (Equations A11-12).

$$n = c\left(\frac{s}{\bar{x}}\right)^2 \qquad [A11]$$

$$c = \frac{4}{\epsilon^2} \qquad [A12]$$

where *n* is sample size, *s* is standard deviation, *x* is mean, *c* is a factor where *ε* is the desired percentage error of the mean (Armour et al. 1983, cited in Gordon et al. 1992). A worked example from an Eastern Cape site on the Elands River (M1Eland-Cyph) is provided in Table A4, for which 392 days of mean daily water temperatures were available. Note that while the value of 100 days seems anomalous, this is purely a function of the relationship of the mean and the standard deviation. The higher the standard deviation, the closer the ratio of the mean to the standard deviation approaches one. From this may be derived the principle that the more variable a water temperature time series is, the larger the sample size required to be statistically meaningful. Thus, mid-reaches of rivers with a wide daily range require longer periods of sampling than stable, groundwater-fed headwater sections.

**Table A4 Values for variables necessary for estimated sample size at desired level of confidence**

| Variable | S | Mean | $\varepsilon^2$ (5% error) | c | n |
|---|---|---|---|---|---|
| Value | 4.58 | 18.35 | 0.0025 | 1600 | 100 |

- **Testing data for stationarity and absence of trend.** A dataset is stationary if its statistical properties (mean and variance) are unaffected by the choice of the time origin. There are two methods for trend analysis, viz.
  - Non-parametric approach (more robust), in which trend is assessed using Spearman's Rank-Correlation method.
  - Parametric approach, using normal least-squares regression

- **Testing for homogeneity** (i.e. the sample has been drawn from the same statistical population). A time series may stop being homogenous because the logging technology has been changed at some stage, or the equipment has aged and requires calibration. Two approaches apply here, viz.
- Using cumulative residuals, which is computationally complex, but does allow for testing for significance in homogeneity;
- Double mass plots, by plotting the cumulative data from site m versus site n (Figure A6). In the example provided, the double mass plots between two sites indicate no discontinuity in the data, and it can be safely concluded that the data are homogeneous. The approach is based on the logic that the relationship between the two sites exists as a straight line as long as the two sites' temperatures continue as a fixed ratio. It is recommended that each site be plotted against a "base station" dataset against which individual stations can be compared, and based on averaged values from a number of surrounding stations, preferably all from the same thermally homogenous region. Should the data be non-homogeneous, there will be a break in the regression line, the location of which can be estimated by plotting the residuals from the linear double mass plot. The less-homogenous data set should be corrected using linear regression where the driver variable is the base station (Gordon et al. 1992).

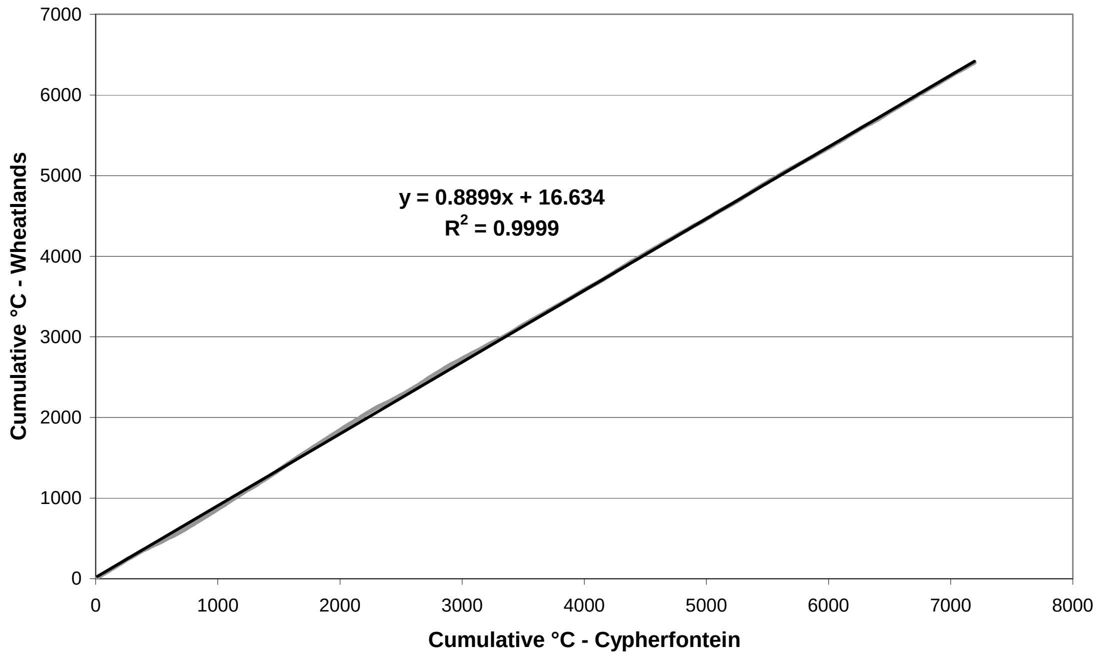



**Figure A6 Double mass plots for two sites on the Elands River in the Eastern Cape. Note that these data show a strong linear correlation, indicating that data are homogenous**

- **Test data for absence of persistence** (i.e. that the residuals are uncorrelated). In practice, the residuals are rarely uncorrelated, and there is usually some degree of serial correlation, especially if the data are collected in sequence (Abraham and Ledolter 1983). The absence of persistence can be tested for using serial autocorrelation (see Rivers-Moore et al. 2004).

Assuming the data meet these requirements, or have been corrected, the final step prior to using time series data is to patch the gaps in the data.

- Short gaps can be filled using simple linear interpolation;
- Bigger gaps can be filled using linear regression from a “base station” (see under double mass plots).

## Appendix 4. Decision tree: setting ecological reserves for water temperatures

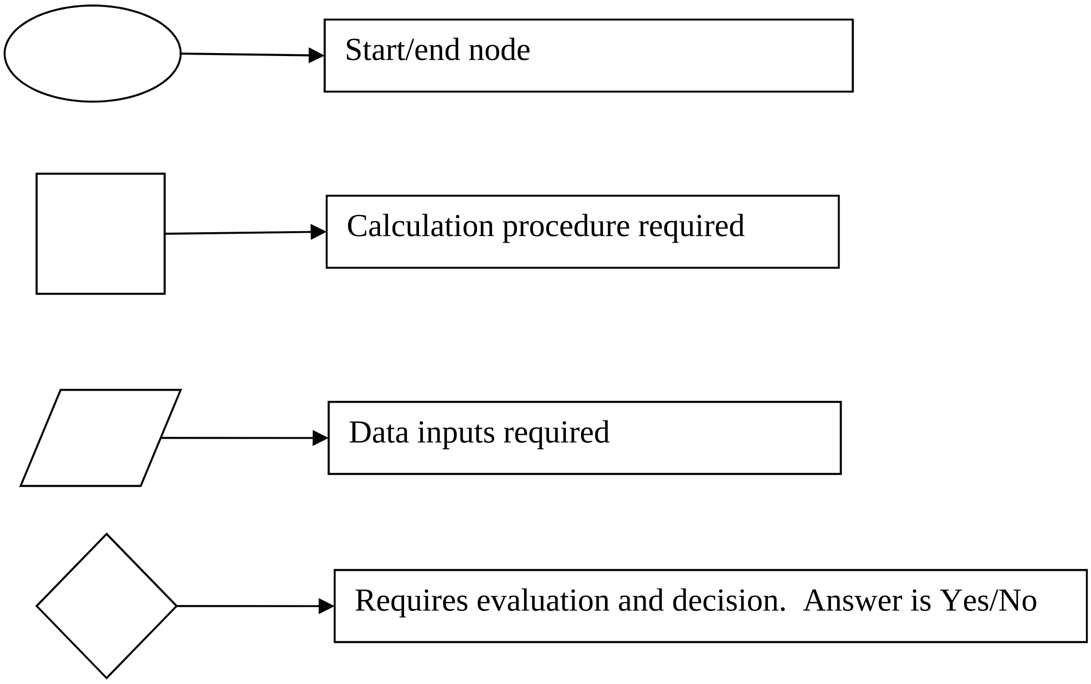
The flowchart presented below is an iterative process, where a Resource Manager is required to interact with each node of the flowchart to collect data for inputs, undertake calculations based on these data, and take decisions (Yes or No answers) from the outputs of these calculations. Determining whether the thermal ecological Reserve is met or not requires at least one year's sub-daily (one or two hourly) water temperature data for the site being assessed, and a comparable reference site from the same thermal region. Water temperature time series should ideally be based on observed data, but in their absence can be simulated using a suitable water temperature model.

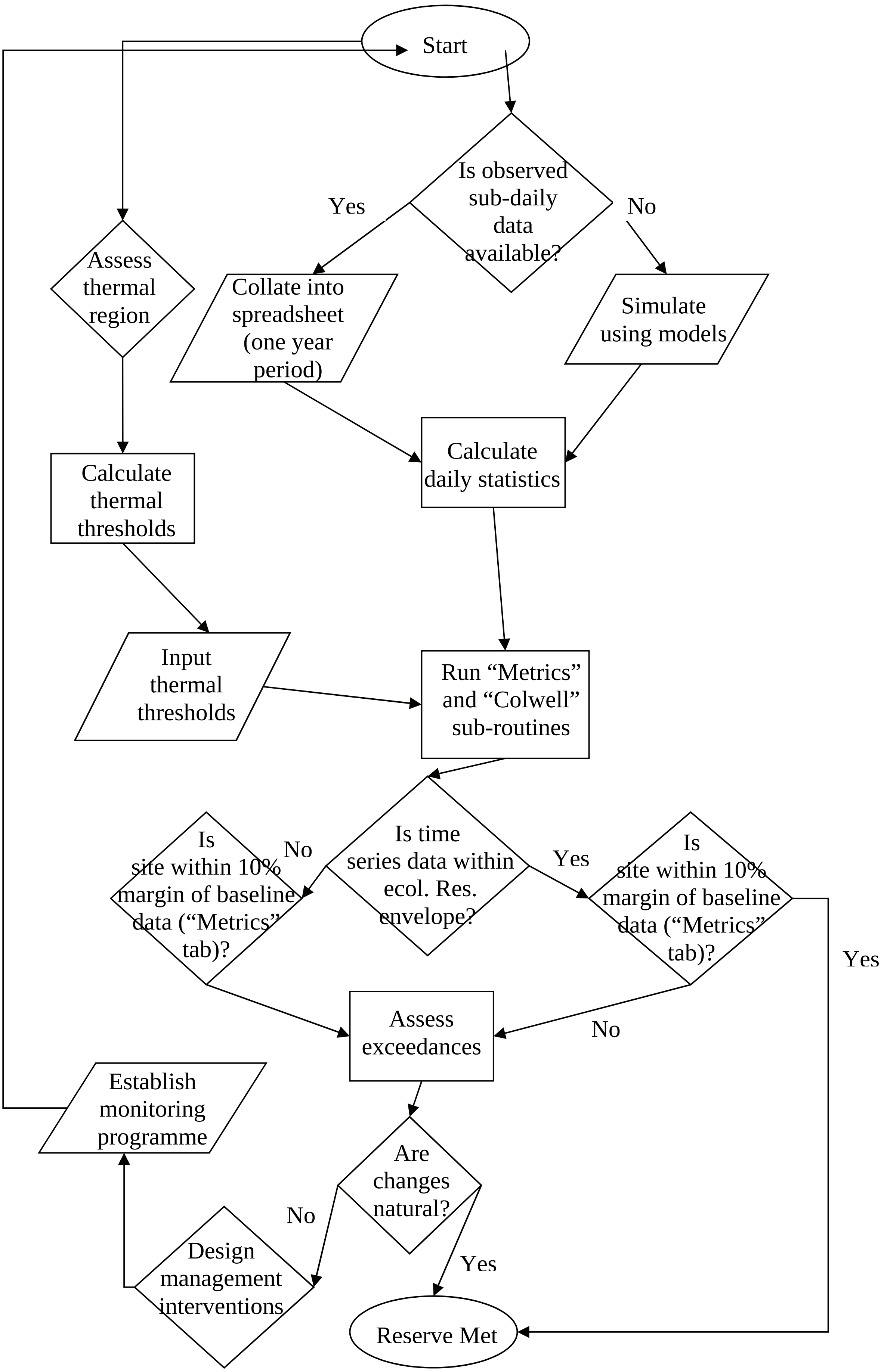
Once the data have been converted to daily data (mean, minimum and maximum daily water temperatures), thermal metrics relating to magnitudes of monthly temperatures, and frequencies, durations and timing of extreme thermal events are calculated. Included in such calculations is the determination of smoothed daily temperature means based on a 7-D moving average, and smoothed daily range using daily minima and maxima. These plotted time series are set within a 95% confidence envelope.

Assessing whether the thermal Reserve has been met or not is based on a two-step process of checking whether the thermograph from the site being assessed falls within the thermal confidence envelope, and whether the suite of metrics falls within a 10% range of the values of metrics from the reference site. If both steps fall within the recommended bounds, the site can be said to fall within acceptable thermal limits for the year of assessment, and based on the reference site data.

The thermal Reserve is not met if one/both steps indicate that values fall outside the expected thermal range from the reference site. Expert opinion and a site visit will be required to ascertain the likelihood of the Reserve not being met as a result of human-induced changes, or on natural trends. If departures from reference thermal conditions are shown to have a higher likelihood of being anthropogenic, it is strongly recommended that further investigation be undertaken to undertake the probable drivers of such change (hydrological, land use, etc.). Following on from these studies, managing intervention options should be assessed, while ongoing monitoring of water temperatures at the reference site and the site under investigation should continue.

In the decision tree below, the symbols mean the following:




Start
Is observed sub-daily data available?
Yes
No
Assess thermal region
Collate into spreadsheet (one year period)
Simulate using models
Calculate daily statistics
Calculate thermal thresholds
Input thermal thresholds
Run "Metrics" and "Colwell" sub-routines
Is time series data within ecol. Res. envelope?
No
Yes
Is site within 10% margin of baseline data ("Metrics" tab)?
Is site within 10% margin of baseline data ("Metrics" tab)?
Yes
No
Assess exceedances
Establish monitoring programme
Are changes natural?
No
Yes
Design management interventions
Reserve Met