THE HYDROLOGICAL BASIS FOR THE PROTECTION OF WATER RESOURCES TO MEET ENVIRONMENTAL AND SOCIETAL REQUIREMENTS

by

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VOLUME 1 OF 2

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PREFACE

The experimental work described in this dissertation was carried out in the School of Bioresources Engineering and Environmental Hydrology, University of KwaZulu-Natal, Pietermaritzburg, from July 2001 to September 2005, under the supervision of Professor Graham Jewitt.

These studies represent original work by the author and with the exception of (i) some material relating to the Background of the Mkomazi Catchment case studies and the *ACRU* model configuration of the Mkomazi Catchment in Chapter 6, (ii) some material relating Historical Flow Methods in Chapter 3 and (iii) Appendix 6A, have not been submitted in any form for any degree or diploma to any University. The material described (i) and (ii) above is reproduced from a Masters dissertation by the author, being, *Taylor, V. (2001). Hydrological modelling applications for water resources management in the Mkomazi Catchment. Unpublished MSc. Dissertation. School of Bioresources Engineering and Environmental Hydrology, University of Natal, Pietermaritzburg, RSA.* Appendix 6A is a revision of a chapter in that Masters dissertation. Appendix 6A in this thesis has been published in the Southern African Journal of Aquatic Science and is submitted here as an integral component of this thesis, based on its value to this study. Appendix 6A is reproduced with the kind permission of NISC Pty, Publishers, Grahamstown, South Africa.

Where use has been made of the work of others it is duly acknowledged in the text.

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* * * *

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"I'm truly sorry Man's dominion,
Has broken Nature's social union,
An' justifies that ill opinion,
Which makes thee startle,
At me, thy poor, earth born companion,
An' fellow mortal!"

Robert Burns, from "To a mouse", 1785

ABBREVIATIONS

ACRU	Agricultural Catchment Research Unit
Alt-BFI	Alterative Baseflow Index
ANR	Accumulated Natural Recapture Zone
BBM	Building Block Methodology
BEEH	The School of Bioresources Engineering and Environmental Hydrology, University of KwaZulu-Natal, Pietermaritzburg, South Africa
BFI	Baseflow Index
BHNR	Basic Human Needs Reserve
CD	Coefficient of Dispersion
CDB	Alternative Overall Variability Index
CMP	Catchment Management Plan
CSIR	Council for Scientific and Industrial Research
CV	Coefficient of Variation
CVB	Overall Variability Index
DPSIR	Driving Forces-Pressure-State-Impact-Response
DRIFT	Downstream Response to Imposed Flow Transformations
DWAF	Department of Water Affairs and Forestry
EEA	European Environment Agency
EFA	Environmental Flow Assessment
EFR	Environmental Flow Requirement
EIA	Environmental Impact Assessment
EFR	Environmental Flow Requirement
ER	Ecological Reserve
ERC	Ecological Reserve Category
ESA	Environmental Sustainability Assessment
FSR	Flow-Stress Response
HELP	Hydrology for the Environment, Life and Policy
HFI	High Flow Component Index
HPWS	High Precipitation Water Source Zone
IFR	Instream Flow Requirement
IHA	Indicators of Hydrologic Alteration

IWMI	International Water Management Institute
IWR	Institute for Water Resources, University of Rhodes, Grahamstown, South Africa
IWRM	Integrated Water Resources Management
LPWS	Low Precipitation Water Source Zone
MAF	Mean Annual Flow
MAP	Mean Annual Precipitation
MAR	Mean Annual Runoff
MMTS	Mkomazi-Mgeni Transfer Scheme
MNR	Mainstream Natural Recapture Zone
MPWS	Moderate Precipitation Water Source Zone
NWA	National Water Act
PC	Principal Component
PCA	Principal Component Analysis
QC	Quaternary Catchment
RIDe	Resource, Infrastructure, Demands and Entitlements
RDM	Resource Directed Measure
RDP	Reconstruction and Development Programme
RQO	Resource Quality Objective
RU	Resource Unit
RVA	Range of Variability Approach
SARES	South African Reserve Model
SEA	Strategic Environmental Assessment
SFRA	Streamflow Reduction Activity
TPC	Threshold of Probable Concern
TWP	Thukela Water Project
WBF	Wetted Bed Flow
WCD	World Commission on Dams
WISA	Water Institute of Southern Africa
WMO	World Meteorological Organisation
WRM	Water Resources Management
WSA	Water Services Act

GLOSSARY OF TERMS

The language and terminology of the environment is evolving rapidly. In particular, the language and terminology describing the South African Reserve is far from static and is continually being redefined. The main purpose of this Glossary is to explain as clearly as possible the scientific and sociological terms used in this thesis. Some definitions are wholly the reasoning of other authors and in such instances reference is duly noted. Terms are listed alphabetically; terms used in a definition and which appear elsewhere in the Glossary are printed in *italics*.

Biodiversity Essentially, biodiversity is the variety of living organisms, including animals, plants, fungi and micro-organisms, as well as the habitats which they occupy. Biodiversity is represented at different spatial scales, from globally defined major regions to the genetic level. Biodiversity is threatened by the impacts of socio-economic development, yet the benefits provided by biodiversity and social *well-being* are inextricably linked.

Disturbance In ecological studies, disturbance refers to the extent to which ecosystems are stressed beyond general, or average, conditions in their natural state or to any divergence from their natural state. For example, in the context of aquatic ecosystems, the natural streamflow regime includes both low flow and high flow episodes, which result in natural stress to the ecosystem's biota. While aquatic ecosystems have evolved to their natural disturbance regime, the ability of an ecosystem to function with stress episodes outside the natural range is a measure of its *resilience* to change (*inter alios*, Townsend, 1989).

Ecological component An ecological component refers to both biotic (living) and abiotic (non-living) attributes of an *ecosystem*, including soil and sediment, water and aquatic organisms.

Ecological functioning In the simplest terms, ecological functioning refers to the purpose of an *ecological component* in an *ecosystem*, or "what the ecological component does, or contributes, towards the *ecological integrity* of the ecosystem". For example, in aquatic ecosystems, the ecological functioning of a large infrequent flood pulse is to connect the stream channel with the stream terraces, banks and the fringing terrestrial ecosystem. In doing so, the *ecological processes* that bring about this functioning, inundate and deposit sediment on the floodplain and, in exchange, return nutrients in the form of organic detritus to the channel as the flood subsides (Junk *et al.*, 1989).

Ecological health Ecological health is used to describe the condition, or *ecostatus,* of a river that is altered from its natural state. Thus, ecological health is not the synonymous with *ecological integrity* (Karr, 1996).

Ecological integrity The ecological integrity of an ecosystem is generally understood to describe the wholeness of that system and its ability to continue to function in a natural way, implying correlation with its unimpaired or original state (Karr, 1996).

Ecological process An ecological process refers to the operation of the two basic principles governing the composition of an ecosystem, *i.e.* energy flows and nutrient cycling. For example, in the context of aquatic ecosystems, energy flows refer to, *inter alia*, the kinetic energy of water and sediment flows downstream as described by the River Continuum Concept of Vannote *et al.* (1980), whereas nutrient cycling refers to, *inter alia*, the uptake of nutrients from sediments by instream or riparian vegetation (the primary consumers) which, in turn, transform the nutrients and, together with the energy of sunlight, provide energy and nutrients for secondary consumers (*e.g.* fish, birds or mammals).

Ecological Reserve The Ecological Reserve (ER) relates to the quantity, quality and variable flow of water to protect the aquatic ecosystems of the water resource (DWAF, 2003). In South Africa the ER has become synonymous with the "Water Resource Base" and largely minimizes the importance of human "needs", since this is addressed in a separate Basic Human Needs Reserve. There is, however, increasingly more recognition of the dependence of humans and society on the ER as the foundation and provider of ecosystem functions, goods and services for economic growth.

Ecological Reserve Category An Ecological Reserve Category, ERC, refers to the *ecostatus* of a *Resource Unit* identified by a specialist team appointed by the South African Department of Water Affairs and Forestry, DWAF, to determine the *Instream Flow Requirements* of a water resource (IWR Environmental, 2003).

Ecological water requirements Ecological water requirements refer to the water that is left in an aquatic ecosystem, or released into it, to manage the health of the channel, banks, wetlands, aquifer, floodplains or estuary (Southern Waters, 2002). Ecological water requirements are generally assessed on a species basis. Such water allocation should include a human dimension, if only in recognition of the interrelationships and linkages between humans, other species and the *ecological processes* operating within *ecosystems*. Humans should / could be viewed as just another species within ecosystems which, in turn, have requirements from the water resource base.

Ecologically sustainable This implies environmental practice, including *environmental governance,* which protects the *ecological integrity* or *ecological health* of ecosystems while meeting the intergenerational human and societal needs for the *ecosystem goods and services* provided by fully functioning *ecosystems*.

Ecostatus In South African water resources management, and in particular in determinations of the *Ecological Reserve*, ecostatus is used to describe the state (historical, present or potential future) of aquatic ecosystem functioning. The ecostatus of different ecosystem components, as well as the overall system, can be measured and, as such, ecostatus is a reflection of *ecological integrity* or *ecological health* and provides a reference point against which any change in an ecosystem (or *ecological component*), from one state to another, can be measured.

Ecosystem This is the fundamental ecological unit (Lindeman, 1942), representing an interconnected and interacting system organised by energy flows and nutrient cycling and comprising living organisms and their non-living environment. Aquatic ecosystems are defined in the South African water quality guidelines (DWAF, 1996) as the "abiotic (physical and chemical) and biotic components, habitats and ecological processes contained within rivers and their riparian zones, reservoirs, lakes and wetlands and their fringing vegetation".

Ecosystem goods such as food, freshwater, natural fibre and biomass, fuels, crops, and many pharmaceuticals and their components are the products of *ecosystems*. Ecosystem goods are sometimes referred to as the short-term benefits of ecosystems (Baron *et al.*, 2002).

Ecosystem services are the "conditions and processes through which natural *ecosystems*, and the species they comprise, sustain and fulfill life" (Daily, 1997), including human and societal life. Ecosystem services can be classified in terms of their production, regulation, habitat as well as information functions (*de* Groot et al., 2002) and are sometimes referred to as the long-term benefits of *ecosystems* (Baron *et al.*, 2002). Ecosystem services "maintain *biodiversity* and the production of *ecosystem goods"* (Daily, 1997).

Environmental flow requirements Environmental flow requirements refer to the quantity and patterns of streamflows required to meet a particular state of *ecological functioning*. These streamflows may be required instream (*instream flow requirements*) or beyond the channel to inundate the floodplain, thereby ensuring *hydrological connectivity* between aquatic and terrestrial ecosystems.

Environmental governance This generally refers to environmental policy which addresses the interrelationships (both concurrences and conflicts) between the environment, people, society as well as governments and the market economy, and recognises these relationships as being the main pillars of sustainable development. Good environmental governance addresses the economic, social and political conditions that drive environmental decision-making. Environmental governance is a complex environmental perspective, much evolved from the traditional *stewardship* perspective of environmental issues.

Ephemeral Ephemeral rivers or streams are located in dry regions, with the majority of flow events occurring in direct response to precipitation. Flow events are frequently of short duration with some ephemeral rivers flowing only very rarely, after exceptionally heavy rainfall while others flow more frequently, with relatively regular, seasonally intermittent discharge (Boulton and Lake, 1988). Most of the streamflow in ephemeral rivers emanates from runoff generated on the catchment surface, with only small contributions from drainage from saturated soils (Hughes, 2000). However, the contribution from subsurface storage may form a larger component of the total flow at the end of a prolonged rainfall event (Hughes, 2000). In some regions, contributions to the stream channel may be sufficient to sustain pools of water, as a result of slow seepage from groundwater sources, even although high evaporation losses may prevent the generation of flow through the channel (Hughes, 2000). Ephemeral rivers are characterised by highly variable *hydrological regimes* as a result of flood pulses of short duration which cause "rapidly moving, longitudinal throughput" with steep rises and falls in the hydrograph (Jacobson, 1997). Thus, ephemeral rivers represent spatially and temporally patchily distributed *ecosystems*, providing goods and services maintained by flood pulses, which are the key to survival of many species in arid regions.

Hydrological connectivity In an ecological context, hydrological connectivity refers to "the water-mediated transfer of matter, energy and / or organisms within or between elements of the *hydrological cycle*" (Pringle, 2003). Hydrological connectivity is represented in different spatial dimensions (Ward and Stanford, 1989), *viz*:

(a) latitudinal connectivity which describes the linkages between the main river channel, the riparian zone and floodplain areas,

(b) longitudinal connectivity which is represented by the upstream-downstream relationship and is largely determined by the river network and

(c) vertical connectivity which describes the linkages between surface flows, baseflows and groundwater.

Hydrological cycle This refers to the circulation and conservation of the Earth's water and in simplest terms comprises the hydrological processes of evaporation, condensation, transportation, precipitation, storage, transpiration and runoff. The hydrological cycle "begins" with evaporation from the Earth's terrestrial and oceanic surfaces. This moisture condenses and accumulates as clouds in the atmosphere. Clouds are then transported by circulation patterns until they release the moisture as precipitation. When precipitation falls on terrestrial surfaces it either seeps through to groundwater storage or forms part of the surface water component of the hydrological cycle. Surface water is (a) released back into the atmosphere by plants through evapotranspiration processes or evaporation from upper soil layers and non permeable surfaces, or (b) forms runoff which collects in water bodies. In turn, these water bodies evaporate, thereby "initiating" the hydrological cycle (Bramer *et al.*, 2005)

Hydrological functioning Hydrological functioning refers specifically to the purpose of a hydrological component of an *ecosystem* or, in simplest terms "what the hydrological component does, or contributes, towards the *ecological integrity* of the ecosystem". For example, in aquatic ecosystems a major purpose of the baseflow component of the *hydrological regime* is to maintain the perenniality (where it exists) and seasonality of a *streamflow regime*. On the other hand, in terrestrial ecosystems a major purpose of the evaporation component of the *hydrological cycle* is to cycle nutrients and to fix energy into vegetative biomass through *ecological processes*.

Hydrological regime In the broadest sense, the hydrological regime represents the entire state of water movement in a given area. As such, the hydrological regime is a function of the climate. The hydrological regime of a stream includes the physiological (*e.g.* water, gas, temperature and energy exchange) and biological (*e.g.* nutrient cycling) processes in stream channels, banks, floodplains and hyporheic zones (Hynes, 1970; Poff, 1996). Hydrological regimes incorporate variation (both natural and unnatural) in these *ecological components* over time in response to different environmental conditions and are characterised by their different temporal patterns within a region. While stream hydrological regimes can be described by flow rates, flow volumes, baseflow and additional storm water flows, they are frequently described by flow duration curves. However, flow duration curves do not fully address flow variability. Hydrographs of daily time series of flows or monthly time series of seasonal distributions can be used to describe streamflow hydrological variability. Indices of flow variability are increasingly used in eco-hydrological studies to describe the hydrological regime.

Hydrological variation This refers to the fluctuation of the hydrological regime over time. For example, the magnitude, duration, timing, frequency and rate of change of water conditions vary (naturally and unnaturally) over time in response to different environmental conditions. In the *hydrological regime*, variation is a key selective pressure on aquatic and riparian organisms and a primary control on channel form and processes (Poff and Ward, 1990). *Hydrological and streamflow variation*, as a result of anthropogenic or climatic alterations, can result in stress to the ecosystem, with impaired *ecological functioning*, reduced *biodiversity* and reductions in the range of *ecosystem goods and services*. On the other hand, hydrological variation as a result of impoundment of streamflows for abstraction of water, one of the aquatic ecosystem goods and services, can result in reduced stress, particularly for human and societal freshwater requirements. However, maintaining the maximum volume of *ecosystem goods and services* requires streamflow quantities and patterns that closely resemble the intra- and inter-annual variability of the natural *streamflow regime*.

Hydronomic zone The concept of hydronomic (*hydro* water + *nomus* management) zones was developed by Molden *et al.* (2001) to describe conditions that may occur within areas, or zones, and for defining, characterising and developing water management and conservation techniques for areas with similar characteristics. Hydronomic zoning is applied to sub-divide catchments into units, or zones, which have different hydrological, physiographical, societal and economic relevance, in order that each zone may have a set of water management strategies. Molden *et al.* (2001) propose that a generic set of zones and strategies can be developed to characterise various conditions within a catchment. In their proposal Molden *et al.* (2001) address the fate of the water after use and, as such, their approach focuses on downstream use of water resources, with the emphasis on saving water. However, in essence, hydronomic zoning can be applied as a planning tool for catchments, based on a variety of conditions and approaches. For example, in this thesis hydronomic zoning is approached by how societal use interrupts the *hydrological cycle on the catchment*, and water management strategies are formulated in terms of a proposed framework for ecologically sustainable management, which may include trade-offs among hydronomic zones as well as rezoning.

Hyporheic zone This zone is the saturated and biologically active part in, and adjacent to, an alluvial river bed. It provides a habitat and refuge for aquatic organisms, thereby performing a buffering role which assists rapid recovery of aquatic ecosystems after *disturbances* such as floods and droughts (Xu *et al.*, 2002).

Instream flow requirements In South African water resources management, instream flow requirements refer to the quantity and patterns of streamflows required to meet a particular *ecostatus* for a *Resource Unit*.

Management class In South African water resources management, the present, historical and potential condition of a water resource, its importance, vulnerability and potential for restoration (*resilience*) are all factors which determine the management class and *Resource Quality Objectives* of a water resource (DWAF, 1999).

Perennial river Physiologically, a perennial stream has a well-defined channel containing water which flows throughout the year. The majority of perennial rivers are located in relatively wet regions with streamflow resulting from contributions from runoff, subsurface flow and groundwater seepage. The water occurring in perennial rivers in drier regions is often derived from streamflow generation in response to rainfall events in distant water source areas. In general, the flow regimes of perennial rivers are less variable than those of *ephemeral rivers*, with less steeply rising and falling hydrographs. In addition, certain chemical, hydrological and biological processes are characteristic of perennial rivers, including the presence of region, or site, specific biota.

Predictability Predictability is a measure of the propensity of natural or artificial systems to perform in a particular manner, over an interval of time. Predictability is linked with ecological *disturbance*. This attribute is often overlooked in favour of defining spatial or temporal *variability* in natural systems. However, the predictability of a system is related to the variation in frequency, recurrence interval or magnitude of an event. Events with low variance, or which occur regularly, are more predictable than those with higher variance or irregular patterns. As with *variability*, predictability has value as an indicator of both ecological and social systems, since the performance of an environmental variable can be driven by, and influence, societal conditions.

Protection In relation to a *water resource*, protection means maintenance of the ecostatus of the water resource so that it may be used in an ecologically sustainable way, and can include the prevention of the degradation of the water resource and the rehabilitation of the water resource (DWAF, 1999). Resource protection is a key objective of the South African National Water Act and is addressed through *Resource Directed Measures*.

Recharge This is a process which comprises the absorption and addition of water to the zone of saturation (Vegter, 1995), either via the soil profile or in the river channel, or by artificial means.

Reference Conditions In South African water resources management, reference conditions are used to define baseline conditions of natural resources (*e.g.* water or vegetation resources). Reference characteristics can be selected for "natural", or unimpacted, conditions or for present conditions, depending on the particular study. Where reference characteristics are selected to describe the natural conditions of a water resource unimpacted by human activity on the catchment or in the channel or aquifer, reference conditions describe the *ecological integrity* of a *water resource.* Reference conditions include water quality, water discharge and *recharge*, water level and the intra- and inter-annual variations thereof.

Reserve In South African water resources management, the Reserve refers to the quantity and quality of water required

(a) to satisfy basic human needs, by securing a basic water supply as prescribed under the Water Services Act (WSA, 1997) for people who are now, or who will, in the reasonably near future, be relying on, taking water from, or being supplied from the relevant water resource, and

(b) to protect aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant *water resource* (NWA, 1998).

Resilience The resilience of an ecological or social system is a measure of its capacity to absorb *disturbance* (*e.g.* extreme natural events such as earthquake, volcanic eruption and tsunami or major societal perturbation from social, economic or political disruption) while maintaining functioning. If a disturbance is sufficiently great, resulting in change to the composition and structure of the system, "resilience provides the components of renewal and reorganisation for the system" (Gunderson and Holling, 2002). The antithesis of resilience is *vulnerability* (Folke *et al.*, 2002).

Resource Directed Measures In South African water resources management, Resource Directed Measures refer to three essential steps involving the protection as well as the ecologically sustainable development and use of any relevant water resource (NWA, 1998), *viz*:

(a) Classifying the ecological, social and economic relevance or importance of the water resource

(b) Determining the *Reserve* at the spatial level of *Resource Units* for different *Ecological Reserve Category* scenarios, and

(c) Devising attainable *Resource Quality Objectives* to realise the *Reserve*.

Resource Quality Objectives In South African water resources management, Resource Quality Objectives refer to the specific objectives set to deliver the recommended *Ecological Reserve Category* for a *Resource Unit*. Resource Quality Objectives should be achievable, in an acceptable time frame, and are synonymous with the Best Management Practice concept applied globally in environmental management plans.

Resource Unit In South African water resources management, resource units are relatively homogenous areas within catchments, each of which is significantly different to warrant an Ecological Reserve determination. Resource units are based largely on ecoregions, which in turn are determined by a variety of physiographical, geomorphic, hydrological, and biological factors (DWAF, 2003).

Riparian This essentially means the habitat adjacent to rivers and streams, which is occasionally inundated or flooded. The proximity of water and alluvial soils ensures that riparian zones support plant species that are distinct from those of adjacent upland areas (Elmore and Beschta, 1987). Riparian areas are important within catchments since they provide a wide range of valuable *ecosystem goods and services*.

Stewardship In society there are different perspectives on the value of the environment, each of which is distinguished by different attitudes to the environment. Traditional attitudes to the environment range from strict conservationism to utilitarianism, whereas more contemporary attitudes range from deep ecology to radical environmentalism. The stewardship perspective of the environment is one of the most traditional attitudes and regards each species, including humans, as having a place, or niche, "in a divinely ordained scheme of things". This biblical or early conservationist perspective on the environment presents views on "how humans ought to treat other species in the environment they share". According to the stewardship perspective, other species "do not exist merely for humans to exploit", and indeed, humans "are accountable for their treatment of other species". Traditionally, the stewardship perspective of the environment regarded biodiversity as a measure of the richness of divine creation. However, this environmental perspective has developed to encompass the view that humans would wish to "preserve the species on which they depend" (all sourced from Silvertown and Sarre, 1990).

Streamflow Regime The streamflow regime comprises the quantity and rate of water flowing in a surface water body. It links many *ecological processes* in freshwater and riparian *ecosystems* and plays an important role in determining the structure (*e.g.* channel morphology), composition (*e.g.* occurrence and distribution of aquatic and riparian biota) and functioning (*e.g.* water quality, water temperature, transportation of sediment and organic matter, estuarine inflow and other environmental conditions) of aquatic systems (Junk *et al.*, 1989; Poff *et al.*, 1997; Richter *et al.*, 1996; 1997).

Variability Variability in *ecosystems, ecosystem components*, *ecological functioning* and *ecological processes* is the key to *biodiversity* and the basis of *ecological integrity* as well as societal *well-being*. In the context of aquatic ecosystems, variability in the *hydrological regime* (processes) and the *streamflow regime* (components) largely determines the instream and riparian biodiversity. Sustaining biodiversity and *ecological health*, so that aquatic ecosystems can generate the *ecosystem goods and services* that humans and society need, depends on maintaining a degree of the natural *ecological processes* and *ecological functioning.*

Vulnerability The propensity of societal or ecological systems to experience damage to its functioning from external pressures and stress (Folke *et al.*, 2002). Vulnerability is the product of a system's sensitivity to stress and its adaptive capacity to change (Kasperson *et al.*, 1995).

Well-being This is the human or ecosystem condition of being healthy or successful. Human well-being is measured in terms of quality of life or economic prosperity. Ecosystem well-being is measured in terms of ecological *resilience* and *ecological sustainability*. Fraser *et al.* (2005) define five human well-being categories (health and population, wealth, knowledge and culture, community, and equity) and five ecosystem well-being categories (land, water, air, species and genes, and resource use) to make up an overall well-being assessment.

Water resource The South African NWA (1998) defines a water resource as including a watercourse, surface water, estuary or aquifer.

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CONTENTS

PART III: SYNTHESIS

ABSTRACT

In common with other natural systems, aquatic ecosystems provide a wealth of economically valuable services and long-term benefits to society. However, growing human populations, coupled with increased aspirations for improved quality of life, have lead to intense pressure on the world's finite freshwater resources. Frequently, particularly in developing countries, there are both perceived and genuine incompatibilities between ecological and societal needs for freshwater.

Environmental Flow Assessment (EFA) is essentially a tool for water resources management and its ultimate goal should be the integration of ecological and societal systems. While other ecological components (*i.e.* biological and geomorphological) are equally important to EFA, this thesis investigates the role of the hydrological cycle and the hydrological regime in providing the ecosystem goods and services upon which society depends. Ecological and societal systems operate at different temporal, spatial and organisational scales and hydronomic zoning or sub-zoning is proposed as an appropriate water resources management technique for matching these different scales.

A major component of this thesis is a review of the South African water resources management framework and, in particular, the role of the Reserve (comprising a basic human right to survival water as well as an ecological right of the aquatic resource to maintain ecological functioning) in facilitating ecologically sustainable water resources management. South African water resources management is in the early stages of water allocation reform and the Department of Water Affairs and Forestry has stated that "the water allocation process must allow for the sustainable use of water resources and must promote the efficient and non-wasteful use of water". Thus, new ways of approaching the compromise between ecological and societal needs for freshwater water are required. This thesis argues that this requires that the focus of freshwater ecosystems be extended beyond the aquatic resource, so that societal activities on the catchment are linked to the protection of instream flows.

Streamflow variability plays a major role in structuring the habitat templates that sustain aquatic and riparian ecological functioning and has been associated with increased

biodiversity. Biodiversity and societal well-being are interlinked. However, there is a need in EFA for knowledge of the most influential components of the streamflow regime in order that stakeholders may anticipate any change in ecosystem goods and services as a result of their disruption to the hydrological cycle. The identification of high information hydrological indicators for characterising highly variable streamflow regimes is useful to water resources management, particularly where thresholds of streamflow regime characteristics have ecological relevance. Several researchers have revisited the choice of hydrological indices in order to ascertain whether some indices explain more of the hydrological variability in different aspects of streamflow regimes than others. However, most of the research relating to hydrological indices has focused primarily on regions with temperate climates. In this thesis multivariate analysis is applied to a relatively large dataset of readily computed ecologically relevant hydrological indices (including the Indicators of Hydrological Alteration and the South African Desktop Reserve Model indices) extracted from long-term records of daily flows at 83 sites across South Africa. Principal Component Analysis is applied in order to highlight general patterns of intercorrelation, or redundancy, among the indices and to identify a minimum subset of hydrological indices which explain the majority of the variation among the indices of different components of the streamflow regimes found in South Africa. The results indicate the value of including several of the IHA indices in EFAs for South African rivers. Statistical analysis is meaningful only when calculated for a sufficiently long hydrological record, and in this thesis the length of record necessary to obtain consistent hydrological indices, with minimal influence of climatic variation, is investigated. The results provide a guide to the length of record required for analysis of the high information hydrological indices representing the main components of the streamflow regime, for different streamflow types.

An ecosystem-based approach which recognises the hydrological connectivity of the catchment landscape in linking aquatic and terrestrial systems is proposed as a framework for ecologically sustainable water resources management. While this framework is intended to be generic, its potential for application in the South African Water Allocation Reform is illustrated with a case study for the Mkomazi Catchment in KwaZulu-Natal. Hydronomic sub-zoning, based on the way in which societal activities disrupt the natural hydrological processes, both off-stream and instream, is applied to assess the incompatibilities between societal and ecological freshwater needs. Reference hydrological, or pre-development, conditions in the Mkomazi Catchment are simulated using the *ACRU* agrohydrological model. Management targets, based on the statistical analysis of pre-development streamflow regimes, are defined to assess the degree of hydrological alteration in the high information hydrological indices of the Mkomazi Catchment as a result of different societal activities. Hydrological alteration from pre-development conditions is assessed using the Range of Variability Approach. The results indicate that the proposed framework is useful to the formulation of stakeholder-based catchment management plans. Applying hydrological records (either observed or simulated) as an ecological resource is highly appropriate for assessing the variability that ecosystems need to maintain the biodiversity, ecological functioning and resilience that people and society desire.

CHAPTER 1 GENERAL INTRODUCTION TO THESIS

CONTENTS

FIGURES

1 GENERAL INTRODUCTION TO THESIS

1.1 Introduction: Incompatibilities between Ecological and Societal Systems

In common with other natural systems, aquatic ecosystems provide a wealth of economically valuable services and long term benefits to society. Humans and society benefit from aquatic ecosystem services by using freshwater from rivers, springs, wetlands and lakes for many different agricultural, industrial, urban, household and recreational activities (Baron *et al.*, 2002). However, aquatic ecosystems are at the forefront of concerns for threatened natural systems, probably because the state or condition of these systems has direct impacts on the daily lives of people. In many parts of the world, growing human populations, coupled with increased aspirations for improved quality of life, have lead to intense pressure on the world's finite freshwater resources. Water resources and the state of their ecosystem goods and services have been likened to the "canary in the mine" and, as such, represent potential environmental indicators of the consequences of a range of human activities associated with demographics, planning, accessibility, resource utilisation and development (AWRA, 2004). Increasingly, there are heated disputes over access to the waters of rivers flowing through or along international boundaries, exploitation of water resources by some sectors to the detriment, or exclusion, of downstream users and water shortages resulting from both climatic and anthropogenic impacts. Thus, water-related conflicts span from international to local scale, occurring first in those areas that are water stressed (Ashton, 2003). While it may seem that such potential conflicts are inevitable, there is a case for proposing that water be used as a medium for peace (WWF, 2003), since the survival of the human species in an otherwise hostile natural environment depends on resolving any impending water crisis.

While climate variability, resource availability and social adaptability are major influences on the benefits provided by aquatic ecosystems (*c.f.* the DPSIR approach in Chapter 2 of this thesis), there are myriad examples of watershed mismanagement and unsustainable use of water resources (*e.g.* Gleick, 2000). Unsustainable use of freshwater has overlooked its value in supporting different ecosystems, including the aquatic environment (Baron *et al.*, 2002). Freshwater ecosystems act as centres of organisation within the landscape, providing natural resources as well as cultural and ecological services (Naiman *et al.*, 2002). Thus, the intimate linkage between terrestrial and aquatic ecosystems means that any land use management decision is also a water resource management decision, and *vice versa* (Falkenmark, 2001), emphasising the concept that human needs and culture and the environment should be viewed holistically. Ecosystem goods and services play multiple roles in water resources management by integrating economic, societal and ecological issues. Nonetheless, maximising the benefits of ecosystem goods and services requires an understanding of ecological functioning and the links between "natural capital", environmental benefits and the response of both natural and societal systems to reduced ecosystem functionality.

Issues over ecologically sustainable water resources management compound the "upstream-downstream" conflicts in socio-economic systems. It has been increasingly recognised that "freshwater ecosystems and the environment are legitimate users of water, requiring the same level of respect, advocacy and protection allocated to societal needs if water resource management is to achieve success" (Naiman *et al.*, 2002, page 455). Consequently, just as issues of "human rights" addressing societal welfare and fairness gained legal status in civil society, issues of "environmental rights" which are perceived to underpin ecological functioning and sustainability are gaining stature in the legislature of both developed and developing nations. Water related issues of availability, rights, use, regulation, quality and sanitation have high priority in the societal and environmental domains of both developing and developed regions, yet Falkenmark (2001) emphasises that "the inability to link environmental security, water security and food security" is the greatest water problem that society faces. Consequently, any assessment of ecological needs for freshwater, and the methods used in such assessments, have to be scientifically defensible, particularly where the environment and people are perceived to be in competition for scarce water resources. Given the increasing water demands of socio-economic systems there is a real threat that scientists will be pressured into recommending flows for the environment that are both too low and constant.

1.2 Biodiversity, Ecological and Societal Well-being are Interlinked

While there have been, and continue to be, genuine incompatibilities between ecological and societal systems, in both developed and developing nations there is growing recognition that both humans and ecosystems depend on the same water (Moberg and

Galaz, 2005). As well as maintaining a wide range of ecosystem goods which promote socio-economic benefits for people and society, functioning aquatic ecosystems maintain ecosystem services and biodiversity. Maintaining a high degree of biodiversity requires that aquatic ecosystems have a high degree of ecological health or well-being. Paradoxically, the greatest threats to ecosystem services (and consequently, biodiversity) result from a wide range of unsustainable human activities and from underestimating the value of ecosystem services to long-term, societal well-being (Daily *et al.*, 2005). The potential cost to society of the loss of any of the planet's ecosystem services is immeasurable. However, the message that biodiversity, ecological and societal well-being are inextricably interlinked is particularly evident in the increase of ecologically sustainable frameworks for aquatic resource management. The most advanced frameworks for ecologically sustainable water resources management have broken away from the premise that aquatic ecosystems are resilient to unlimited human stresses and disturbances and incorporate adaptive management strategies to safeguard both societal and ecological resilience (Moberg and Galaz, 2005).

1.3 Major Environmental Pathways and Societal Activity

Aquatic ecosystems are highly complex and variable and their management requires integrated approaches which recognise the interconnectivity of all the major biomes and the central role of humans and society as agents of environmental change. In addition, understanding the relationships within and among different ecosystem types and societal systems is complicated by issues of scale. Fraser *et al.* (2005) present a schematic (*c.f.* Figure 1.1) to assist policy makers and managers in understanding how disparate stakeholders are affected by resource management decisions. The approach in Figure 1.1 is based on established assessments that follow the pathways of chemicals which have been released into the environment (Lewis, 1901, cited by Fraser *et al.*, 2005) and has relevance for ecosystem management. The natural processes of general atmospheric and circulation patterns, nutrient cycling, water, sediment and energy flows are all interlinked, yet in Figure 1.1 human influence is shown as being central to the environmental pathways and inter-relationships. This factor reflects the growing interest in describing the relationship between people, society and their environment.

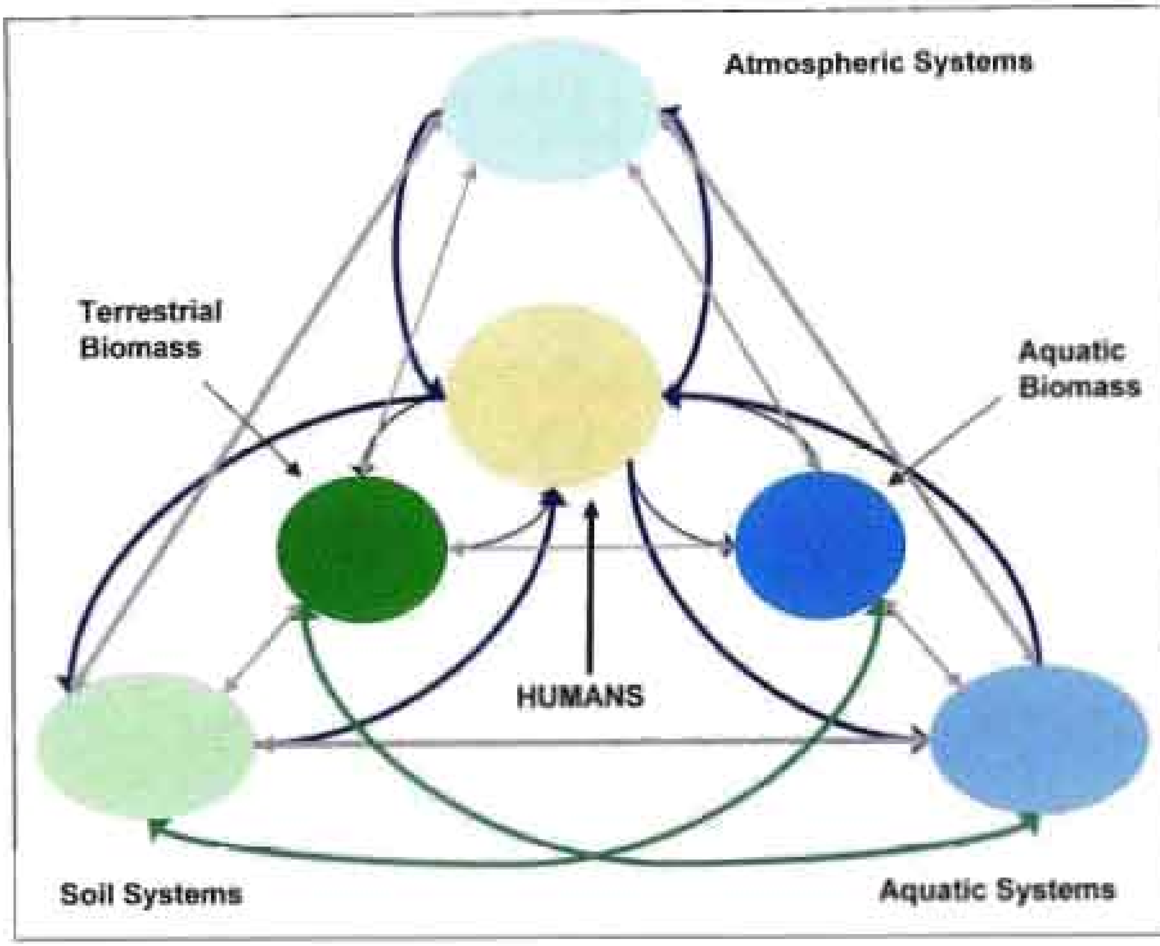

Figure 1.1 Schematic representations of environmental pathways for defining understanding and defining environmental factors that affect societal or institutional planning and management (after Fraser *et al.*, 2005)

Clearly, the best umbrella scale for water related problems is the watershed, yet the schematic also highlights the relevance of the "airshed" for atmospheric problems (*e.g.* climate change) and the connectivity of agricultural tracts, or corridors, for terrestrial problems (*e.g.* diffuse pollution and pest dispersal) Fraser *et al.* (2005). However, as suggested by Fraser *et al.* (2005), the crux of the schematic in Figure 1.1 is the indication of the major environmental pathways *through the landscape*. This acknowledgment is imperative for the management of aquatic ecosystems, since the source of an aquatic problem or disturbance can be far beyond the stream channel.

1.4 Hydrological Connectivity

The hydrological cycle plays a critical role in linking the major biomes by providing life-sustaining services. Society relies on the hydrological cycle not only for water supplies, but also for a wide range of ecosystem goods and services, many of which are hidden, undervalued and easy to take for granted (Postel and Carpenter, 1997). The various components of the hydrological cycle (condensation, transportation, precipitation, infiltration, evapotranspiration, and runoff) are linked through water, chemical, biological and sediment flows and energy transfer at different temporal and spatial scales by hydrological connectivity. Hydrological connectivity plays an important role in the ecological integrity of the landscape at the global, regional and local scales, yet the capacity of humans to change hydrological connectivity can have major, often cumulative, environmental effects which are temporally and spatially removed from their source (Pringle, 2003). Some of these effects are better understood than others and Pringle (2003) cites the example that whilst the effects of altered hydrological connectivity by dams on fish migration routes and the reduction of nutrient recycling in entire riverine ecosystems are well known, "the indirect biogeochemical effects are more elusive and difficult to identify". Given the frequent paucity of information on how hydrological connectivity between different components of the hydrological cycle and the landscape structures both terrestrial and aquatic ecosystems, Pringle (2003, page 2688) highlights the need for interdisciplinary research between hydrologists and ecologists "to understand how cumulative human alterations of hydrological connectivity influence ecological patterns".

While the complexities and dynamics of aquatic ecosystem functioning are uncertain over both time and space, the basic ecosystem functioning principles whereby energy *flows* through ecosystems and minerals *cycle* through them are a good foundation to understanding how the hydrological processes and functioning of aquatic, riparian and terrestrial ecosystems are intimately linked. These basic ecosystem principles are applied in this thesis by focusing on how people and society interrupt the hydrological processes and functioning which operate over the catchment.

While Baron *et al.* (2002) describe five major driving forces that influence freshwater ecosystems, *viz.*, the streamflow regime, sediment flux, chemical / nutrient flux, sunlight and biotic assemblage, (*c.f.* Figure 1.2), the *hydrological processes* associated with water, vapour and sediment flows together with *hydrological functions* (*i.e.* habitat formation and inundation; channel, terrace and floodplain maintenance; transport and deposition of sediments and nutrients) are the two major drivers of the biodiversity of aquatic ecosystems (Townsend, 1989).

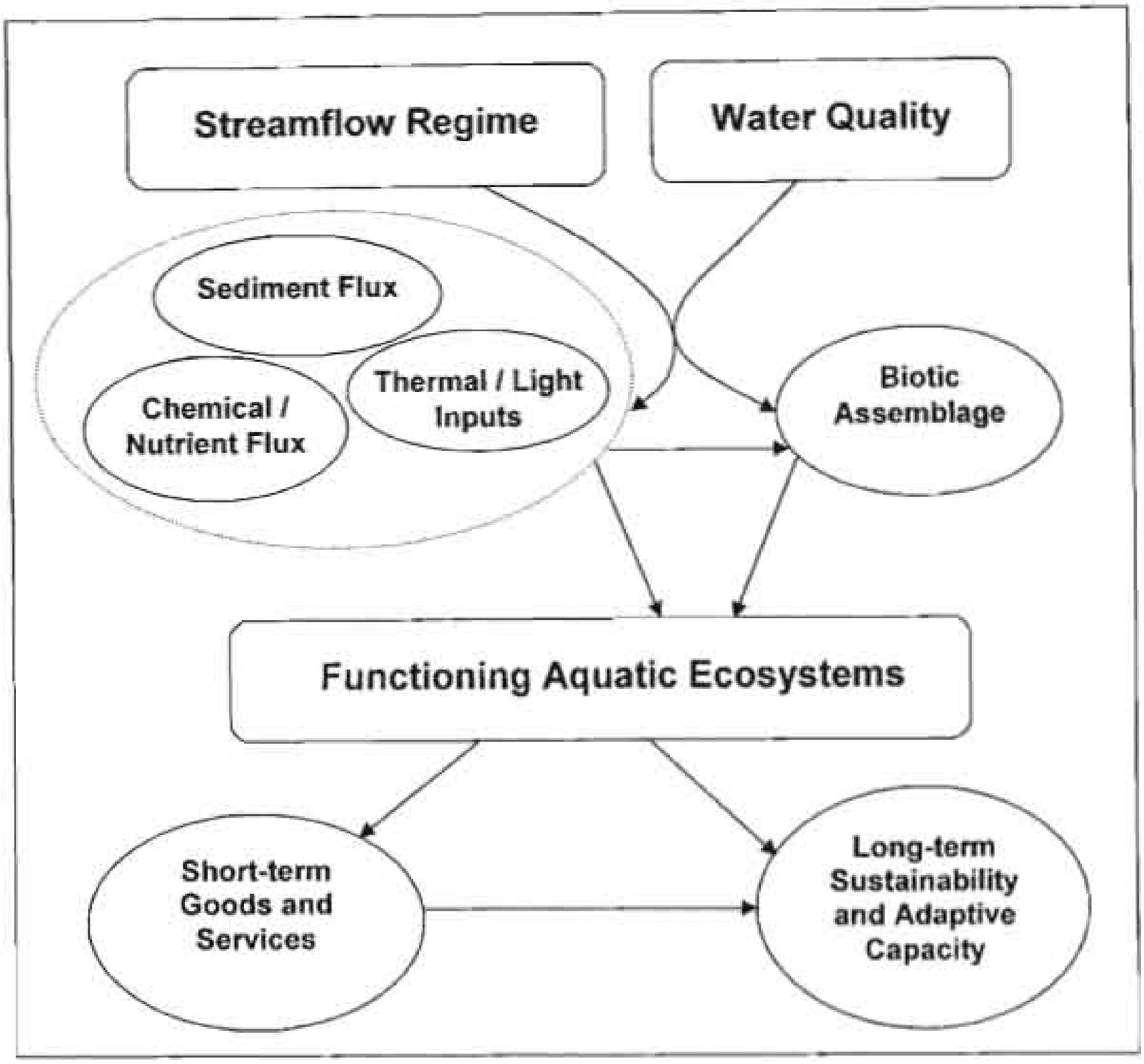

Figure 1.2 Conceptual model of major driving forces that influence freshwater ecosystems (after Baron *et al.*, 2002)

These ecosystem processes and functions have evolved with the natural streamflow regime and are adapted to the natural disturbance regime, including the natural range of both low flow and high flow components (Poff *et al.*, 1997). Human and societal use of aquatic ecosystems, and of the benefits provided by those ecosystems, interferes with the natural disturbance regime and, depending on the resilience of the system, continued use, or over-use, can cause an ecosystem to enter a state of dynamic flux. Unstable ecosystems are unable to function properly, with the effect that ecosystem integrity is lost, and the health of the system, or its ecostatus, becomes impaired. Any change in the ecostatus of an aquatic ecosystem translates into changes in ecosystem goods and services available for humans and society. Thus, methods which measure any change in ecostatus are invaluable to ecologically sustainable water resources management.

1.5 Linking Major Environmental Pathways and Aquatic Ecosystem Functioning

This thesis argues that ecological and societal well-being and biodiversity are inextricably interlinked. Aquatic resources are critical to human development, but their water resource management needs to be operated in an integrated, adaptive and participatory manner in order to ensure a high degree of ecological health. Naiman *et al.* (2002) emphasise the fundamental links between climate, land and freshwater in defining the relationships between physical and ecological processes and pose three ecological principles guiding the maintenance of long-term ecological health:

(a) "The natural flow regime shapes the evolution of aquatic biota and ecological processes;

(b) every river has a characteristic flow regime and associated biotic community; and

(c) aquatic ecosystems are topographically unique in that they occupy the lowest position in the landscape, thereby integrating catchment scale processes". (Naiman *et al.*, 2002).

The research comprising this thesis considers and reviews these principles, together with the following additional guiding principles that promote ecologically sustainable water resources management:

(d) The resilience of a river's flow regime links ecological functioning with societal well-being and thresholds of resilience can be identified from the natural flow regime.

(e) The ecosystem goods and services generated by the hydrological cycle can be "saved for a dry day", *i.e.* they provide society with future options, since ecologically sustainable water resources management ensures that compromise is reached between ecological and societal needs for freshwater by deferring the full utilisation of the benefits of both ecosystem goods and services.

1.6 Aims of this Thesis

The main thrust of this thesis relates to the management of water resources so that society can continue to receive the ecological benefits provided by functioning aquatic ecosystems. The purpose of this section is to define the aims of this thesis (*i.e. what* the thesis sets out to achieve) and the objectives formulated to meet those aims (*i.e. how* the aims are addressed). Accordingly, the general aims of this thesis are:

(a) to identify the interrelationships between the hydrological cycle, the ecosystem goods and services it supports and the societal mechanisms that influence ecological integrity (*c.f.* Objective 1 in Figure 1.3);

(b) to identify an approach to link the societal mechanisms that influence the ecological integrity of freshwater ecosystems at different spatial, temporal and organisational scales (*c.f.* Objectives 2, 3 and 4 in Figure 1.3);

(c) to complement the Resource Directed Measures in the South African water resources management process (*c.f.* Objective 3 of Figure 1.3);

(d) to assess the ecological significance of highly variable streamflow regimes and the societal consequences of their management as a resource for ecosystem goods and services (*c.f.* Objectives 1, 2, 3, 4 and 5 of Figure 1.3); and

(e) to apply the approach identified in aim (b) to typical socio-ecological systems and environmental concerns in water resources management policies of South Africa (*c.f.* Objective 5 of Figure 1.3)

1.7 Outline of this Thesis

The text of this thesis comprises a series of chapters that have been written essentially as "stand alone" papers with the intention that they be submitted for publication in internationally distributed literature. Given this structure, there is some (yet relatively limited) repetition among the chapters of the key philosophies underlying the study. This repetition is unavoidable and has been included for clarity.

This thesis has been organised into three main parts. **Part I** comprises three chapters (Chapters 2, 3 and 4) which focus on the information that water users need in order to make informed decisions relating to their utilisation of not only water resources, but also

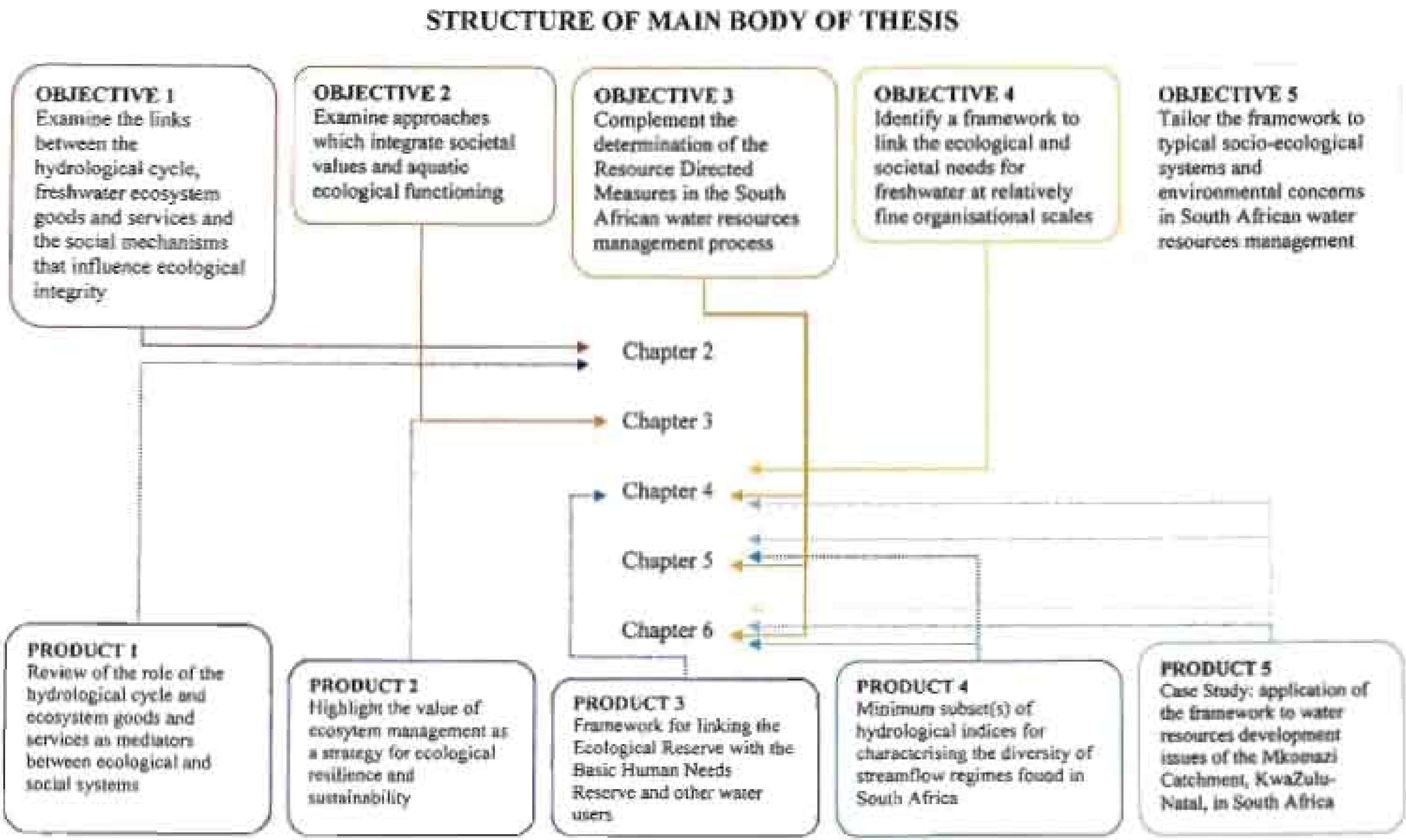

Figure 1.3 Structure of main body of thesis

the impact of their activities beyond the stream channel, on hydrological connectivity, via the major environmental pathways.

Chapter 2 represents a review of contemporary understanding of the interrelationships between the hydrological cycle, the ecosystem goods and services it supports as well as aquatic ecosystems and societal systems. The ecological basis for the protection of water resources is considered before investigating the ecological benefits of the integral components and characteristics of the hydrological regime. The disparities between the spatial, temporal and organisational scales of ecological systems and those of societal systems, and the role of the hydrological regime in linking the mismatches, are highlighted.

Following this broad introduction, ***Chapter 3*** reviews the assessment of societal and ecological water requirements based on ecological functioning and the generation of ecosystem goods and services. The focus of Chapter 3 is that (1) reserving water for ecological functioning is essentially about maintaining ecological *and* societal well-being, (2) the "value" of ecosystem goods and services integrates societal and ecological water requirements and (3) ensuring ecological as well as societal resilience requires ecosystem management approaches which engage adaptive management and participatory processes.

The focus of ***Chapter 4*** is the ecologically sustainable development and use of South Africa's water resources. South African water law was radically reformed in the latter part of the 1990s, culminating in the National Water Act (NWA) of 1998 which legitimised the environment as a priority "water user" with an "Ecological Reserve" set aside to meet aquatic ecological functioning at a prescribed "ecological reserve category" (NWA 1998). In assessments of "allocable" water for licensed use, the Ecological Reserve enjoys the highest priority save for the allowance of a "Basic Human Needs Reserve" required by humans for essential drinking, washing and cooking water. The principle of an Ecological Reserve has raised some very valid questions in water management, stakeholder and research groups. Not least of these are the different perceptions of the entity of an Ecological Reserve among different sectors of society (van Wyk *et al.*, 2006). The entire philosophy of "reserving" water to sustain the environment that we wish now as well as for future use may be jeopardised by misunderstanding the ecological, societal and management basis behind the concept. This is likely to become all the more relevant in light of the Department of Water Affairs and Forestry's proposed Water Allocation Reform process (DWAF, 2005), particularly in water-stressed catchments. The societal and environmental concerns relating to the protection and utilisation of the nation's water resources are explored in Chapter 4. More pertinently, Chapter 4 supports the identification in Chapter 3 of a need for ecosystem-based approaches to water resources management and presents a generic framework for ecologically sustainable water resources management. The principle aims of the framework presented in Chapter 4 are that (1) the influence of societal activities in catchments is inevitable and (2) ecological and societal needs for freshwater need not be in competition.

Part II of this thesis comprises the methods developed to explore the hydrological basis for the protection of aquatic ecological functioning so that societal functioning can be sustained. Part II focuses on different environmental issues relating to variable streamflow regimes, and presents a case study for the Mkomazi Catchment, a relatively undeveloped catchment with no major impoundments, in KwaZulu-Natal, South Africa.

Chapter 5 serves as an introduction to Part II of this thesis and describes ecologically relevant hydrological indices of rivers with variable flow regimes. There is growing awareness of the pivotal role of the streamflow regime as a key driver of the ecology of rivers and floodplain ecosystems (Junk *et al.*, 1989; Poff *et al.*, 1997; Richter *et al.* 1997). However, while aquatic ecologists are conversant with the principles regarding the relationships between streamflow and aquatic biodiversity, they still face major difficulties in predicting and quantifying biotic responses to altered streamflow regimes (Bunn and Arthington, 2002). Chapter 5 examines the use of long-term hydrological records as an "ecological resource" and identifies key hydrological indicators for the management of the variable river systems found in South Africa. Hydrological indicators are useful for many reasons: they identify the state of a river system in terms of quality and quantity and allow for its evaluation over different time and spatial scales. Meaningful, ecologically relevant hydrological indicators which provide information on societal and ecological needs also facilitate communication between policy makers and stakeholders. Given the different levels of South African Reserve Classification of water resources (ranging from "natural" to "severely modified", *c.f.* Chapter 4), each of which represents the costs and benefits of various ecosystem goods and services to different users at different scales, there is a need to identify hydrological indices which adequately describe recognised, critical components of the streamflow regime and which are readily understood by a diverse group of people.

Together ***Chapter 6*** and ***Appendix 6A*** form a Case Study on the development potential of the Mkomazi Catchment in KwaZulu-Natal, South Africa. The Mkomazi Catchment is relatively undeveloped, is sparsely populated and presently (2006) and has no major impoundments. As a consequence, the Mkomazi River presently has a high ecostatus. In addition, the Department of Water Affairs and Forestry's streamflow gauge in the mid to upper portion of the catchment records "reasonably natural flows" (*c.f.* Chapter 5) over a relatively long period (from 1960 onwards). Chapter 6 explores the hydrological issues of the tributaries of the Mkomazi Catchment and applies the framework presented in Chapter 4 together with the key hydrological indices of variability, identified by following the procedures outlined in Chapter 5. Appendix 6A explores the hydrological issues relating to the mainstream Mkomazi River.

Finally, **Part III** of this thesis discusses the relevance of the entire study to the management and conservation of rivers with variable flows. Since Chapters 2 to 6 indicate that societal well-being and biodiversity are inextricably linked, ***Chapter 7*** focuses on the challenges of meeting ecologically sustainable water resources management. The purpose of ***Chapter 8*** is to summarise the main findings of the thesis.

Plans containing the structure of the main body of the thesis, and individual plans for each of Chapters 2 through Chapter 6 and Appendix 6A are provided for clarity and to guide the reader through the content of this thesis. In each case the objectives, relating to the thesis in general, or specific objectives (for Chapters 2 through 6 and Appendix 6A) are stated together with descriptions of the products relating to the thesis in general, or specific products (also for Chapters 2 through 6 and Appendix 6A), with the interlinkages shown for each Chapter and Section. The specific objectives and specific products relating to Product 5 of the thesis (the Mkomazi Case Study) are distributed over Chapters 6 and Appendix 6A.

In addition, a Glossary of terms has been compiled and included (*c.f.* Pages viii to xxi), since throughout the development of this thesis it became evident that there exists a great deal of ecological terminology that is used by non-ecologists. Language and terminology are well-known constraints in interdisciplinary research, since among scientists and stakeholders terms can be used and understood differently. Moreover, the language related to the environment is evolving rapidly. In particular, the language describing the South

African Reserve is far from static and is continually being redefined. The main purpose of the Glossary is to explain as clearly as possible the scientific and sociological terms used in this thesis.

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CHAPTER 2 THE HYDROLOGICAL CYCLE AND THE GENERATION OF ECOSYSTEM GOODS AND SERVICES

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STRUCTURE OF CHAPTER 2 THE HYDROLOGICAL CYCLE AND THE GENERATION OF ECOSYSTEM GOODS AND SERVICES

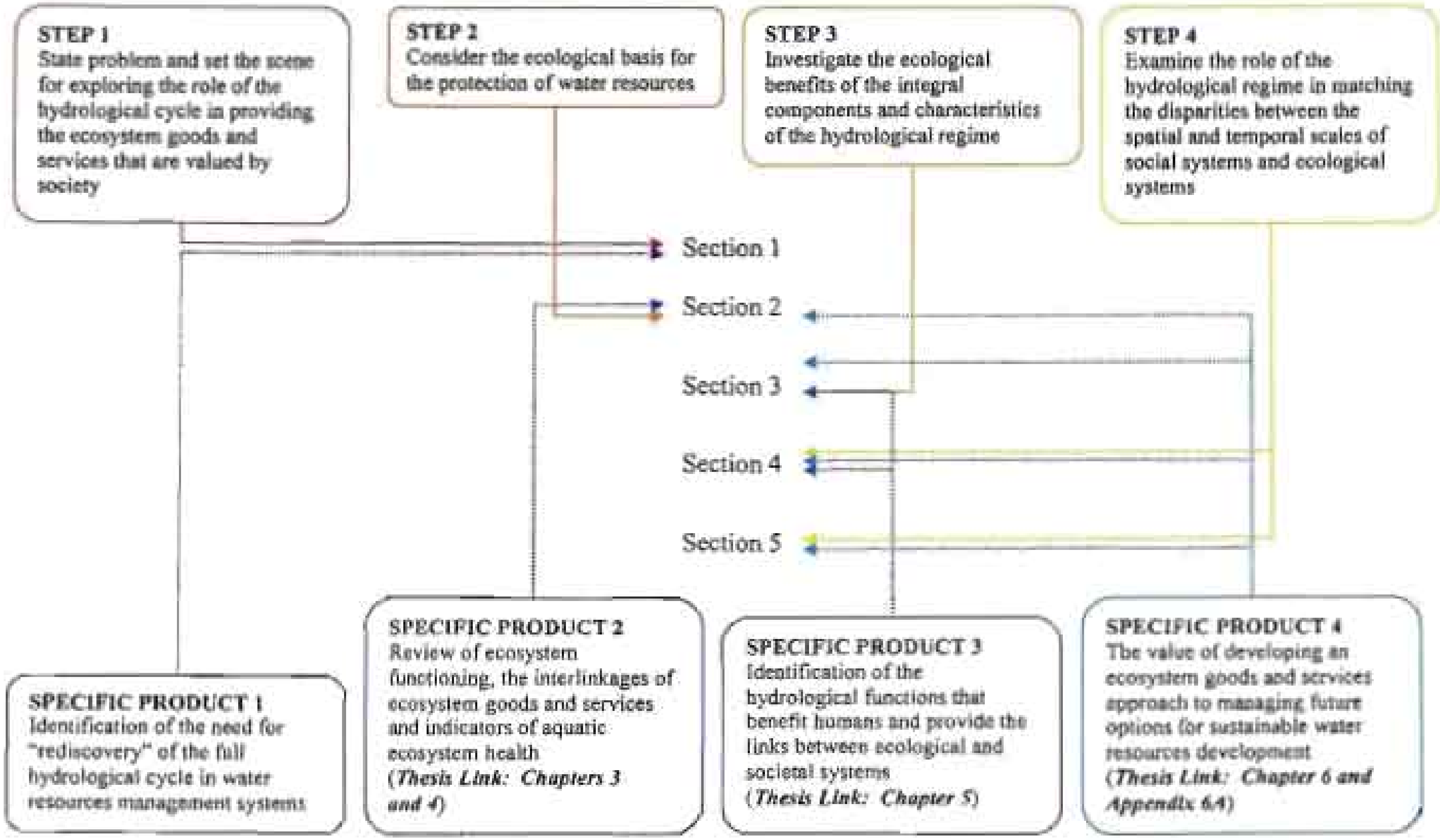

Figure 2.1 Structure of Chapter 2: The hydrological cycle and the generation of ecosystem goods and services

CHAPTER 2 THE HYDROLOGICAL CYCLE AND THE GENERATION OF ECOSYSTEM GOODS AND SERVICES

1 INTRODUCTION

There is an intimate relationship between human societal systems and the ecological systems upon which they depend for prosperity, economic growth, and global exchange of capital and resources (Daily, 1997). However, humans are using natural resources at unprecedented scales, altering ecosystems, hydrological and biogeochemical cycles and the environment at large. Globally, the immediate focus of water resources management is to meet the water-related needs of society while ensuring that there is sufficient water to realise economic potential, without degrading the water resource base (IWMI, 2002). Aquatic ecosystems have functions that provide goods (*e.g.* water for direct consumption) and services (*e.g.* nutrient cycling) to people (Mander and Quinn, 1999). Worldwide, the human demand for ecosystem goods and services continues to intensify as agricultural, domestic and industrial uses increase to meet the aspirations of burgeoning populations. Increasingly, there is recognition of the need to allocate water to sustain the environment and the ecosystem goods and services that are highly prized by society.

Worldwide, and particularly in arid or semi-arid regions where the scarcity of water as a renewable resource is more acute, the variability associated with climatic and hydrologic regimes continues to dominate the challenges that water managers face in balancing the needs of humans and their social systems with those of ecological systems. In particular, this has led to a quest to ascertain how much water aquatic systems need to maintain their physical and biological functioning and to ensure that thresholds of ecosystem sustainability and resilience are not exceeded through over utilisation. Deciding how much water should be reserved for the maintenance of ecosystems to provide "natural" goods and services and how much water should be used by agriculture and industry to provide "artificial" goods and services necessitates some measure of Environmental Flow Assessment (EFA). More frequently, the EFA philosophy and concepts are being recognised as tools for addressing flow requirements at all relevant spatial scales, from river reach to basin, with instream and beyond the stream channel flows assessed in an "ecosystem approach" to integrate ecological and societal functioning (King *et al.*, 2000; Marchand *et al.*, 2002). Consequently, EFAs form integral components of water resource planning and river basin management and have increasingly moved towards more holistic approaches, integrating the needs, as well as the impacts, of societal and economic systems on the water resource (Tharme, 2003).

However, determining flows to sustain physical and biological processes of river systems, groundwater, wetlands and estuaries presents major difficulties to scientists, stakeholders (water users) and water managers. Aquatic ecosystems are complex, with many interlinked components, and are susceptible to alteration resulting from a range of human activities. There is a need to understand how changes to the hydrological cycle change an ecosystem's ability to sustain ecosystem services, so that indicators of environmental quality and of potential, or irreversible, damage to ecological systems can be developed. These indicators could provide the information that catchment stakeholders need to make decisions about the future utilisation of a water resource.

In recognition of the linkages between terrestrial and aquatic ecosystems and atmospheric water, Jewitt (2001) highlights the need for rediscovery of the full hydrological cycle and water resources management systems in order to achieve sustainable use of water resources and the maintenance of ecosystem services. The biophysical processes influencing the movement of water at the land and water interface as runoff, baseflow and groundwater recharge, as well as the flow of water vapour as evapotranspiration, need to be better understood, spatially, temporally and organisationally, in order to adopt an ecosystem approach to the management of water resources.

The purpose of this Chapter is to review the current understanding of the interrelationships between the hydrological cycle, the ecosystem goods and services it supports and societal systems. This Chapter explores the role of the hydrological cycle in providing the ecosystem goods and services that are valued by society. The ecological basis for the protection of water resources is considered before investigating the ecological benefits of the integral components and characteristics of the hydrological regime. The disparities between the spatial, temporal and organisational scales of ecological systems and those of societal systems, and the role of the hydrological regime in linking the mismatches, are highlighted.

2 ECOSYSTEM GOODS AND SERVICES AND INDICATORS OF AQUATIC ECOSYSTEM HEALTH

2.1 Introduction

In order to achieve environmental security, stakeholders need to know how their utilisation of resources (natural and artificial) correlates with the quality and quantity of ecosystem goods and services that can be expected under different streamflow regimes. This requires some measure of baseline conditions so that desired future conditions can be negotiated among the various water users. It is, however, useful to first consider the ecological basis for the protection of water resources.

2.2 Ecosystems: Functioning, Linkages and Humans

An ecosystem can be considered as a level of ecological organisation, with emphasis on the links between species and their physical resources (Silvertown, 1990). However, the traditional concept of an ecosystem, whereby energy is supplied to primary producers, and then flows to higher trophic levels before being recycled by the mineral pathway within a fairly self-contained complex, is not valid to aquatic ecosystems (Hynes, 1970). For rivers, wetlands and estuaries, any flow of energy and cycling of nutrients is displaced in a downstream direction and also spreads out over the floodplain and estuaries of larger rivers when inundation occurs. McCartney *et al.* (1999) describe freshwater ecosystems as comprising features that can be classified as components, functions and attributes (Box 1).

Two basic concepts are regarded as forming the foundations for understanding the spatial and temporal interactions of the ecological functioning of natural river ecosystems. The ***river continuum concept*** (Vannote *et al.*, 1980) describes a linear and longitudinal organisation of the physical habitat template and biological productivity of the river channel based on stream order, whereby nutrients and sediments generated upstream are recycled and drive primary production downstream. The ***flood pulse concept*** (Junk *et al.*, 1989) describes the connectivity of the river channel to the riparian zone and floodplain, whereby inundation of the floodplain flushes out accumulated salts and brings in nutrient rich silt. Hence, the flood pulse introduces a lateral component (and an annual cycle) to the dynamics of lotic systems, extending the spatial focus beyond the main channel (Johnson *et al.*, 1995).

Box 1 Features of freshwater ecosystems *(McCartney et al., 1999)*

The ***components*** of the ecosystem are the biotic and non-biotic features, including soil and sediment, water and aquatic organisms.

The interactions between the components comprise the hydrological, biological, chemical and physical processes that result in ecosystem ***functions*** such as evaporation, respiration, photosynthesis, retention of water, nutrient transformation, productivity and habitat maintenance and development.

The ecosystem itself possesses ***attributes***, such as biodiversity that derive from the composition, structure and functioning of the ecosystem.

However, neither of these concepts adequately addresses the differences in ecosystem processes between different types of river systems (*e.g.* small *vs* large; incised channel *vs* delta; pristine *vs* developed; located in temperate *vs* arid conditions) Johnson *et al.* (1995). Moreover, river ecosystems are generally organised in a nested hierarchical structure, both spatially and temporally, with the physical and biological response (including the habitat type that can evolve) of successively lower levels, being determined by the level above it (Frissell *et al.* 1986; Urban *et al.*, 1987; Noss, 1990). Ecosystems are not closed systems: their components, functions and attributes (ecosystem goods and services, Box 2) are directly and indirectly beneficial to society as indicated in the comprehensive classification described by Mander and Quinn (1999) and shown in Table 2.1.

Box 2 Ecosystem goods and services

Daily (1997, page 3) defines ecosystem services as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life", and which "maintain biodiversity and the production of ecosystem goods".

Table 2.1 Examples of aquatic ecosystem goods and services, functions and ecological benefits (modified from Mander and Quinn, (1999), after Costanza *et al.*, 1997)

ECOSYSTEM GOODS AND SERVICES	*ECOSYSTEM FUNCTIONS*	*ECOLOGICAL BENEFITS*
Gas regulation	Regulation of chemical composition of the atmosphere	Carbon sequestration, oxygen and ozone production
Climate regulation	Regulation of local temperatures and precipitation	Urban heat amelioration, wind generation
Disturbance regulation	Regulation of episodic and large environmental fluctuations on ecosystem functioning	Flood control, drought recovery, dilution of pollution events
Water supply and regulation	Supply and regulation of water flow	Provision of water for agricultural, industrial and household use (spatially and temporally)
Sediment supply and regulation	Regulation of sediment supply to estuary and marine environment	Maintenance of beaches, sand bars, sand banks
Erosion control	Retention of soil within an ecosystem	Prevention of soil loss by vegetation cover, and by capturing soil in wetlands
Soil formation	Soil formation processes	Weathering of rocks by water and accumulation of organic material in wetlands
Nutrient cycle	Storage, recycling, capture and processing of nutrients	Nitrogen fixation, nitrogen cycling through food chains
Waste treatment	Recovery of nutrients, removal and breakdown of excess nutrients	Breaking down of waste, detoxifying pollution
Biological control	Regulation of animal and plant populations	Predator control of prey species, maintain population balance
Refugia	Habitat for resident and migratory populations	Nurseries, habitat for migratory fish and birds, regional habitats for species
Food production	Primary production for food	Production of fish and plants
Raw materials	Primary production for raw materials	Production of craftwork materials, house-building materials and fodder
Genetic resources	Unique biological materials and products	Genes for food and ornamental fish species, plant fibres
Nature appreciation and recreation	Provision of opportunities for appreciation of natural features and wildlife, sports, fishing	Providing access to nature and wildlife for viewing, walking, fishing, swimming, sailing, canoeing
Transport	Provision of opportunities for water based transport	Harbours, ferries
Cultural	Providing opportunities for non-commercial uses	Life enrichment values of ecosystems

Activities such as agricultural development of the riparian zone, damming of the river channel and groundwater mining interrupt the spatial and temporal dimensions of aquatic systems, thereby shifting the natural equilibrium between biological and physical features, and reducing the habitat diversity of river channels and floodplains (Johnson *et al.*, 1995; Poff *et al.*, 1997). Beyond the stream channel and floodplain, land based activities interrupt the hydrological cycle, altering the natural flow-paths of precipitation and streamflow generating mechanisms.

The interconnectedness of the Earth's processes, governed by oceanic, terrestrial and atmospheric circulation mechanisms and patterns, ensures that even those regions which have not been subjected to land use change are affected by human activity, the major consequences being pollution and climate change. Hence, human activities influence ecosystem processes through the utilisation of ecosystem goods and services, and humans should therefore be included in studies of the relationships of organisms with each other and with their environment. However, ecosystems do not behave in either a linear or continuous manner and, consequently, holistic approaches should be applied to the management of ecosystem processes for either ecological or social functioning. While many researchers consider that reductionist, numerical modelling is inappropriate to the synergistic attributes of ecosystems, an understanding of the dynamic roles of aquatic and riparian vegetation and biota, their life cycles, ecological relevance and intrinsic value to people can lead to quantitative predictive (*i.e.* cause and effect) modelling. Thus, the management of aquatic ecosystems requires a better understanding of the:

(a) mechanisms of interactions between ecological systems and hydrological alteration,
(b) interactions between surface and subsurface ecosystems,
(c) linkages between land and water interfaces and
(d) influence of human activities on aquatic systems.

Land based impacts within catchments are strongly linked to aquatic resources, and riparian ecosystems represent the interface between aquatic and terrestrial ecosystems. Toledo and Kauffman (2001) describe the linkages for riparian ecosystems as:

(a) vegetation influences channel morphology through erosion and accretion,
(b) channel morphology determines water availability to the riparian and floodplain zones and

(c) water availability to the riparian and floodplain zones delivers nutrients, enhances seed dispersal and influences plant species composition.

Since aquatic and terrestrial ecosystems are intricately interlinked, it follows that the ecosystems services they provide to people are interlinked (Box 3). Changes in terrestrial ecosystem functioning through human induced alteration of habitat or the unsustainable utilisation of ecosystems services can also impact on the ecological processes that regulate aquatic and riparian ecosystems (Johnson *et al.*, 1995). In addition, human induced alterations occur at varying spatial and temporal scales, ranging from catchment-wide land use change to point source pollution pulses (Townsend and Riley, 1999).

***Box 3 Interlinked ecosystem functions and services** (de Groot et al., 2002)*

Regulation functions:

Atmospheric composition, climate regulation; biological, flood and erosion control; soil formation; nutrient cycling; waste treatment

Production functions:

Food and raw material; fuels and energy; genetic material

Habitat functions:

Habitat organisation; refugia and reproduction habitat; biodiversity

Information functions:

Spiritual enrichment; cognitive development; recreation

2.3 The Value of Ecosystem Goods and Services

An ecosystem's value is measured (*i.e.* in terms of both quantity and quality) by the volume of ecosystem goods and services that it provides (Mander and Quinn, 1999). In addition to its value for basic human needs, water is intrinsic to the provision and quality of a diverse range of ecosystem goods and services used by society (Dugan *et al.*, 2002). Although, ecosystem goods and services are essential to sustain human life and well-being they are not traded in the conventional economy and attaching a price to this "natural capital" is difficult. Furthermore, ecosystem services operate on such vast and intricate scales that their value may be beyond pricing, since most could not be replaced by technology (Daily, 1999).

Mander and Quinn (1999) maintain that the relationship between ecosystem goods and services and the value of those services to people will not be linear since the value will depend on abundance, which ultimately depends on the state of the ecosystem. The primary benefit to society from aquatic ecosystems is usually considered to be the provision of water for agricultural, industrial and domestic use. Water abstraction for socio-economic activities will have greatest repercussions on ecosystems when society most needs functioning aquatic ecosystems, particularly during sequences of dry years and in those catchments that are water stressed. Water scarcity, as a result of abstraction, affects the value of ecosystem goods and services when ecosystems respond negatively to changes in water quality (Mander and Quinn, 1999). However, the protection of water resources comprises more than ensuring water quality and quantity (Karr, 1996). Water is also essential to sustain the ecosystem functions and processes that regulate the quantity and quality of water (Dugan *et al.*, 2002) and as emphasised by Jewitt (2001), focus on aquatic ecosystems as users of water diverts attention from their ecosystem functions. Therefore, the value of aquatic ecosystem goods and services to society is dependent on the state of the ecosystem, which is a function of its "natural state" and its utilisation by people.

Changes in water quality impact on the quality and quantity of "natural" goods and services that ecosystems can provide, resulting in changes in the "artificial" benefits associated with human activities or changes in the costs of those activities. McCartney *et al.* (1999) describe this relationship as the trade-off between the short-term socio-economic benefits of "artificial" services (*e.g.* reservoirs or irrigated fields) with the long-term environmental costs (*e.g.* pollution) to modified ecosystems (Figure 2.2). While many researchers advocate that sustaining modified ecosystems over the long term can be achieved through appropriate management, lasting benefits depend on maintaining essential ecosystem processes, functions and biodiversity (McCartney *et al.*, 1999).

2.4 Indicators of Aquatic Ecosystem Health

Ensuring that aquatic ecosystems continue to provide the goods and services desirable to society requires healthy ecosystems and that some degree of ecosystem integrity be maintained. Defining ecosystem health and ecosystem integrity, and the assessments and monitoring thereof, is debated by many researchers (*e.g.* Karr, 1996 and 1999; Rogers and

Biggs, 1999). However, the integrity of an ecosystem is generally understood to describe its wholeness and ability to continue to function in its natural way (McCartney *et al.*, 1999), implying correlation with its unimpaired or original state.

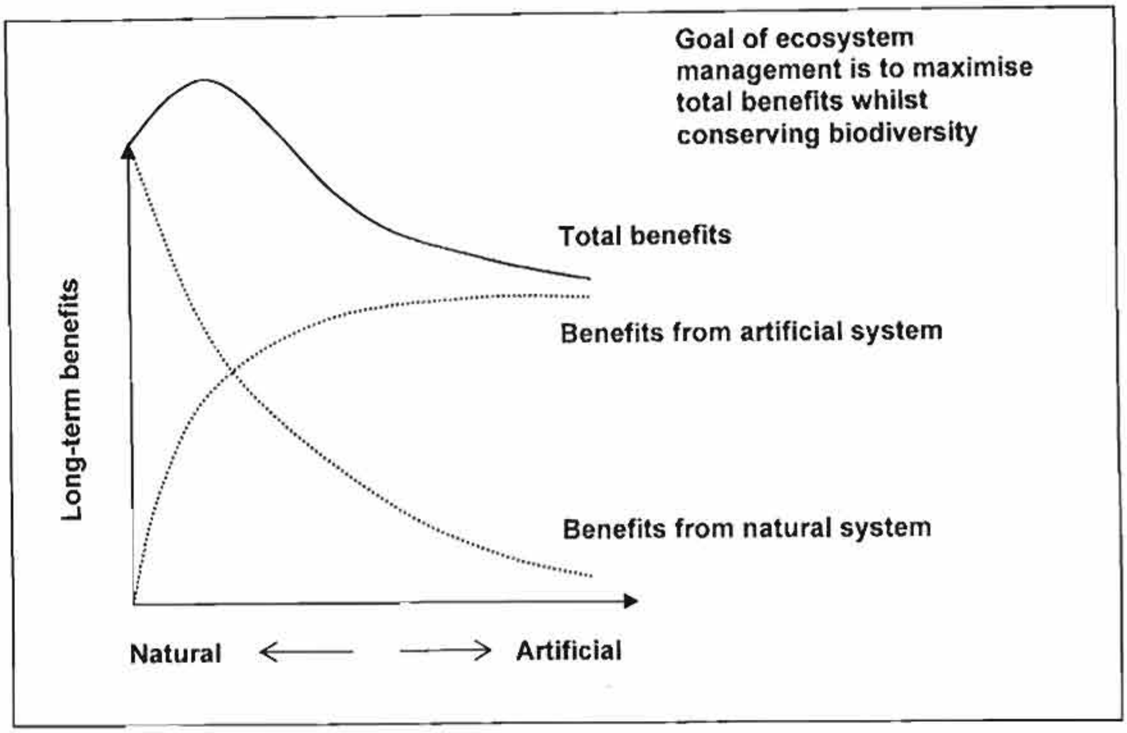

Figure 2.2 Maximising the benefits from freshwater ecosystems (after McCartney *et al.*, 1999)

Conversely, ecosystem health describes the goal for conditions of ecosystem services desired by humans, thereby incorporating an ecosystem's resilience to environmental and human pressures (Karr, 1996). The concept of aquatic ecosystem health holds different significance for different people (Karr, 1999). Bulk water suppliers may consider a river to be healthy if its impoundment provides sufficient water, at minimum treatment cost to support industrial and agricultural operation and domestic supply. Rural inhabitants would consider an aquifer to be healthy if it provided environmental security and a reliable water supply that did not impair their physical well-being. While qualitative expressions such as river health and aquatic integrity may be conceptually useful, aquatic ecosystems needs have to be quantifiable if management goals and targets to ensure their protection are to be achieved. Environmental indicators that quantify how far altered ecosystems depart from natural patterns are attractive management tools (Sparks, 1995; Richter *et al.*, 2003) and can be useful for assessing goals, such as biodiversity, that are valuable to society (Noss, 1990). Noss (1990) describes a nested hierarchy of four levels of ecological organisation, ranging from landscape to genes, for the interdependent compositional, structural and functional attributes of biodiversity in order to identify indicators for environmental monitoring and assessment (Figure 2.3). This hierarchical approach may be useful for the selection of indicators of ecosystem health, since indicators may be required to represent several spatial and temporal scales of ecological organisation. Nonetheless, Rogers and Biggs (1999) caution that there is a risk that assessing, monitoring or measuring ecosystem health can become surrogates for managing it.

Many environmental attributes are recognised as influencing the habitat templates that determine aquatic and riparian species richness and distribution, including flow velocity, depth and wetted perimeter, water temperature, oxygen content, turbidity, substrate size distribution, nutrients, and other physico-chemical and biological conditions (McBain and Trush, 1997; Richter *et al.*, 1997; Norris and Thoms, 1999; MacKay, 2001). However, water and sediment quality, together with the temporal patterns of water and sediment flows, are the major drivers of the biophysical composition, structure and function of aquatic ecosystems (Poff and Ward, 1990; Richter *et al.*, 1996; McBain and Trush, 1997) and strongly influence instream and riparian habitat structures, trophic base and biotic interactions (Sparks, 1995).

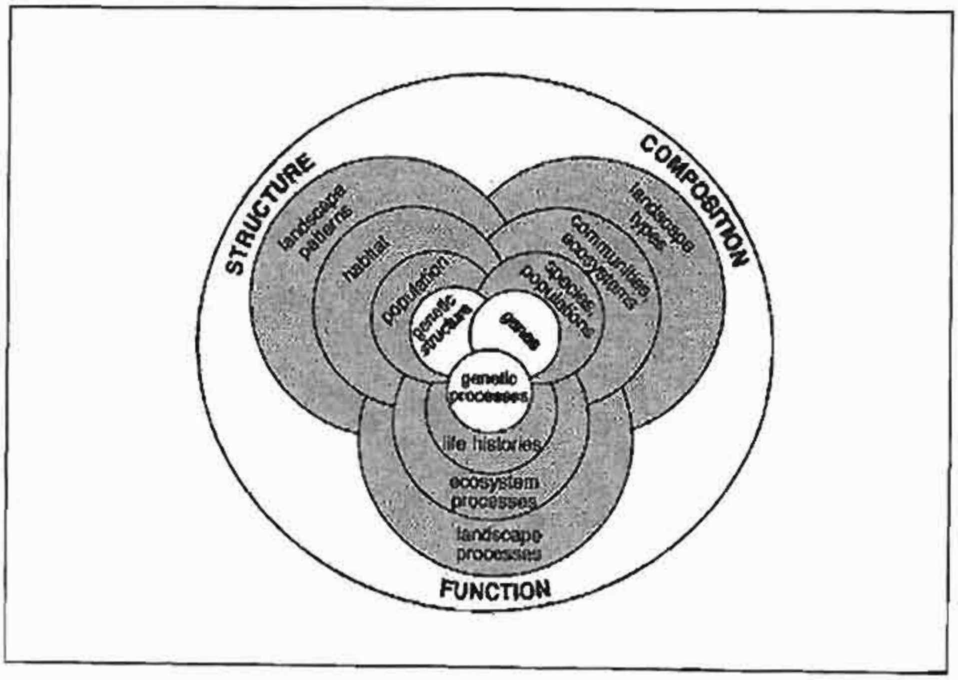

Figure 2.3 Compositional, structural and functional attributes of ecosystems with multiple levels of spatial and temporal organization (Noss, 1990)

There are strong links between hydrology, geomorphology and biota, with positive feedbacks (Figure 2.4) resulting from the change in one to the others (Heeg and Breen, 1994; Kauffman *et al.* 1997 cited in Toledo and Kauffman, 2002). Human activities in catchments add another dimension by influencing the physical (hydrologic and geomorphologic) and the biological (biotic) processes of the aquatic and riparian ecosystems and feedbacks through the utilisation of ecosystem goods and services. The abiotic-biotic linkages in Figure 2.4 suggest four potential categories of indicators to measure aquatic ecosystem health: anthropogenic, riverine biotic, geomorphologic and hydrologic.

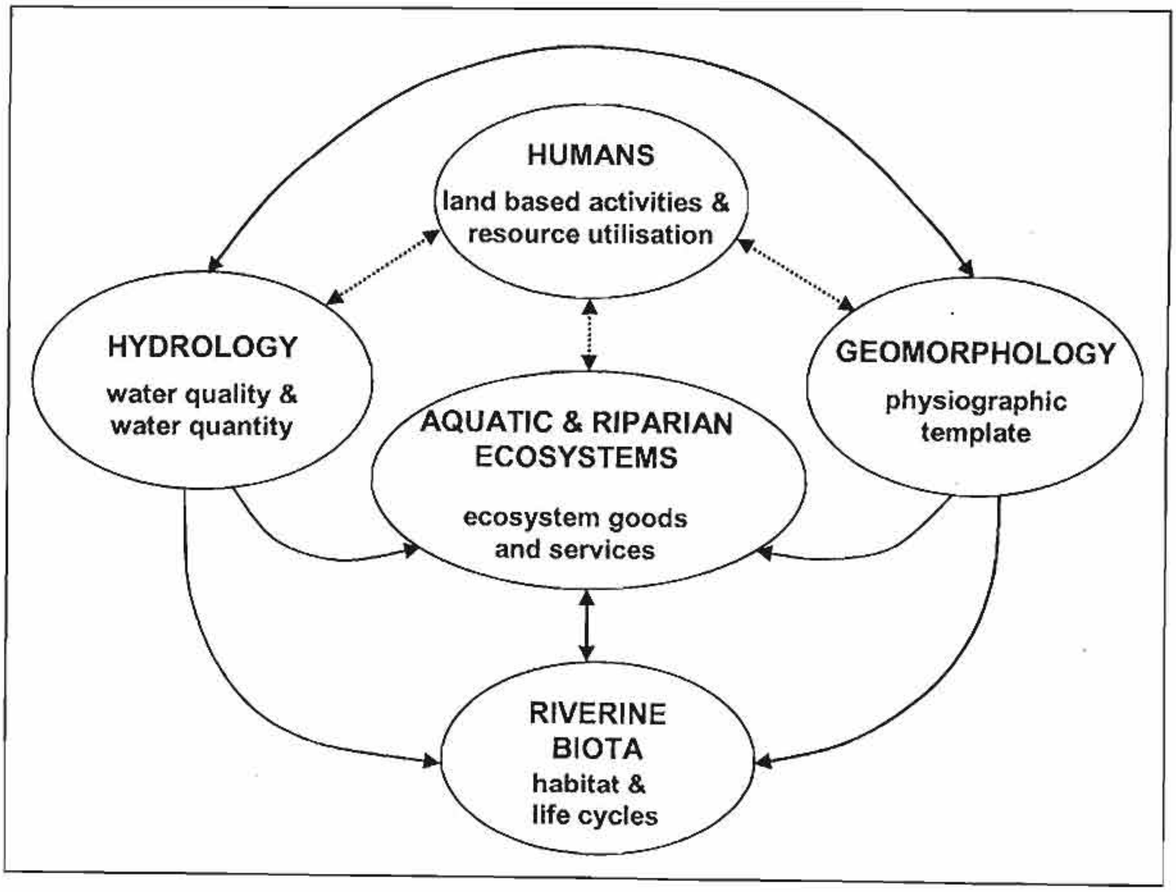

Figure 2.4 Linkages between hydrology, geomorphology and riverine biota and human impacts (modified from Toledo and Kauffman, 2002)

2.4.1 Anthropogenic indicators of aquatic ecosystem health

There are many human pressures exerted on aquatic ecosystem integrity, some of which are referred to later in this Chapter. The impacts of humans on aquatic ecosystem health are a consequence of both direct and indirect activities, occurring over the entire catchment. These impacts are frequently amplified by the vagaries of the climatic regime. McCartney *et al.* (1999) apply the Driving Forces-Pressure-State-Impact-Response (DPSIR) sustainability assessment framework, now adopted by the European Environment Agency (EEA, 2003), to describe a simple model of the way in which societal, economic and ecological systems interact and impact on water resources (Figure 2.5).

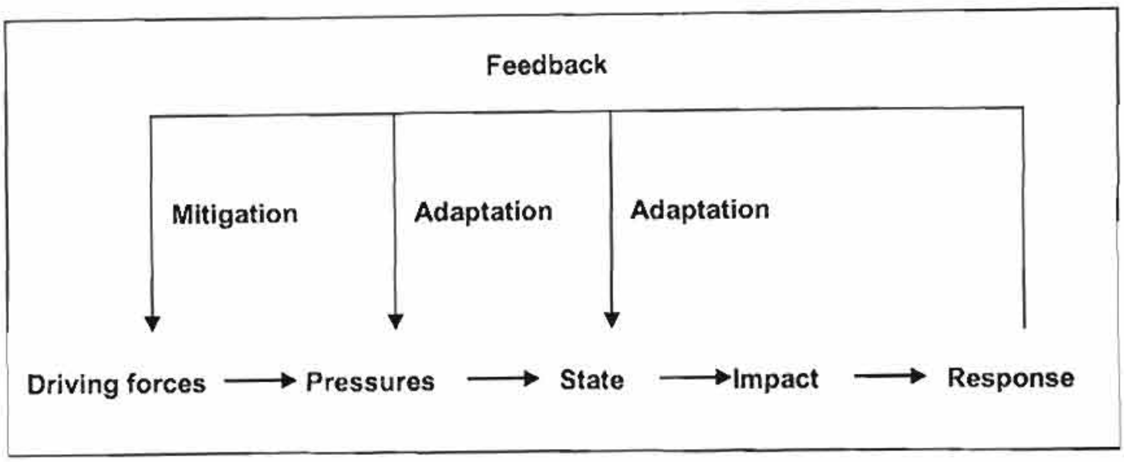

Figure 2.5 Simple model of the interactions within socio-economic-ecological systems (McCartney *et al.*, 1999)

Population growth is a major ***driver*** of human ***pressure*** on water resources. Environmental degradation has occurred in densely populated areas, and in some instances aquatic ecosystems have collapsed beyond their natural resilience. The proportion of people living in cities can be used as an aquatic ecosystem indicator, since densely populated areas cause ***pressure*** to ecosystems, consuming more "natural capital" than is locally available, which has to be supplemented by "artificial capital" before being disposed off in volumes that overload the capacity of natural systems (McCartney *et al*, 1999), thereby compromising the long term benefits from the ecosystem (see Figure 2.2). Nonetheless, the intricacies of socio-economic-environmental systems are such that even population growth, as an indicator of aquatic ecosystem health, can have different impacts in different regions. Changes in the volume of ecosystem goods and services provide an indicator of human pressure on aquatic ecosystem health, since in many regions the effects of over-utilisation of ecosystem goods, such as fish catch and size, are leading to unsustainable harvesting practices (Abramovitz, 1996).

More direct ***pressure*** indicators of aquatic ecosystem health include the number and size of dams together with the operating release and allocation rules (WCD, 2000). Dam construction is a ***response*** mechanism to the pressures imposed as a result of the global driving forces (Figure 2.5). However, dam construction is an example of both ***negative feedback*** (increased alteration of flow regime) and ***positive feedback*** (environmental security for, perhaps, the short term) within the socio-economic-environmental system. Water quality or pollution indicators (*e.g.* biological oxygen content, total oxygen content, nitrates, phosphates, suspended solids, *E.coli* count, heavy metals contamination, pH, temperature and total dissolved salts), which alter the quality of receiving waters as a result of human activity, can also be measured directly. However, for many regions in the developing world, monitoring water quality indicators is a low priority for a variety of socio-economic-politico and technological reasons. Furthermore, because of the complexities of the interactions between humans and the environment, it is difficult to ascertain quantitative links between water quality, pressure indicators and aquatic ecosystem health (McCartney *et al.*, 1999).

2.4.2 Riverine biotic indicators

In general, river health assessment and monitoring programmes tend to focus on the ***population-species-life histories*** level of the hierarchical organisation shown in Figure 2.3, and on invertebrates and fish rather than aquatic and riparian vegetation. Aquatic biota such as invertebrates and fish are relatively "fast" variables (*i.e.* they have short life stages, which occur over fast time scales and over narrow flow ranges) when compared to components of geomorphologic or hydrologic processes (Figure 2.4), and their abundance and distribution are strongly correlated to the quality and availability of habitat. Nonetheless, small invertebrate organisms generally have only a narrow distribution range during their aquatic life stages. Fish, which are more mobile and longer-lived, are better biotic indicators of general habitat conditions. However, most of the knowledge on the relationships between river systems and biota is from field studies conducted in small and accessible streams (Poff and Allan, 1995) and long-term databases for fish and other aquatic biota are rare (Poff and Ward, 1989; Richter *et al.*, 1996). Moreover, the availability of habitat requirements of keystone species or guilds in a localised section of a river is unlikely to meet the ecological requirements of entire river reaches or zones (Poff *et al.*, 1997) and has limitations for any causal links between habitat richness and the management of specific human needs. In addition, emphasis on biota to measure river health tends to overlook the non-biological processes (hydraulic, physico-chemical and geomorphic) that operate within ecosystems (Box 1 and Figure 2.4), which are important to ensure protection of aquatic ecosystem health (MacKay, 2001).

2.4.3 Geomorphic indicators

The geomorphological structure of a river provides the template upon which its ecosystems are structured (Heritage *et al.*, 2000). The fluvial geomorphic processes of erosion and sedimentation that maintain river channels, floodplains and deltas (the ***landscape*** and habitat levels in Figure 2.3) are, generally, "slow" variables (*i.e.* they have accumulative formation processes, which occur slowly over long time scales and over wide flow ranges. However, processes such as the initiation of channel bed mobility, scour deeper than the channel bed surface, floodplain inundation as well as deposition and riparian seedling scour can happen relatively quickly and over narrow flow ranges, (McBain and Trush, 1997). Generally, the temporal and spatial scales at which geomorphologic processes function are many magnitudes greater than those at which ecologists work (Norris and Thoms, 1999), although analysis of sediment deposits which reveal the palaeo-ecological record may address some of the mismatch between the different record lengths and dimensions of ecosystem processing (Thoms *et al.*, 1999).

Attempts to develop a geomorphical classification of the longitudinal zonation of rivers include using channel gradient as a predictor (indicator) of the river reach types observed within prescribed zone classes (Rowntree *et al.*, 2000). However, the validity of geomorphic zones as indicators of habitat availability and biological diversity remains uncertain. Nonetheless, Heritage *et al.* (2000) propose that knowledge of the structure and functioning of representative river reaches or zones of repeatable geomorphological types, gained through an understanding and monitoring of the mechanisms controlling local hydraulic parameters, permits prediction of the effects of altering streamflow and sediment regimes in similar geomorphic zones.

2.4.4 Hydrological indicators

Hydrological regimes play an important role in determining channel morphology (***habitat level*** in Figure 2.3), occurrence and distribution of aquatic and riparian biota (***population-species level*** in Figure 2.3), water quality, water temperature, transportation of sediment and organic matter (***ecosystem processes level*** in Figure 2.3), estuarine inflow and other environmental conditions. Hydrological conditions vary spatially and temporally within ecosystems, thereby influencing habitat diversity. As described earlier, hydrological conditions fluctuate spatially in longitudinal (upstream-downstream), lateral (channel to riparian) and vertical (channel to groundwater) dimensions, whereas temporally, hydrological conditions vary inter and intra-annually (Ward, 1989).

Aquatic science literature consistently narrates five streamflow components as being influential in the spatio-temporal heterogeneity of aquatic habitats: the magnitude, timing, frequency, duration and rate of change of water conditions (*e.g.* Poff and Ward, 1989; Bayley, 1995; Richter *et al.*, 1996, McBain and Trush, 1997). These streamflow variables are ecologically relevant, yet sensitive to human influences such as land use change, reservoir operation, groundwater pumping and agricultural diversions (Richter *et al.*, 1996). Many hydrological statistics, including the medians of annual, monthly and daily flows, and the coefficient of variation of flows, have been proposed as indices to characterise the temporal variability associated with the environmental conditions of rivers.

Hydrological indicators of aquatic ecosystem health are particularly useful since long-term historical records, which inherently include the impacts of human induced change relative to natural conditions, are often available. Hence, it is suggested that hydrological indicators can be used to assess the integration of societal and ecological systems.

2.5 Summary

Fully functioning aquatic ecosystems are driven by the relationship between the streamflow regime and water quality. Alteration in the streamflow regime and water quality results in changes in the generation of ecosystem goods and services. Hydrological indicators are valuable and appropriate for measuring the extent, degree and location any such alterations.

3 ECOLOGICAL BENEFITS PROVIDED BY THE HYDROLOGICAL CYCLE

3.1 Introduction

Given both the differences and interactions between ecological and societal functioning, spatially, temporally and organisationally, it is important to identify ***how*** the hydrological cycle influences the ecosystem functions that deliver the ecosystem services required by people. There is a need to identify those characteristics of the hydrological regime that are important for the provision of a variety of ecosystem goods and services (Strange *et al.*, 1999). It is becoming clear that this requires integration of the understanding of hydrological, geomorphological, and ecological processes, since the habitats used by aquatic biota, *e.g.* an alternate bar in a river, are not formed and maintained by a single flow event with a particular magnitude, timing, frequency and duration (Poff *et al.*, 1997; McBain and Trush, 1997).

3.2 The Variability of Flow Conditions

The intra- and inter-annual variability of flow is a key selective pressure on aquatic and riparian organisms and a primary control on channel form and process (Zeimer and Lisle, 1998). In many regions, the hydrological regime is naturally highly variable. Even with storage and flood control infrastructure, prevailing climatic and hydrological conditions constrain the extent to which floods can be avoided, or a reliable supply of water assured during periods of drought (IWMI, 2002). There is extensive literature relating the impacts of impoundment, diversion and withdrawal of water on the spatial and temporal distribution of water and sediment flows. Strange *et al*, (1999) describe some of the important changes in hydrological regimes as:

(a) dewatering of downstream streams (*e.g.* as a result of inter-basin transfer)

(b) higher baseflow contributions as a result of supplementary irrigation in the dry season

(c) increased daily flow fluctuations

(d) increased salinity as a result of irrigation return flows and

(e) excess nutrients as a result of increased point source pollution from waste treatment works

Other important changes include increased temperature as a result of industrial return flows and reversal of seasonal flows as a result of irrigation abstractions.

Nonetheless, changes from natural to agricultural land cover represent substantial and direct threats to freshwater ecosystems since they cause changes in the natural energy flow paths and mineral cycling, weakening the linkages within the hydrological cycle (McCartney *et al.*, 1999).

While the impacts of altered hydrological regimes on ecosystem processes are not fully understood, there is growing recognition in water resources management of the role of the natural flow regime in generating and maintaining the ecological integrity of the aquatic and riparian ecosystems (Sparks, 1995; Richter *et al.*, 1997; Poff *et al.*, 1997). There are, however, limits to defining the natural flow regime of rivers in catchments that have been modified by human activity (Poff *et al.*, 1997). While it is not practical to restore natural flow regimes in human dominated catchments, some restoration of the natural flow variability is usually an option for water resource management (Richter *et al.*, 1997; Strange *et al.*, 1999). However, "restoring a flow pattern that is simply proportional to the natural hydrograph in years with little runoff may provide few, if any, ecological benefits, because many geomorphic and ecological processes show non-linear responses to flow" (Poff *et al.*, 1997). There is also evidence of the importance of the sequence of flows to both ecological and geomorphic processes (Shafroth *et al*, 1995; McBain and Trush, 1997). In many catchments post-dam channel morphology and habitat can be managed by allowing the natural fluvial processes to function in some years. Ecosystem goods and services can be gained from utilising the natural inter-annual variability of flow regimes, whereby surplus water in years with above average flow can be released to exceed the flows required to initiate essential geomorphic and ecological processes (Poff *et al.*, 1997; Richter *et al.*, 1997). In water stressed catchments, the releases need only just exceed the threshold of flows required to initiate or drive key fluvial processes (Stewardson and Gippel, 2002).

3.2.1 The seasonality of flow regimes

Seasonality (including magnitude, timing and duration of flows) can be an indicator of both water quality and of water quantity (Hughes, 2000). Seasonal fluctuation in flow is a major determinant of river channel structures, providing the physical template of habitat diversity: it is the driving influence that "resets" biological communities throughout river systems (Power, *et al.*, 1995). Seasonality provides appropriate environmental conditions and life cycle cues of still water, freshes or floods for biological communities for different seasons (Tharme, 1998). In perennial rivers, seasonality distinguishes dry from wet season baseflow. Natural seasonal variability in flows is associated with the flow dependent life history requirements (*e.g.* freshes for fish spawning or floods for vegetation seed dispersal) of naturally occurring biota and it can prevent the establishment of alien species (Poff *et al.*, 1997). Seasonal patterns of flow determine the inundation and agricultural potential of floodplains and backwaters, which are the most biologically productive areas of river systems for both ecological and societal systems.

3.2.2 The role of disturbance

Whilst seasonal and annual flow regimes follow broadly predictable patterns over time, individual storms and occasional droughts can produce unpredictable magnitudes, timing, durations, frequencies and changes in rates of rise and fall in the hydrograph (McBain and Trush, 1997). McBain and Trush, (1997) describe this temporal unpredictability, or disturbance, as the "foundation for dynamic river ecosystems". The occurrence of disturbance (and its absence) is critical to ecosystem functioning. For example, the riparian zone requires some degree of disturbance to bring in nutrients or to initiate seed dispersal. Vegetation succession may require major floods to expose moist substrate, followed by a period free from disturbance for seedling germination (Shafroth *et al.*, 1995). Disturbance in flow regimes produces a complex mosaic of habitat and biotic diversity along stream and riparian systems (Michener and Haueuber, 1998). Nonetheless, extreme events have different relevance at different spatial scales. In small streams and headwaters, floods are generally considered to be disturbances that reset ecosystem processes and conditions, whereas the absence of a flood (*e.g.* during periods of drought) is considered a disturbance in river floodplain ecosystems (Sparks, 1995).

The remainder of this section focuses on some of the key components of the hydrological regime. The ecological relevance associated with both low flow and high flow events is considered since they provide the variability of streamflow conditions which influences the provision of the ecosystem goods and services required by society. In particular, the following Sections 3.3 and 3.4 consider the "extreme" high and low flow events that present critical disturbances to river ecosystems and act as ecological constraints and "bottlenecks" providing stresses as well as opportunities for a wide variety of indigenous and invasive aquatic and riparian species (Poff *et al.*, 1997).

3.3 Low Flow Events

Defining low flows is hindered by ambiguous terminology. In a review of the status of low flow hydrology, Smakhtin (2001) describes the discipline of low flow hydrology as one that "deals with minimum flow in a river during dry periods of the year". In addition, Smakhtin (2001) expanded upon the International Glossary of Hydrology (WMO, 1974) definition of low flow, *viz.* "the flow of water in a stream during prolonged dry weather", to distinguish the lowest annual flow which occurs in the same season each year, from drought resulting from "less than normal precipitation for an extended period of time". Both of these flow conditions occur naturally and both can be intensified by human activity. Low flows are usually derived from groundwater discharge, unsaturated lateral flow from the soil profile, *i.e.* delayed throughflow (Hewlett and Hibbert, 1963; Lorentz, 2001) or surface water discharge from wetlands or glacier melts, yet for most of the annual dry season, streamflow discharge consists entirely of baseflow (Box 4). Anthropogenically induced low flows are attributable mainly to water abstractions from both ground and surface water, ineffective dam operation and changes in land use which result in reduced groundwater recharge and delayed throughflow. Ecologists often define low flows as the flows in a river that occur outside of floods, thereby including wet season baseflows as low flows. However, for the purposes of this chapter the definition by Tharme and King (1998), "the range of flows within the dry season baseflow that may be naturally or unnaturally low in magnitude" is used to describe low flow.

Box 4 Baseflow

Chow (1964) describes baseflow (or base runoff) as the sustained or fair-weather runoff occurring at all times of the year. Baseflow is the sum of groundwater runoff and delayed throughflow, although some hydrologists include the total throughflow (Ward and Robinson (1999).

There is limited documented information relating the ecological benefits of naturally occurring low flow events to society. Low flows reduce water availability not only for supply, but also for dilution and aeration of streamflow, degrading water quality for abstractors as well as aquatic ecosystems (Ward and Robinson, 1999). Nonetheless, low flows have important roles to play in aquatic ecosystem functioning. Low flows determine the ephemeral or perennial character of a river, and consequently its biotic composition and conditions for survival (Brown and King, 2002), as well as influencing opportunities for human activity. Low flows provide low velocity water conditions and shallow pools for juvenile fish and aquatic insects. Dry-season baseflow has critical roles in the functioning of river ecosystems and provides wetted habitat, appropriate temperatures and chemical composition for the survival of aquatic and riparian vegetation throughout the months when ecosystems and people most need water (Tharme, 1998).

The detrimental impacts of streamflow reductions to river ecosystems in dry season months as a result of human activity are well known and documented.

3.3.1 Characterising low flows

While several flow conditions can be described as "low", the magnitude and timing of annual low flows mark the seasonality of the flow regime of a river. In perennial rivers, baseflow continues through even prolonged dry periods. However, in intermittent and ephemeral rivers zero flows are common and where this, or any other selected critical low flow discharge occurs, the frequency and the duration of continuous low flow events may be useful indicators of low flow conditions (Ward and Robinson, 1999). Other distinguishing temporal characteristics of low flows include the variability of the streamflow regime, the regionally different rates of streamflow depletion in the absence of rainfall and the relative contribution of low flows to the total hydrograph (Smakhtin, 2001). "Extreme" low flows, occurring over narrow flow ranges of unpredictable magnitude, duration and frequency, are required to occasionally stress aquatic ecosystems. In the natural flow regime, extreme low flow events increase the genetic diversity of biological populations and communities and enhance the resilience of ecosystems. Thus, the benefit of naturally occurring extreme low flow events to society is the maintenance of diversity of ecosystem services.

3.3.2 Processes influencing low flows

Low flow characteristics vary spatially and temporally with different catchment physiographic and climatic features and as a result of human activity. Smakhtin (2001) describes several catchment characteristics and processes, including the distribution and infiltration characteristics of the soils, the hydraulic characteristics and the extent of aquifers as well as the rate, frequency and amount of recharge, the evaporation rates from the catchment, distribution of vegetation types, topography and climate as influencing various aspects of low flow regimes. These characteristics drive low flow generating mechanisms which Smakhtin (2001) groups into natural "gains and losses to streamflow during the dry season of the year". In short, gains are generally releases from groundwater storage which sustain low flows in prolonged dry periods and occur where the streambed intersects a perennially recharged aquifer, or is connected to a perched water table, or alluvial or channel bank storage or wetland area. However, in arid zones, gains to dry season baseflow may result from unsaturated subsurface water storage on hillslopes (Lorentz, 2001). Transmission losses to streamflow in dry periods are influenced by evaporation and transpiration rates, groundwater recharge from streamflow and losses to the streambed, channel banks and riparian vegetation (Smakhtin, 2001). Where bank storage is the dominant streamflow generating mechanism, channel shape, depth and composition influence whether baseflows sustain instream flows (Whiting and Pomeranets, 1997).

Humans impact on low flow generating mechanisms directly through river abstraction or diversion and indirectly though a variety of land based activities, all of which influence the natural gains and losses described by Smakhtin (2001). River diversions and irrigation practices alter the baseflow regime (both wet and dry season) and often baseflows are increased in magnitude and exhibit altered salinity concentration following groundwater recharge in downstream reaches, particularly during dry season months. Groundwater abstraction affects the water table level and unsustainable utilisation of this resource can result in environmental degradation, particularly in the dry season, with loss of habitat for the biological functions on which riparian people depend. Activities that alter vegetative and soil characteristics of the near stream habitat, such as the drainage of wetland areas as well as the afforestation of the catchment, change natural evapotranspiration rates, soil infiltrability and porosity, often with detrimental impacts to the quality as well as the supply of the low flows. On the other hand, natural forests improve soil infiltrability and reduce erosion of the land surface and sedimentation of rivers (Bos and Bergkamp, 2001).

Management of low flows is required to prevent irreversible damage to ecosystems during prolonged or extreme droughts. With increasing pressure to develop catchment land and water resources for economic growth and social equity, there is increasing need to understand the processes influencing low flow generating mechanisms under different catchment and climatic conditions and the impacts of human activities on those processes.

3.4 High Flow Events

The terminology around flow events that contribute to periods of increased discharge in the stream hydrograph is equally ambiguous. However, in contrast to the relatively slow hydrological processes that characterise low flows, the faster and more direct runoff processes contributing to the total streamflow reaching the drainage basin outlet in response to precipitation can be described as high flows (Box 5). High flow is derived mostly from rapid throughflow (or relatively quick, shallow sub-surface flow), surface runoff (or overland flow) and channel precipitation, and represents the major runoff contribution to streamflow during storm periods and most floods (Ward and Robinson, 1999).

> ***Box 5 High flows***
>
> Tharme and King (1998) describe high flows as "the periods of elevated flow that are variable in magnitude, duration, timing and return period (recurrence interval). High flows are typically divided into freshes and floods for ecological purposes".

In contrast to low flows, the societal benefits of high flow events are well documented. High flows regulate a number of riparian ecological processes. Moderately high flows (medium or maintenance flows) sustain sediment flows through the channel, thereby exporting essential nutrients for aquatic and riparian communities (Poff *et al.*, 1997). Moderately high flows also maintain ecosystem productivity and diversity in the channel and at the water and land interface. Higher flows such as the flood pulse, *i.e.* the regular inundation of the floodplain that maintains the biodiversity and biological productivity highly valued by society (Bayley, 1995), is the "key driving force in river-floodplain [including wetlands] ecosystems" (Sparks, 1995). The flood pulse dissolves and flushes out accumulated nutrient deposits and detritus and carries in fresh nutrients and sediment, which are deposited on the riparian terrace and floodplain, thereby integrating terrestrial vegetation with the river system (Heeg *et al.*, 1980). In this way, rivers provide organic material and sediments to floodplains in "exchange" for seasonal breeding grounds for aquatic biota and water purification through absorption of nutrients and pollutants on the floodplain (Acreman and McCartney, 2000).

Small floods (freshes) act as triggers to ecosystem functioning, such as stimulating appropriate conditions for fish spawning or migration, and contribute to the variability of flow conditions (Brown and King, 2002). Larger floods play a major role in determining the geomorphic character, shape and size of the river channel and are necessary for channel and estuarine scouring and maintenance. However, Sparks *et al.* (1990) developed a definition for "disturbance in a river floodplain" which can be applied to describe large infrequent floods, *viz.*, "an unpredictable, discrete or gradual, event that disrupts the structure or function at the ecosystem, community or population level" (*c.f.* Section 3.2.2). It is pertinent to note that their definition is also intended to apply to extreme low flows.

High flow events provide water for storage and abstraction for a variety of societal functions (irrigation, industrial and domestic use, Table 2.1), yet the infrastructure required to utilise the supply of water as well as the encroachment by people, buildings and agriculture proximate to the floodplain compromises the integrity of ecosystems to regulate episodic and large floods. Inundation of the floodplain is a river's natural defense system to high flows, yet human-built flood defense systems that restrict the natural river path can result in greater damage downstream. However, any direct relationship between human modifications of catchment and freshwater ecosystems and the likely impacts of floods is obscured by the spatial and temporal change in catchment physiographic conditions and the interactions of social systems, as indicated in Figure 2.5 (McCartney *et al*, 1999).

3.4.1 Characterising high flows

Similarly to low flows, the frequency and magnitude of high flows often determines the composition and abundance of species that are present in a river (Poff, *et al.*, 1997). The flood pulse (flow of predictable magnitude, timing, duration, frequency and rate of rise and fall) is in most instances an annual event controlling the predictable advance and retraction of water to the floodplain (Bayley, 1995). The rate of rise and fall of high flow events is ecologically relevant, since flood pulses that are too long may delay the recovery of riparian vegetation, whereas flood pulses that are too short, or at unusual times, may be insufficient or desynchronised with the reproduction cycles of flood-dependent organisms (Michener and Haueuber, 1995). Flood events outside the predicable range of magnitude, frequency, duration or timing of flows are disturbances to the river system (Sparks *et al.*, 1995). Whilst these events can be man-induced, such disturbances are important in the natural flow variability of river systems. Reice (1994) postulated "the normal state of communities and ecosystems is to be recovering from the last disturbance".

3.4.2 Processes influencing high flows

The flow paths of precipitation that generate high flows are governed mainly by the distribution and infiltration characteristics of catchment soils, hydraulic conductivity of the soil profile, "piston displacement" of moisture and water storage in the soil and rock layers, presence of soil biological activity, distribution of vegetation types and topography (Ward and Robinson, 1999). In addition, various catchment and channel characteristics intensify high flow or flood conditions. Ward and Robinson (1999) group the mechanisms driving high flow events into stable and variable conditions. Accordingly, stable catchment characteristics include area, shape, slope, aspect and altitude, which together define the general geomorphology of the drainage basin and river channel drainage pattern. Variable catchment characteristics include interactions between climate, geology, soil type, vegetation cover, wildfire and human influence. These interactions influence the capacity of water storage, infiltration rates and transmissibility associated with the timing and magnitude of the streamflow response to precipitation (Ward and Robinson, 1999).

3.5 The Resilience of Ecosystems

As discussed in Section 2 of this Chapter, ecosystems are naturally dynamic: the flow paths of energy and nutrients vary constantly over time and space. The natural ability of ecosystems to absorb stress or disturbance, as a result of change in the components and processes that characterise the system behaviour (and discussed in this Section) is a measure of system resilience (McCartney *et al.*, 1999). Ecosystems with low natural resilience are more susceptible to changes, with relatively small changes resulting in impacts with greater magnitude or duration than in more resilient ecosystems. Generally, streams that are physically dynamic, or ecologically diverse, have greater buffering capacity and greater potential for natural recovery. It is this capacity (or resilience) of ecosystems to absorb disturbances which links the dynamic attributes of ecological systems with the institutional attributes of societal systems (Berkes and Folke, 1998).

3.6 Summary

Different characteristics of hydrological variability influence aquatic ecosystem composition, structure and function. Aquatic ecosystems require both seasonal and inter-annual variability in streamflows to maintain the natural dynamics that provide a diversity of ecosystem goods and services. Maintaining natural hydrological variability, including disturbances, can be expected to provide the greatest ecological and societal benefits.

4 LINKING AQUATIC ECOSYSTEMS WITH HYDROLOGICAL FUNCTIONING AND SOCIETAL SYSTEMS

4.1 Introduction

Current perspectives on environmental security promote the philosophy that water resources should be managed as renewable resources to sustain the ecosystem goods and services that are beneficial to people (MacKay, 2001; Richter *et al.*, 2003). Given that very few water resources exist in a pristine state and that any anthropogenic catchment development has the potential to impact on the hydrological cycle, water managers require guidance on the likely response of ecosystems to altered flows (Richter *et al.*, 1996; 1997; 2003). However, Water Resource Management (WRM) practices that single out specific ecosystem components, or address the needs of single water uses, have "hastened the decline in river ecosystem health" (Woo, 1999). Thus, the development of robust, holistic, water resources management frameworks is required to match the perspectives of different water users, thereby ensuring that desired ecosystem functions are maintained and that human and societal needs are met in an optimal and sustainable way (McCartney *et al.*, 1999).

4.2 Shifts in Water Resources Management

There has been a move to embrace the linkages between land use patterns, the hydrological cycle and biotic response to catchment issues through Integrated Catchment Management (ICM). ICM can be conceptualised as an umbrella strategy, within which WRM operates in conjunction with land care management: it presents a physical context for the management of inter-linked ecosystems (McCartney *et al.*, 1999). Over the past decade there has been a substantial shift in WRM towards the integration of the interests of water users and the environment to achieve sustainable utilisation of water resources. The Dublin and Rio statements of the early 1990s on holistic management and safeguarding basic needs and ecosystems have been followed by numerous initiatives to address the sustainable use of water and a vision for a "water secure world" (Kabat *et al.*, 2002). Increasingly, Environmental Flow Assessments (EFAs) have adopted an integrating role in WRM, with greater recognition of the linkages between ecological and societal systems. However, while public perception of a healthy aquatic ecosystem may be related to how the resource looks, for example in regard to river level or water clarity (King *et al.*, 2000; Calder, 2002), scientists and water managers may focus more on the value of ecological functioning, for example the socio-economic benefits of restoration of natural river flows (Naussauer, 2001). This difference in perceptions needs to be addressed with greater stakeholder involvement and greater awareness of the value of ecosystem services.

4.3 Information Requirements of Stakeholders

Integrating a societal dimension into a traditionally biophysical evaluation of the environmental flow requirements of aquatic systems emphasises the need for comprehensive and reliable data and information. Water users require information that correlates scenarios of modifications of a flow regime with the ecosystems services (both "natural" and "artificial") that would be provided. Mander and Quinn (1999) express this as a need to ascertain "how varying quantities and qualities of water within the river influence the volume and quality of ecosystem goods and services". Water resource managers require information on ecosystem integrity and resilience that translates EFAs into sustainable flow regimes to be implemented at the appropriate scale (MacKay, 2001). In addition, there is a need for WRM to incorporate an economic consideration so that stakeholders have an understanding of the value of their use of a water resource and the cost of such use to other water users, now and in the future (Mander and Quinn, 1999).

4.4 Managing Ecosystems to meet Environmental and Societal Needs

Matching the complexities of ecological scale to societal needs and the complexities of societal dimensions to ecological needs, has given rise to an "ecosystem approach" to the management of natural resources for the benefit of society and the environment. The concepts, merits and shortcomings of "ecosystem management" have received wide coverage in many ecological research papers (*e.g.* Fitzsimmons, 1996; Frissell and Bayles, 1996 and McCartney *et al.*, 1999) and will not be deliberated here (but see Chapter 3, Section 4). It is sufficient to note here that "taking an ecosystem approach to freshwater management means assessing water availability (quantity and quality), identifying inter-relationships at the ecosystem level, predicting the environmental and societal impact of any proposed action and evaluating the consequences before any decision is made on use" (IUCN and WWF, 1998). Consequently, the goal of ecosystem management is to identify mechanisms by which human interaction with the environment can be changed in order to enhance the long-term benefits to people, as well as the integrity of the ecosystem (McCartney *et al.*, 1999). Ecosystem management is a proactive approach to environmental issues and the effective management of freshwater ecosystems necessitates:

(a) evaluation of proposed development projects, through some form of Environmental Impact Assessment (EIA),

(b) identification of opportunities to integrate the environment in socio-economic development using approaches such as Strategic Environmental Assessment (SEA) and

(c) ascertaining baseline or threshold conditions, recognising the resilience of natural systems, to assure the sustainability of development proposals and opportunities through approaches such as Environmental Sustainability Assessment (ESA).

Holistic EFA methods, which integrate environmental, societal and economic aspects of aquatic ecosystems, have considerable potential to contribute to the processes associated with such assessments and to enhance environmental security. Contemporary reviews of approaches to EFA reveal that there is increasing recognition of the need to incorporate socio-economic components to the determination of environmental flow requirements (Marchand *et al.*, 2002; Tharme, 2003; Richter *et al.*, 2003). An example of the development of a holistic approach to EFA is the South African Downstream Response to Imposed Transformations, DRIFT (Brown and King, 2000; *c.f.* Chapter 3, Section 2.5), which arose as a scenario-based assessment of the socio-economic impacts of progressive reductions in river discharge from reference conditions on the biophysical functioning of the resource (Tharme, 2003).

4.4.1 A vision and goals for water resources

Many water managers, especially those in water scarce regions and where conflict over water issues is a reality, recognise that the sustainable and equitable distribution of water resources can be approached by setting aside water to provide for basic human needs as well as for ecosystem protection (*e.g.* the concept of the "Reserve" in the South African National Water Act of 1998, (NWA, 1998)). In order to achieve the goals of sustainability and equity in WRM, the participation of all stakeholders is paramount in the formulation of a "vision for the resource" with goal-oriented frameworks for the management of catchment water resources. Essentially, a vision for a water resource is based on the level of risk that stakeholders are prepared to accept for using the resource (MacKay, 2001). As such, a resource with high ecological significance should incur less risk than a resource which is either substantially modified or is more resilient to utilisation (MacKay, 2000).

Specific objectives to meet goals should be designed to protect water resource quality (including the quantity, pattern, timing, water level as well as assurance of flows, water quality and the integrity of aquatic and riparian biota and habitat) at a level of risk defined by the catchment stakeholders (MacKay, 2001). Specific objectives for a water resource are a statement of the vision for a water resource and, as such, should be measurable. This necessitates that "reference conditions" or "benchmarks" relating ecosystem "resilience", "sustainability" and "acceptability" be ascertained so that stakeholders can anticipate the desired future state of their water resource. Nonetheless, McCartney *et al.* (1999) maintain that setting specific water management goals should aim to increase environmental security by building resilience in ***both*** societal and ecological systems so that future or unintended changes for ecological integrity or societal well-being are minimised. This requires understanding of the interactions among societal, economic and environmental systems such as those shown in Figure 2.5.

4.4.2 Linking the environmental flow requirements of different water resources

Sustaining ecosystem goods and services relies on the maintenance of ecosystem health and function. Consequently, EFAs should focus on the identification and protection of ecosystem functioning and should address all linked components of water resources, including surface water (flowing or standing, in rivers, lakes, wetlands or impoundments), estuarine and groundwater as well as water quality attributes (MacKay, 2001). To date (2005), most efforts have focused on surface streamflows and little work has been done on the Environmental Flow Requirements (EFRs) for groundwater, wetlands or estuaries, despite the high ecological and societal significance of these resources. There is a need for greater understanding of the relationships between groundwater and other aquatic components, particularly the processes of groundwater recharge to surface streamflows, and the dependency of terrestrial ecosystems on groundwater. A valuable discussion of the implications of groundwater interaction with surface water bodies is provided by Xu *et al* (2002).

Groundwater resources recover slowly from over-utilisation, are susceptible to pollution from land-based activities and require a different system of evaluation to surface water resources (DWAF, 1999). However, it is essential that methodologies that address the linkages between groundwater and surface water, and the linkages between subsurface water and surface water be developed (Xu *et al.*, 2002), since groundwater flow and unsaturated lateral flow from the soil profile into streams as baseflow, represents the main long term components of total runoff in catchments (Ward and Robinson, 1999; Lorentz, 2001), thereby sustaining the surface water requirements during dry weather conditions. In catchments where groundwater is strongly linked to surface water, the impacts of supplementing basic human needs with groundwater resources could impact on the ecological functioning of surface water resources (Pollard *et al.*, 2002).

4.4.3 Determining flows for basic human needs

Notwithstanding the numerous economically important and societally beneficial goods and services provided by healthy ecosystems and referred to in Table 2.1, the priority water allocation is that which is essential for basic human needs. While there is controversy about just what constitutes "basic human water needs", South African water law (NWA, 1998) describes this ***human right*** as "the essential needs of individuals served by a water resource, which includes water for drinking, for food preparations and for personal hygiene". South African water law stops short of specifying a quantity or assurance level of delivery, however, in water resources assessments and Reserve determinations this right has been equated with the World Health Organisation guidelines of 25 litres per person per day as a "ballpark" quantity. However, and in common with other water-developing regions, the South African Department of Water Affairs and Forestry (DWAF) anticipates the priority of basic human water needs to intensify as the expectations for "essential needs" increase in the longer term and as the population being supplied with water increases (DWAF, 2001). Determining the domestic water for basic human needs, and the assurance of its supply, is anticipated to redress some of the issues of poverty, alienation from natural resources and environmental degradation in rural areas (Pollard *et al.*, 2002).

In water stressed catchments, or when there is drought, basic human water needs will most likely be sourced from a combination of surface water and groundwater, further complicating the assessment of ecological water requirements and linkages of these resources. Additional constraints to the determination of basic human water needs relate to differences among the spatial distribution of people, supply schemes and resource availability. Unlike an EFR, which is a non-consumptive instream requirement, basic human water needs is a consumptive off-stream water use: it is not sufficient to merely allocate the requirement at a point in a river, dam or aquifer within a catchment some distance from the intended population (Pollard *et al.*, 2002). As such, the determination of basic human water needs should also anticipate evaporation, transmission and transportation losses.

4.4.4 Linking environmental flows and societal requirements

Presently, the aquatic environment has a relatively strong "voice" in WRM as a result of global developments in water related legislation, which provided a springboard for the determinations of EFRs. Nonetheless, first priority should legally be given to basic human water needs. Therefore, this allocation should be clearly defined in terms of quantity, quality and assurance of supply. Any theories linking the ecological requirements and the societal water needs should be concisely articulated and rigorously tested since it can be contended that meeting societal water requirements does not rely on sustaining aquatic ecosystems. For example, in rural Andhra Pradesh, in India, aquatic ecosystems are severely compromised through over-utilisation of groundwater resources, yet water for basic human consumption and subsistence agriculture is met with transported deliveries of water to storage tanks (Batchelor, 2001, pers. comm.). However, for countries where large portions of the population lack the infrastructure to access even basic water supplies, this practice is inordinately more expensive than supplying rural areas with piped water and is unlikely to be sustainable (Stein, 2002). Nonetheless, what this does emphasise is the difference in spatial scale between the water needs of people beyond the channel and the instream needs of aquatic biotic communities.

Another major difficulty in establishing the relationship between the flows required for ecological functioning with those required for societal functioning is that there is no single scale for any of the components, nor of the issues at stake (Zeimer and Reid, 1997). Ecological systems operate at the life cycles of many different organisms as well as the temporal and spatial scales of ecological and geomorphic processes. Societal systems focus on the activities of one species, humans.

Increasingly, scenario-based assessments of socio-economic impacts of the flow responses to different water-management activities on the biophysical functioning of the resource are being developed. These assessment methods include the Downstream Response to Imposed Transformations, DRIFT, developed by Brown and King (2000) and the proposed interface between societal demand and aquatic ecosystem supply for the evaluation of downstream water functions and uses, designed by Marchand *et al.* (2002). Developments of this nature assess the relative importance of different ecosystem services, including seasonal use, to various sectors of riparian communities. This incorporates local knowledge on the relationships between flow events (typically, water levels of baseflows, high flows and disturbances) and the volume of ecosystem goods and services that can be expected. The abundance and distribution of valued ecosystem goods and services over time provides an ecosystem link to the societal requirements of river flows (King *et al.*, 2000).

Notwithstanding any assessment of the "natural" ecosystem goods and services required by riparian communities, there is a need to assess the "value-added", or "artificial", water related services (McCartney *et al.*, 1999), such as water for subsistence agriculture and food security as well as industry. The societal requirements of environmental flows are not restricted to in-channel use and increasingly the scope of assessment needs to be extended beyond the immediate riparian zone to include the impacts of terrestrial ecosystem utilisation by society on water resources. In South African water resources management, this need to link environmental flows with societal requirements for freshwater across the catchment has recently been approached by assessing various ecological flow scenarios against present day and planned water use. In the approach a Water Resource Yield Model (WRYM) is used to determine the water available to different socio-economic sectors across the catchment for each flow scenario investigated. Time series of monthly flows at various nodes in the WRYM are compared with equivalent time series of instream flow requirements set for different levels of ecological protection (Hughes, 2005). The water available for societal use is derived from the WRYM using a set of assurance and restriction rules to maximise use of the available water after meeting flows for different levels of ecological protection (Pienaar, 2005), thereby allowing stakeholders to make informed decisions.

4.5 Zoning of Water Resources to meet Environmental and Societal Needs

Typically, the delineation of water resource units in EFAs is strongly linked to "ecoregions", comprising geohydrological response units within a catchment, so that reference conditions can be ascertained. An "ecoregional type" can be determined by a combination of major physiographic factors, geological, geomorphological and geochemical attributes, regional natural hydrological characteristics, major natural vegetation types and biotic composition (MacKay, 2001). While these factors take cognisance of the ecosystems that might be expected under natural, or reference, conditions and are useful to determine the protection required for ecological requirements, the sensitivity of land uses and socio-economic conditions is omitted from this biophysical delineation. Furthermore, delineation based on biophysical systems bears little relation to the human population distribution and any linkages to the ecosystem goods and services required, including basic human water needs. In addition, there are difficulties associated with defining reference, or natural, conditions for biotic, hydrologic and geomorphic factors, not least of which is the lack of available recorded data. Even with rigorously tested models and computation techniques, verification of reference conditions is open to conjecture. While it is imperative that thresholds for ecosystem resilience, sustainability and acceptability have a reference benchmark to be meaningful, "the most natural thresholds are difficult to define until they are reached" (Granholm, 1987, cited in Godfrey and Todd, 2001).

In general, hydrological characteristics vary according to river reach as a result of differences in land and water use in different physiographic and climatic areas of catchments. An important basis of the so-called "receiving water quality objectives" in many water quality management approaches is that with increasing demands for supplies of fresh water to meet a variety of human activities there is increasing concern for the maintenance of healthy river flows for downstream use. The linkages of upstream to downstream water use, and the economic implications thereof, were highlighted by Savenije (2001) with his emphasis that the water system, at the catchment scale, should be regarded as a single system. Savenije (2001) maintains that water is a "special" economic good and that beyond the localised catchment scale, water has a "virtual" value, which is tradable as watershed goods and services (*e.g.* timber, food, drinking water and sanitation). Thus it is for the common good of society that water users adopt precautionary principles in their water use.

An alternative approach to the zoning of resource units based not only on hydrological, geological and topographical conditions, but also on *the fate of water outflow from the "unit"*, as suggested by Molden *et al.* (2001), could be a useful approach to managing ecosystems and maintaining the goods and services that societal and economic systems require for prosperity. Molden *et al.* (2001) developed a set of six hydronomic (*hydro* water and *nomus* management) zones to spatially represent water use within a catchment. These hydronomic zones (water source zone; natural recapture zone; regulated recapture zone; final use zone; stagnation zone and environmentally sensitive zone) are reaches or areas within catchments, defined on the basis of similarities of hydrology, geology, topography and whether or not the water outflow from a zone is recoverable for downstream use. The fundamental principle is that of the water available for use, part is converted to evapotranspiration and the remaining flows are either utilisable downstream, contribute to groundwater storage or become too polluted for re-use downstream. The zoning approach is used to develop sets of water management strategies that are suited to the different conditions that exist within catchments. The zones are described in detail in Molden *et al.* (2001) and need not be repeated here. It is, however, pertinent to note that the "water source zone" is the area where most of the runoff or groundwater recharge for downstream use is generated and is also where water management strategies, can have catchment-wide impacts (Molden *et al.*, 2001). Strategies could include tradeoffs with other water users in different zones. For example, upstream communities may be compensated for employing farming techniques that prevent erosion and degradation of downstream water.

The hydronomic zoning approach can identify catchment areas where water should be conserved, areas for which protection is required and those areas where genuine water use efficiency can be implemented to enhance water quality and quantity for downstream use (Molden *et al.*, 2001). Catchment areas or river reaches could be identified and managed as water conservation zones, "workhorse" zones, or those of intermediate water utilisation, depending on the societal value attributed to the ecological benefits of the resource. Consequently, hydronomic zoning can address the inter-linkages between humans and catchment land and water resources. Parallels can be drawn between the hydronomic zoning approach and the spatial and temporal energy and nutrient flows provided by the hydrological regime referred to in Section 2 of this Chapter. Generally, though not exclusively, the relationship is of an upstream-downstream nature and bears resemblance to the principles of Vannote's river continuum concept. The hydrological regime provides the downstream continuum of energy flows to the hydraulic habitat, which, in turn, supplies the aquatic and riparian ecosystem goods and services upon which people depend.

This upstream-downstream type of approach could be adapted when defining the temporal and spatial relationship between the requirements of ecological systems and societal systems beyond the river channel, especially where rural water supply is heavily dependent on groundwater supplies and there is a need to maintain the groundwater table for ecological functioning. Assuming a borehole as a water source zone, special zoning and management strategies for groundwater utilisation could be employed, based on groundwater levels or gradients and recharge rates to assure supply. A "circular protection" resource delineation envisaged as a "buffer zone" could be adopted, with constraints on abstraction or any polluting activity where the resource is ecologically or societally important or susceptible to over-utilisation (Batchelor, pers. comm., 2001).

Alternatively, the nested hierarchical structure of catchments and their river ecosystems, referred to in Section 2 of this Chapter, provides a useful analogy for linking land uses, socio-economic conditions and ecosystem goods and services with biophysical resource units. The generic framework for assessing EFRs, designed by Marchand *et al.* (2002) and referred to in Section 4.4.4 of this Chapter, is an example of a hierarchical approach, integrating ecological and societal systems in a range of spatial and temporal scales. The biophysical component is based on the combined ecosystem classification of Klijin (1997) and the river classification of Frissell *et al.* (1986), whereas the societal component is based on the value that stakeholders attribute to different ecosystem goods and services. The interface between the biophysical and societal components gives an indication of the appropriate spatial and temporal scales at which to assess different ecosystem functions and processes (Marchand *et al.*, 2002). In this way the relevant spatial and temporal management objectives can be identified for each ecosystem level. An example of matching the relevant scales (adapted from the said work of Marchand *et al.*, 2002 and research on ecosystem management by Rogers and Bestier, 1997) is provided in Table 2.2. However, the perceived value of ecosystem goods and services will differ for different regions and different socio-economic conditions.

4.6 Summary

Ecologically sustainable water management is required to meet both societal and ecological needs for water. This type of management strategy adopts an ecosystem based approach to address any incompatibilities between societal and ecological systems. Either hydronomic zones or a nested hierarchical structure of catchments and their river ecosystems, both based on the hydrological and physiographical environment as well as

Table 2.2 Relevant scales and management objectives for ecosystem functions determining ecosystem goods and services (modified from Marchand *et al.*, 2002 and from Rogers and Bestbier, 1997)

Ecosystem classification (Klijin, 1997)	**ECOZONE**	**ECOPROVINCE**	**ECOREGION**	**ECODISTRICT**	**ECOSECTION**	**ECOSERIES**	**ECOTOPE**	**ECO-ELEMENT**
River classification (after Frissell *et al.*, 1986 and modified by Marchand *et al.*, 2002)	Entire river basin	Entire river basin or part of it	River stream system	River segment	River reach	River reach	Pool / riffle system	Micro-habitat
Spatial Scale (km)	1000 – 10 000	100 - 1000	100 - 1000	100 - 1000	10 - 100	1 -10	0.01 - 1	0.001 – 0.01
Time span of processes (years)	> 1000	> 1000	> 1000	100- - 1000	100- - 1000	10 - 100	1 - 100	0.1 - 1
Management scale	Water Resources Planning	Water Resource Planning and Management	Conservation policy	Conservation managers make decisions	River conservation goals and monitoring		Species management scale	
Dominant processes and factors	Climate; parent material	Climate; parent material; geomorphology	Geomorphology; (drainage network development, floodplain and delta formation) altitude, groundwater flows	Geomorphology; (river meandering), ground and surface water; slope	Geomorphology, ground and surface water, slope, soils	Ground and surface water, slope, soils, bank erosion	Ground and surface water, changes in bed-form, soil and vegetation development	Seasonal depth, velocity changes, accumulation of fines, soil and vegetation development
Objectives	Maintain biodiversity to provide human benefits	Manage catchments to conserve diversity of structure	Maintain aquatic ecosystem biodiversity as part of landscape	Maintain natural ecosystem health and biodiversity	Determine whether biodiversity changes exceed Thresholds of Probable Concern	Catalogue riverine biodiversity	Minimise change in species population	
Drinking water								
Floodplain agriculture								
Fisheries								
Livestock								
Forestry								
Purification								
Flood mitigation								
Health								
Gene pool								
Recreation								

societal conditions could provide a useful approach to spatially represent water use and development within a catchment.

5 DISCUSSION AND CONCLUSIONS

Sustainable management of freshwater ecosystems is a key driver to environmental security and human health. However, management strategies need to match the spatial, temporal and organisational scales of biophysical systems with those of societal systems. Greater understanding of the linkages between hydrological, geomorphological and ecological processes and ecosystem response to changes would improve EFAs and support the effectiveness of resource quality objectives for the provision of the ecosystem goods and services on which people depend. This is especially so for slow variables since long-term changes associated with "resource collapse, surprises and new opportunities often derive from relatively slow variables" (Carpenter *et al.*, 1999).

Few studies have been conducted to ascertain the value of instream and riparian resources or of the socio-economic costs of their loss as a result of changes to flow regimes. However, the socio-economic value of aquatic systems should be incorporated in any EFA since flow related changes impact on the well-being and livelihoods of riparian and rural communities (Brown and King, 2000; King *et al.*, 2000; Marchand *et al.*, 2002). There is a need for the real costs of water resource utilisation to be ascertained so that stakeholders can make informed decisions about the "vision for the resource" or "desired future state". There are recent examples in South African water resources management of the application of ecological economics to the determination of environmental flow requirements (Pienaar, 2005). However, incorporating economic scenarios with flow-related scenarios is generally restricted to a limited number of ecosystem goods, and with the focus on water abstractions. The major shortcoming of such attempts is that ecosystem services are vastly undervalued or overlooked. In addition, water allocation to "artificial" ecosystem goods and services should include the costs of environmental degradation and stakeholders should be aware of the costs of restoring those "natural" ecosystem services that have largely been thought of as "free". For present activities to be considered sustainable to future water users, this should include the economic capital and discount rate of technical solutions. For the most part, technological fixes are expensive alternatives to natural ecosystem processes and, while "fixes" represent useful mitigation measures, they carry a risk that weakens the focus of protecting natural capital from degradation beyond the resilience of the ecosystem.

Increasingly, EFAs recognise the role of the natural flow regime in providing a variable and complex river ecosystem, creating a diversity of habitats. However, according to Robertson (1997) mimicking natural flows is not sufficient for the restoration of natural material cycles in regulated floodplain rivers. For example, vegetation clearance for agriculture and grazing on floodplains and wetlands contributes to changes in the linkages between river channels, riparian terrace and floodplains. The flood pulse determines the quantity and quality of nutrients and organic resources exchanged during high flows, yet the nature and degree of land-based activities on the floodplain impacts on these ecosystem processes (Robertson, 1997). This emphasises the necessity to incorporate land management practices with river management.

There is renewed interest in the role that the hydrological cycle provides in generating biodiversity and ecosystem services instream, at the riparian terrace, on the floodplain (including wetlands) and the greater catchment region. Jewitt (2001) identified that a greater understanding of the hydrological cycle and the impacts of land-based activities on the water cycle can assist in closing the knowledge gap between management practice and the theory and philosophy underlying Integrated Water Resources Management strategies. Studies to ascertain which hydrological parameters influence the ecosystem processes and functions that deliver valuable ecosystem services could offer water resource managers greater decision-making potential to achieve sustainable catchment management. However, quantitative databases of the flow requirements of different components of aquatic ecosystems are rare and more research is vital to make the links between ecosystem processes, hydraulic habitat and the socio-economic value of the diverse ecosystem goods and services.

Given the enormity and complexity of determining the flow requirements of different water resources, it is anticipated that developing the tools required to do so will take time. However, for many water-stressed regions, and despite far reaching water legislation, there is still a real threat that the protection of water resources may be implemented by restricting the allocation of licenses for socio-economic water use (Rogers *et al.*, 2000).

Ultimately, there is a risk that unintended impacts through the reallocation of water, to say additional water for irrigated agriculture, could result in unforeseen impacts on the ecological requirements. Meeting even basic societal demands could impact on the aquatic environment. For example, the emerging practice of rainwater harvesting for supplementary crop production and small-scale enterprise in rural areas could pose unforeseen impacts on the hydrological regime through the removal of precipitation from the hydrological cycle before it has the opportunity to perform key ecological processes. In addition, the site of any water harvesting could be relevant, since capture in upstream areas of catchment could lead to water shortages for downstream users.

It is important to have a clear understanding of the role of different components of the hydrological regime and their relation to "catchment zones". Perceptions relating to the functions of aquatic ecosystems and whether societal development is dependent on the maintenance of these systems need to be rigorously tested. For example, contrary to the assumption that headwater wetlands regulate streamflow by storing wet season flows for release in the dry season, a study by McCartney (McCartney, 2000; cited by Kiersch and Tognetti, 2002) in Zimbabwe showed that a large portion of dambos (wetlands) evaporates, with only a small portion contributing to dry season flows. Rather than reducing floods in the wet season, dambos were found to generate flood runoff once the soils are saturated. McCartney (2000) concluded that shallow rooting crops could be planted in dambos to utilise water in the wet season without impacting on dry season flows. In addition, there is growing awareness that many of the perceptions surrounding water use by trees and their interruption of the hydrological cycle, in particular by alien trees, needs to be re-examined. Water use by trees is complicated by species type and size as well as by site characteristics, climate and management practices. For example, in a study on the hydrological impacts of alien invasive vegetation in dry climates, Calder and Dye (2001) report study sites in India, where young alien eucalypts growing in soils of medium depth, and thus limited moisture availability, had the same water use as indigenous trees. At different sites, with deeper soils, the fast growing alien species had greater access to soil moisture as a result of deeper rooting, and consequently greater water use than indigenous species. However, Calder and Dye (2001) found no evidence of aliens extending roots to the water table. These studies provide evidence representing important divergences from collective perception.

The concept of rivers as spatial, temporal and organisational structures transporting the ecosystem processes required to maintain ecosystems is useful to the understanding of the relationship between flow and ecosystem goods and services. In this Chapter, two different approaches (*i.e.* a nested hierarchical approach and a hydronomic zoning) were suggested for integrating and managing the freshwater needs of both ecological and societal systems. Both of these approaches address the differences in spatial, temporal and organisational scales between ecological and societal systems. However, where people and the environment compete for water, the hydronomic approach has the advantage over the more biophysically-based, "scale" unit of the nested hierarchical approach, in that the former approach prioritises flexible water management strategies rather than the biophysical context of the ecosystem.

The intrinsic value of aquatic ecosystems and the services they provide to people ranks them among the key driving forces of ecologically sustainable water resources development: ecosystem goods and services integrate ecological, economic and societal systems. Maintaining ecosystem goods and services and developing ecosystem management approaches ***could be the key to managing future options*** for ecologically sustainable water resources development.

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CHAPTER 3 THE VALUE OF AQUATIC ECOSYSTEMS IN SUSTAINABLE WATER RESOURCES MANAGEMENT

CONTENTS

FIGURES

STRUCTURE OF CHAPTER 3 THE VALUE OF AQUATIC ECOSYTEMS IN SUSTAINABLE WATER RESOURCES MANAGEMENT

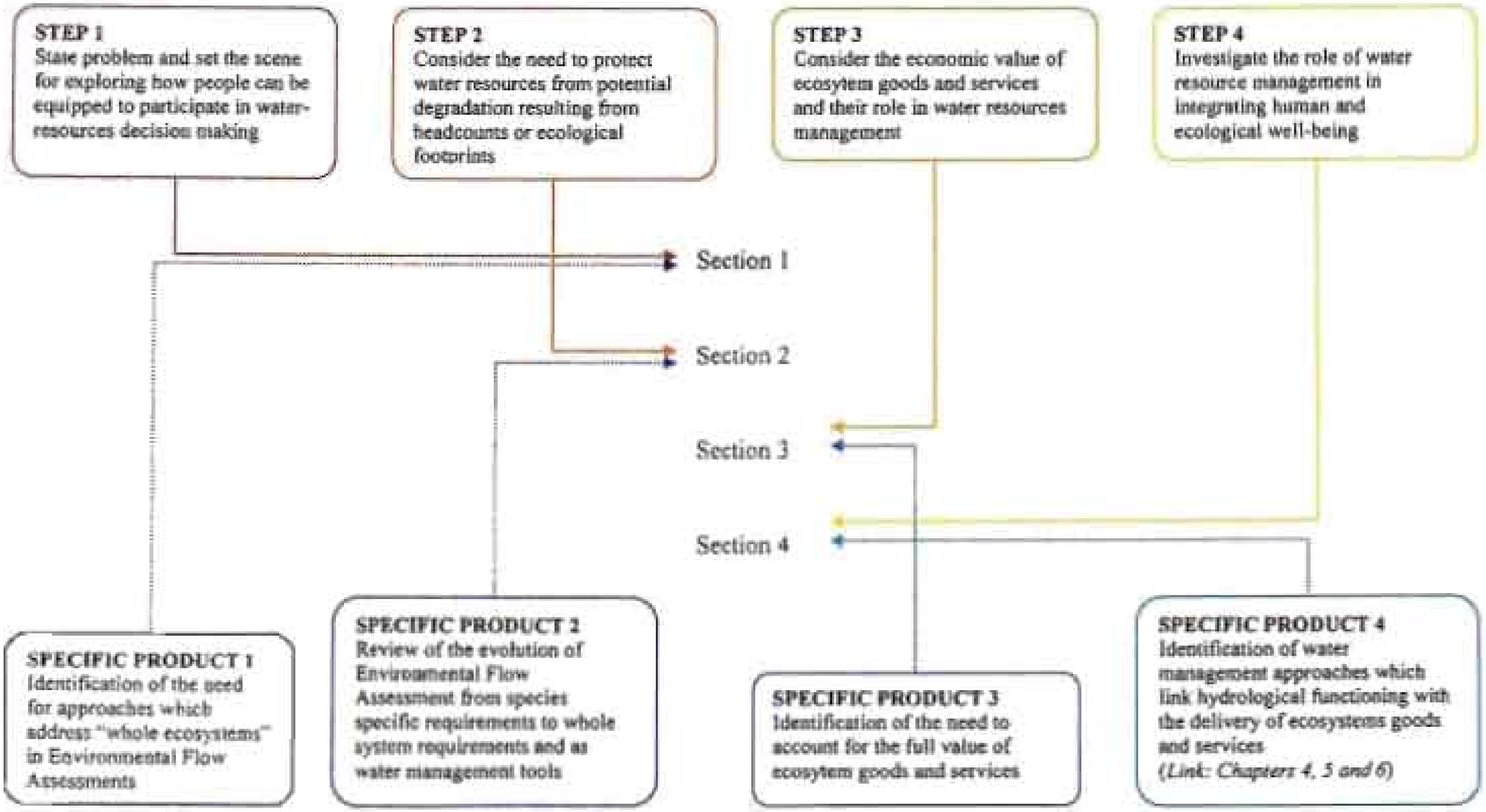

Figure 3.1 Structure of Chapter 3: The value of aquatic ecosystems in sustainable water resources management

CHAPTER 3 THE VALUE OF AQUATIC ECOSYSTEMS IN SUSTAINABLE WATER RESOURCES MANAGEMENT

1 INTRODUCTION

As the planet's water resources become scarce, over-utilised and more uncertain, increasing attention is focused on environmental flows and their "requirements" for managing sustainable water use. Water resource planning is being directed more frequently by the concept of "environmental flows". The committee of the Environmental Flows for River Systems, an international working conference on assessment and implementation [of environmental flows], held in Cape Town, South Africa, in March 2002 defined "environmental flows" as:

> *"Water that is left in a river system, or released into it, to manage the health of the channel, banks, wetlands, floodplains or estuary. [Environmental flows are] used as a management tool for conserving river species and habitats, [and are] resources used for subsistence, and recreational features. [Environmental flows are] an essential part of managing for sustainable use of water resources."*

Paradoxically, the "environment" does not have "flow requirements", nor does the environment depend on the functioning of natural systems as we know them for survival. As acknowledged by Levin (1999), natural systems are complex and adaptive, and may very well be robust in the face of environmental stress. Rather, it is the maintenance of those services on which humans depend that is fragile. The planet and its "environment" would probably survive an ecological catastrophe, as it has before (Limburg *et al.*, 2002), yet there is no reason to expect systems to be robust in protecting those services which are essential to human welfare (Levin, 1999). It is society rather than the environment that is most at risk from neglecting to "set aside" a portion of the water resource, frequently termed "environmental flows", for the functioning of the resource base.

The realisation that maintaining an aquatic resource base is necessary for the ecosystem functions, goods and services that people need to survive and prosper, has been accompanied by an evolution of the philosophies of water management and the terminology used to describe aquatic resources, their environments, ecology, use and protection. What were once referred to in water management as "compensation flows" for downstream river users, subsequently became known as the "instream flow requirements" of keystone species or species' guilds, before "environmental flows" was more commonly adopted in recognition of the need to link all components of the water resource (channel, banks, wetlands, floodplains and estuaries) flowing as surface water or groundwater. More recently in Environmental Flow Assessment, EFA, reference is made to "ecologically relevant" and "ecologically sustainable" flows, with the emphasis on the interrelationships and interdependence of organisms with each other, their environment and the hydrological cycle in "ecosystem-based management approaches". Ecological studies and ecosystem-based management approaches should, of course, include humans since people and societies interact with other ecosystem components as well as influencing the feedback mechanisms and processes linking different types of ecosystems and their components. Consequently, there has been a move to integrate the assessment of the water needs of human societies with the determination of environmental flows in approaches which are considered to be ecologically sustainable.

Together with the shift in philosophies of water management, more attention has been directed to the dynamics and valuation of ecosystem goods and services. The ecological and socio-economic evaluation of the functions, goods and services provided by natural or artificial ecosystems is a complex issue. In this regard, de Groot *et al.* (2002) presented a standardised framework for the classification of ecosystem functions, goods and services for comparative ecological economic analyses. However, such frameworks have not been widely tested and may be more suitable to decision-making processes in comparatively developed regions where water resources management includes more options than in water stressed regions. Moreover, while ecological economics employs techniques such as market pricing, replacement costs and travel costs as methods of evaluating the services of natural systems, there is a growing consensus [in river rehabilitation and restoration programs] that the evaluation of ecosystem services "is much less important than providing incentives for their conservation" (Heal, 2000). Conservation practices which integrate land and river management are highly appropriate for public, or "special" (Savenije, 2001), goods such as water which are tradable beyond the stream channel and catchment and link ecological functioning with societal functioning.

The purpose of this Chapter is to highlight the value of aquatic ecosystems in sustainable water resource management. The Chapter provides an important link between the concepts of role of the hydrological cycle and the generation of ecosystem goods and services presented in Chapter 2 with the integration of societal and ecological freshwater requirements (the remainder of this thesis, but in particular Chapter 4). The emphasis of this Chapter, and indeed this thesis, is that judicious management of aquatic ecosystems lays the foundation for a sustainable environment for society as well as the aquatic ecosystems. The Chapter begins with a brief overview of the different approaches to assessing ecological freshwater requirements (Section 2). Thereafter, the relevance of integrating a socio-economic assessment of the benefits of ecosystem functioning to water resources management is evaluated in Section 3. The theme of the societal value of aquatic ecosystem functioning is continued in Section 4, where ecosystem management is the focus for ecologically sustainable water resources management.

2 ENVIRONMENTAL FLOW ASSESSMENT AS A WATER MANAGEMENT TOOL

2.1 Introduction

There is a general perception that Environmental Flows and their Assessment are primarily for nature. Yet, the origins of contemporary EFAs were assessments of the water requirements for fish species *that were valuable to people* (*i.e.* recreational trout fishing in the USA) and there is no evidence in the literature to suggest that environmental flows are reserved for nature alone. The end targets of EFAs are people, or civil society, but misconception has arisen because scientists have generally investigated riverine, non-anthropogenic species or channel forming processes as indicators of river health, or well-being. Notwithstanding the many benefits that environmental flows generate for societal systems as well as natural systems, water resource planners still face many challenges in their assessment. Any evaluation of ecological water requirements has to be defensible, often legally (*e.g.* this is currently so under South African Law), to people who depend on the water resource for socio-economic welfare. In regions which are water stressed, either as a result of human pressure ("headcounts") or human consumption ("ecological footprints"), EFA is advocated as having a key role in sustainable water use. More frequently, decision makers acknowledge the benefits of adopting ecosystem management approaches to resolve environmental problems which have a human dimension. The main focus of this Section is to highlight the role of EFA in holistic water resource management, beginning with a brief discussion of the ecosystem-based approaches to water resources planning which have arisen as a result of the need to protect water resources from potential degradation resulting from headcounts or ecological footprints.

2.2 The Evolution of Environmental Flow Assessment

Water scarcity in many parts of the world has led to an escalating rate at which rivers have been regulated. In turn, the ensuing hydrological alteration, often accompanied by resultant environmental degradation, has led to "the establishment of the science of environmental flow assessment whereby the pattern and quality of water for ecosystem conservation and resource protection are determined" (Tharme, 2003, page 397).

However, defining the environmental flow requirements of rivers, and any other water resource, is complex, and "there is no single best method, approach or framework for determining ecological freshwater needs" (Dyson *et al.*, 2003). Different approaches to water protection arise from different water resources management goals. This has lead to different classifications of the types of methods available. For example, Jowett (1997) classified three categories, each distinguished by differences in their ecological and morphological rationales. More recently, Tharme (2003) classified six categories whilst Dyson *et al.* (2003) classify four categories, in both instances the categories being distinguished by the extent and suitability of their data requirements and analysis with the level of expertise available. The reviews of EFA by Jowett (1997), Tharme (2003) and Dyson *et al.* (2003) are comprehensive. For example, Tharme (2003) cites 207 individual methods recorded for 44 countries within six world regions. Thus, rather than reviewing the findings of these authors, the following sub-sections focus on the *water resources management basis* of three distinct categories of methods designed to the protect aquatic ecosystems.

2.3 Historical Flow Methods

Historical flow methods represent the simplest of the EFA methods and are based on obtainable streamflow values from either observed or modelled (*i.e.* synthesised) records. These methods were among the first approaches developed for EFA, yet they are still appropriate in situations where there is little ecological data and, more pertinently, where they are needed to support other environmental data and information. Historical flow methods are sometimes referred to as hydrological methods. However, this detracts from concepts of historical flow methods since even in the simplest applications there is some ecological rationale to the streamflow values prescribed (Jowett, 1997).

Typically, historical flow methods *link a fixed proportion of streamflow reduction to the historical streamflow data* of a river as in the Tennant Method (*e.g.* 10% of the Mean Annual Flow, MAF, for "poor" or "minimum" flows to represent the discharge required to sustain aquatic ecosystems at short-term survival, to 200% of MAF for flushing flows; Tennant, 1976). Consequently, assessment methods based on historical flows introduced the term "minimum flow". Minimum flow requirements can also be recommended on the basis of a threshold (or benchmark) flow derived from flow duration curves, or an exceedence probability of a specified low flow, as assessed from historical records.

The water resources management basis for using historical flow methods of assessment is that maintenance of aquatic organisms is achieved by recommending a minimum flow that is within the historical flow range. The ecological basis for this is that since existing species have survived under these conditions, it is assumed that the life supporting components of food, water temperature and quality as well as habitat suitability are sufficient at such minimum flows (Jowett, 1997). Clearly prescribing water allocations to ecosystems, based on streamflows which fall within the historical range of the streamflow regime, has merit, particularly if the historical record is representative of reasonably natural flows. However, defining water management strategies based on historical flows should reflect all aspects of the flow regime and include seasonal patterns of flow, low flows, periods of no flow and flood flows rather than prescribing fixed percentages of annual flows (Karim *et al.*, 1995).

The optimal use of historical methods is in low controversy situations (Dyson *et al.*, 2003). By implication, there is little need for stakeholder involvement in historical methods of environmental flow assessments. Consequently, historical flow methods are most suitable for application in regions where the streamflow regime is relatively invariable and where there is little conflict between ecological and societal freshwater requirements.

2.4 Desktop Methods

Desktop methods are also based on historical data such as observed or modelled streamflow records, but can additionally employ other biotic, habitat or hydraulic data or information. Consequently, Dyson *et al.* (2003) separate desktop methods into three categories, those based purely on hydrological data, those which incorporate hydraulic data or information and those which use ecological data.

The Indicators of Hydrological Alteration (IHA, Richter *et al.*, 1996; 1997) is one of the more commonly applied hydrologically based desktop methods in the northern hemisphere, particularly by The Nature Conservancy in the United States of America, although it is also been applied by *inter alios*, the Scottish Environmental Protection Agency in Scotland (Richter, pers. comm., 2002). The ecological basis for the method is that different characteristics of the natural streamflow regime drive ecological functioning and that these streamflow characteristics can be described by statistical measures as "ecologically relevant hydrological indices". When used together with the Range of Variability Approach (RVA, Richter *et al.*, 1998) this desktop method can identify periods of incompatibility between ecological and societal freshwater needs and be used to set water management targets. Consequently, the IHA method, together with application of the RVA, can be used to describe threshold (or baseline) flows for rivers where the primary management objective is the protection of the natural ecosystem functioning (Richter *et al.*, 1997). Despite the knowledge gaps as to whether the statistical measures are indeed linked to ecological functioning (Tharme, 2003), several researchers are cited by Tharme (2003) as expressing the RVA to be an holistic (Arthington, 1998) and ecologically grounded (Bragg *et al.*, 1999) approach to assess how much water rivers need.

The South African Desktop Reserve Model (Hughes and Hannart, 2003) is an example of a desktop method which incorporates ecological information, since it is based on values of instream flow requirements indices derived from research of the more comprehensive South African Building Block Methodology (BBM; King and Tharme, 1994; King and Louw, 1998). The BBM is considered to be a holistic method (see below) and will be discussed again in Chapter 4, since it is particularly relevant to South African water resources management. However, the main benefits of the South African Desktop Reserve Model are that it be applied at a nationwide scale (for South Africa) and that it can output streamflow values and indices relating to the environmental flow requirements for different management levels of water resource ecostatus (*c.f.* Chapter 4, Section 3.2.2), thereby integrating the needs of both ecological and societal needs for freshwater. The main shortcoming of this desktop method is that the streamflow values and indices are based on research extrapolated from the more comprehensive, although less spatially representative, BBM research.

2.5 Holistic Methods

Holistic methods were first documented by Tharme (1996) and refer to approaches which take account of the whole aquatic ecosystem, rather than assessing the habitat requirements of single aquatic species at specific locations or of the flow events which form the habitat. Thus these methods represent a major shift in water resource management, from incorporating an ecological grounding in environmental flow assessment to the concept of an ecosystem-based approach to aquatic management. The major benefit of holistic methods is their applicability to regions where the emphasis is on ensuring protection of the whole river system and where knowledge of the inhabitant biota is sparse or unknown (Tharme, 2003). As a consequence holistic methods are eminently suitable to arid or semi-arid regions and developing regions where there is a need for expeditious and equitable water allocations and where there is a paucity of ecological information for the assessment of the water requirements of individual species or species guilds.

Tharme (2003) describes two processes for holistic methods: a bottom-up approach and a top-down approach. The BBM is an example of a bottom-up approach to EFA where the water management goal is to construct a modified streamflow regime from first principles, on a month-by month basis for different components of the streamflow regime, which meets a particular water resource classification (ranging from natural or protected to severely modified) decided by the stakeholders of the aquatic resource (King and Tharme, 1994). Although the origins of the BBM are applications of the method in South Africa, this holistic approach has also been employed in Swaziland and Australia (Tharme, 2003). The BBM and its application in South African water resources management is discussed again in Chapter 4.

Conversely, top-down approaches are generally scenario-based, where environmental flow requirements are defined by the degree to which natural streamflow regimes can be modified by societal activities. This water resources management approach is evident in the RVA method described in Section 2.4 of this Chapter, and is the main reason why several researchers (including this author) contend that the RVA is an holistic approach to EFA. Another example of a top-down approach is the Downstream Response to Imposed Transformations approach (DRIFT, Brown and King, 2000; Brown and King, 2002) to EFAs. The premise of DRIFT is that certain ranges of streamflow components can be *reduced* from their natural range in the streamflow regime so that both ecological and societal needs for freshwater can be met. Recently in DRIFT applications, the impacts of societal reductions to various streamflow components have been linked with socio-economic factors, thereby advancing the integration of societal systems with ecological systems in terms of mitigation and compensation for stakeholders affected by upstream water use (Brown and King, 1999). This is clearly an important step towards the evaluation of ecosystem goods and services generated by different streamflow regimes.

More recently still, a scenario-based application of a combination of the BBM and DRIFT methods was applied in Zimbabwe for a situation where the rural population were dependent on aquatic ecosystem goods and services (Steward *et al.*, 2002, cited in Tharme, 2003). The relevance of this application is seen in the benefits of merging different approaches of EFA for different water resources management issues.

Most applications of holistic methods have been performed in the southern hemisphere, *e.g.* the BBM and DRIFT in southern Africa and Australia and the Holistic Approach (Arthington *et al.*, 1992; Arthington, 1998) and the Benchmarking Methodology (Brizga *et al.*, 2002) in Australia, thereby emphasising their appropriateness to integrating ecological and societal needs for freshwater resources in arid and semi-arid regions of the world. In addition, these applications emphasise the corroboration of different groups of researchers, even across continents, in striving to meet ecologically sustainable water resources management so that both ecological and societal needs for freshwater are met.

2.6 Summary

Different approaches to the assessment of environmental flows exist in different climatic and socio-economic regions. Yet, if we accept the general premise that EFA is a management tool that focuses on the amount of water needed for sustainable use of water resources, then not only should stakeholder use and consumption be included, but stakeholder contribution to decisions on water use is essential. This calls for better communication between scientists and non-scientists with a clear understanding of what "environmental flows" means to different parties. The integration of ecological and societal systems is the ultimate water resources management goal of any EFA. Thus, holistic methods represent the most advanced approaches to environmental flow assessment.

3 SOCIETAL VALUATION OF ECOSYSTEM FUNCTIONING

3.1 Introduction

Various comprehensive lists of the ecosystem goods and services provided by aquatic ecosystems appear in the literature (*c.f.* Chapter 2, Tables 2.1 and 2.2). The formulation of such lists is often a precursor to prescribing a measure of "value" or "worth" to the different benefits or functions of ecosystems, invariably in an attempt to emphasise human dependence on natural ecosystems. Empirical assessments of the value of ecosystem goods and service to humankind have been made at the global scale (Costanza *et al.*, 1997) and in some instances at a national scale (*e.g* for Scotland by Williams *et al.*, 2003). However, economic theories for the evaluation of ecosystem goods and services may not be appropriate since only a few of the benefits of ecosystem functions can be measured by economic indicators and most ecosystem processes can only be described in qualitative terms. While many theories for the conservation of ecosystems are based on their "existence value", van Wilgen *et al.* (1996) contend that this attribute "is increasingly difficult to quantify and defend, particularly in developing countries where basic human needs and economic growth are the over-riding concerns". A detailed synthesis of resource economics is beyond the scope of this thesis. However, the concept of the "economic value" of ecosystem goods and services and its role in water resources management is briefly reviewed in this Section.

3.2 The Value of Natural Systems

The criterion against which human activity and the conditions of natural systems are often measured is the "value of natural systems" in contributing to the support of human welfare, as well as sustaining that welfare and its equitable distribution (Costanza and Farber, 2002). Ecological economics generally recognises the categorisation of the value of natural systems into five different uses (Turpie and van Zyl, 2002), *viz.*,

(a) direct consumptive use values (*e.g.* the harvest of riparian plants, domestic use, irrigating crops, watering stock);

(b) direct non-consumptive use values (*e.g.* hydro-electric power, transport or recreation);

(c) indirect values (*e.g.* water quality, water flow, water storage, nutrient retention and flood control);

(d) option values (*e.g.* potential future use for a variety of societal needs ranging from pharmaceutical innovation to recreational use); and

(e) existence value (*e.g.* cultural, aesthetic or heritage value).

Whilst Turpie and van Zyl (2002) additionally describe a gradient of "tangibility" and therefore "measurability" to these five categories, the remainder of this Section focuses on the value of two distinct sub-categories of the benefits provided by natural systems, *viz*, market goods and non-market ecosystem services. These two sub-categories represent respectively the short-term benefits and long-term benefits to societal systems referred to throughout this thesis.

3.2.1 Market goods

From an economic perspective, the principal methods of valuing of ecosystem goods and services are correlated to their market price. In essence, if a good is rare and in high demand, the market price is high irrespective of whether the good is important, or even essential, to human societal systems (Heal, 2000). Therefore, the market price of ecosystem goods may not reflect their societal importance, but rather the economic indicators associated with supply and demand mechanisms. Nonetheless, some of the most crucial ecosystem goods for human survival (*i.e.* food and water) have relatively low market prices despite being valuable (Ampomah *et al*, 2004) or locally scarce in many regions throughout the world. In keeping with economic principles, "willingness to pay" could increase the market price of [these] ecosystem goods as they become scarce (Daily and Ellison, 2002). However, this concept only operates when people have surplus funds to exchange for consuming the goods. This is all the more complex in the case of water, since water is required to maintain a reliable supply of aquatic ecosystem goods and services, including water. For example, water provides direct (*e.g.* for drinking) and indirect (*e.g.* for fishing) environmental benefits to human well-being (Howarth and Farber, 2002).

3.2.2 Non-market ecosystem services

While the *provisioning* of ecological goods is the main societal consideration at the local scale, it is becoming increasingly evident that the *regulation* of essential ecological processes operates at and beyond the catchment scale. A prevailing feature of the valuation of ecosystem goods and services is the increasing recognition of the importance of processes such as nutrient recycling in sustaining societal activity. In addition, valuing ecosystem goods and services is difficult since most of them are not traded in the market place (*i.e.* non-market ecosystem services), yet may still provide both direct and indirect benefits (*e.g.* clean water benefits swimmers and commercial fishermen; Howarth and Farber, 2002). As a result, economists often use other methods such as replacement cost (*e.g.* of an ecosystem function by technology) or travel costs (*e.g.* of visiting a site of environmental interest) to assign values to the indirect environmental benefits of natural ecosystems, although these costs are inherently related to a form of market price of the particular good (*e.g.* the cost of a water purification plant or the economic benefit of ecotourism; Heal, 2000).

3.3 Calculating the Value of Ecosystem Goods and Services

Catchment management strategies which focus on the benefits of ecosystem services can evaluate the direct costs of services such as the delivery of clean water. For example, in their evaluation of a South African fynbos ecosystem, van Wilgen *et al.* (1996) cite a study by le Maitre *et al.* (1996) where the unit cost of water was evaluated under scenarios with and without the management of alien plants. Van Wilgen *et al.* (1996) concluded that management practices to remove and prevent alien plant invasions from fynbos-covered catchments not only made "sound economic sense", but would "also restore biodiversity in catchment areas and thus enable the maintenance and growth of economic enterprises based on …[conserved ecosystems]". The last part of this statement emphasises one of the main problems of attributing a value to ecosystem goods by assessing its "costs", *i.e.* that

(a) there are many "hidden" or indirect benefits provided by ecosystems;

(b) only a portion of the ecosystem function is "replaced" by technology;

(c) a scenic view cannot be discount-rated to its future value and any assessment is invariably inadequate.

Consequently, the value derived by any of these replacement, or surrogate, values is often too low, representing a lower monetary limit for the importance of the resource. Nonetheless, empirical assessments of value, including cost-benefit analysis, are typically applied since they have important roles in managing the links between societal and natural systems (Howarth and Farber, 2002).

In 1997, Costanza *et al.* (1997) estimated that the Earth's ecosystems generate goods and services worth around US $33 trillion a year to civil society, based on the calculation of a set of specified services (*e.g.* Chapter 2, Table 2.1) multiplied by a set of corresponding "shadow prices", each representing the estimated expenditure, where it existed, that would be required to purchase the ecosystem goods and services assessed. Their study of the "value of ecosystem services" (defined as VES by Howarth and Farber, 2002) and methodology of "quantity*price" has stimulated considerable discussion, not least of which has focused on the use of the term "value" and its myriad implications for a variety of different ecological, social and economic systems. Nonetheless, Howarth and Faber (2002) further examined the implications of the methodology for national income accounting for the environment and investigated the role of VES in measuring the impacts of environmental changes on human well-being at different organisational levels. At large spatial scales, VES can assist in identifying indicators of human well-being and sustainability (Daly and Cobb, 1989; Fraser *et al.*, 2005), whereas at finer spatial scales, VES can reveal information on the form and function of ecosystems as well as the multi-faceted roles of ecosystems in supporting human well-being (Howarth and Farber, 2002).

However, the VES concept, which reduces life support systems to monetary terms, has its critics. Heal (2000; cited by Howarth and Farber, 2002) contends that the "emphasis on valuing ecosystems and their services is misplaced"; Sagoff (1988; cited by Howarth and Farber, 2002) advocates that ecosystems are linked to critical social values that cannot be evaluated in monetary terms, whilst Howarth and Farber (2002, page 425) themselves emphasise that "scientific uncertainty may obscure the biophysical processes through which a given ecosystem confers benefits on human beings". Nonetheless, Howarth and Farber (2002) highlight the usefulness of economic valuation of ecosystems for societal well-being, particularly for macro-scale ecological trends. Moreover, they contend that estimates of shadow prices of ecosystem services, based on "willingness to pay", are sufficient to evaluate the impacts on societal well-being associated with small changes in ecosystem services, since they do not require detailed information on ecological function.

As well as omitting non-market ecosystem services, evaluations of the market value of goods and services at market prices tend to assume "static" societal and economic systems as well as "invariable" environmental systems. Therefore, Howarth and Farber (2002) identified a need for models which predict future environmental and economic states, thereby allowing the analyses of trade-offs between short term activities and long term well-being. Howarth and Farber (2002) advocate the concept of *full consumption (C^*)*, thereby accounting for the value of both market goods and nonmarket environmental services, as a method of evaluating potential trade-offs and propose the following model,

$$C^* = C + PS$$

where;

C^* is a measure of full consumption, including the value of both market goods and nonmarket environmental services;

C is consumption of market goods;

P is the shadow price of environmental services, or the marginal rate of substitution between consumption and environmental quality, where P measures an individual's willingness to pay for an increase in ecosystem services;

S is the provision of non-market environmental services (Howarth and Farber, 2002).

While simple, this model is beyond the realms of most EFA methods for a variety of socio-economic-political reasons, particularly in developing and water scarce countries. However, as intimated by Howarth and Farber (2002), the concept of evaluating the full consumption of environmental services could be useful in encouraging debate among stakeholders regarding the achievement of sustainable development.

3.4 Summary

The value of ecosystem management is well-known; ecosystem conservation can benefit biodiversity and people. A major problem associated with ascribing a monetary value to ecosystem goods is that there are many "hidden" or indirect benefits provided by ecosystems. While both the "option value" or "existence value" of ecosystems to civil society are undisputable, meaningful methods that quantify the potential of ecosystems to sustain future utilisation are needed.

4 INTEGRATING SOCIO-ECONOMIC VALUES AND ECOLOGICAL FUNCTIONING

4.1 Introduction

Ecosystems, and in particular, freshwater ecosystems inherently possess a high economic value. However, while people and society readily understand the concept of "economic value", economic or monetary measures of the value of ecosystem goods and services to society is just one of the techniques available to manage human interaction with natural ecosystem functioning (Costanza and Farber, 2002). Sustaining the aquatic ecosystem goods and services that people and society need or desire requires that the resource base is not undermined, since any change in the functioning of a component of an ecosystem results in a change in the goods and services that component fulfils. Thus the gap between

EFAs (Section 2 of this Chapter) and societal valuation of ecological functioning (Section 3 of this Chapter) can be bridged by accounting for *ecosystem resilience to change* and, in turn, *ecosystem capacity for sustainability*.

4.2 Resilience and Sustainability

Chapter 2, Section 3.5 introduced the concept of ecosystem resilience, *i.e.* the capacity of a system to cope with change or disturbance (Moberg and Galaz, 2005), whereas ecosystem sustainability is a common thread throughout this thesis. It is not the purpose of this Section to initiate elaborate discussion of either of these ecosystem attributes (but see *inter alios* Folke *et al.*, 2002 and Moberg and Galaz, 2005). However, ecosystem resilience and sustainability are perceived to be the links between societal and ecological systems (Rauflett, 2000) and are important attributes to consider when the benefits to human welfare, social equity and ecological sustainability cannot be easily quantified in economic or monetary terms. In addition, both ecosystem resilience and ecosystem capacity for sustainability are indisputably the key tenets of environmental security for future generations. However, the focus of this Chapter is the value of aquatic ecosystems in water resources management and any further discussion of environmental security is directed towards environmental sustainability in water resources management.

Many researchers believe that the key to environmental sustainability in water resources management lies in protecting aquatic ecosystem resilience from unnatural variability and disturbance (Falkenmark, 2005). The basic premise of this perception is that the protection of ecosystem resilience is necessary to secure the long term functioning of aquatic ecosystems, so that they can continue to deliver the ecosystem goods and services that people and society need (Falkenmark, 2005). However, the concept of resilience itself and its role in environmental sustainability is changing. Moberg and Galaz (2005, page 3) express the opinion that "the days of living with resilient, predictable and self-repairing ecosystems that 'bounce back' from stresses and disturbance are over". Resilience is not unique to ecosystems. Societal systems also possess resilience, measured by their ability to cope with, *inter alia*, environmental change without undermining essential societal functioning or the potential associated with economic and management opportunities (Moberg and Galaz, 2005). Recognition of the connectivity of resilience in *both* ecological and societal systems facilitates a paradigm shift in the perspective of water resources management from one which attempts to control environmental change to one which protects and enhances the capacity of both systems in responding to, and adapting to, change (Moberg and Galaz, 2005). This contemporary perspective in water management embraces adaptive freshwater management for strengthened resilience of both societal and ecological systems.

4.3 Ecosystem Management and Ecosystem Goods and Services

Embracing the concept of adaptive freshwater management requires maintenance of essential ecological functioning while responding to, and adapting to, change. However, adaptive freshwater management is just one of the dominant themes which fall under the umbrella of ecosystem management, a concept which balances ecosystem functions with societal requirements to address any incompatibilities between the two systems. Chapter 2, Section 4.4 presented the concept of "ecosystem management" as an integrator of ecosystem functions and human needs. This stewardship approach to environmental issues is revisited briefly in this Section as a mechanism for facilitating resilience and sustainability in water resources management.

Ecosystem management represents a holistic approach to the way humans interact with natural systems and societal values play an integral part in ecosystem management goals. The overall goal of ecosystem management should be to protect ecological functioning over the long term Grumbine (1994). However, Grumbine (1994) warns that if ecosystem management is to succeed, there has to be compromise between the goal of sustaining ecological integrity, or well-being, and the perception by some water managers that ecosystem functioning is for the provision of ecosystem goods and services for humans. This indicates that a thin line exists between the different perspectives of ecosystem well-being; some researchers (*e.g.* Kessler *et al.*, 1992) perceive that ecosystem management should be conducted so that the importance of ecosystem goods and services is upheld, whereas Grumbine (1994) advises that "there are ecological limits in any system, which constrain human use". Rather than perceiving ecosystems as resources for human utilisation, ecological functioning, including resilience, should be protected in a stewardship approach so that important functions in both ecosystems and societies are not lost. Stewardship principles which result in precautionary practices and long-term planning goals are more likely to result in healthy and productive ecosystems than short-sighted objectives of resource utilisation or a utilitarian perspective of the environment.

4.4 Scenario Assessments of Ecosystem Goods and Services

Environmental flow requirements are anthropogenic determinations of a quantity, pattern or level of flow in a river channel, wetland, groundwater store or estuary that are anticipated to provide a level of ecosystem functioning, and in turn goods and services, for societal well-being. Thus, an essential component of environmental flow determinations, and indeed adaptive water management, is the assessment of different water use scenarios by stakeholders in order to predict the likely impacts on ecological integrity. Inherently, scenario analysis is not an exact science and given the scientific uncertainties associated with ecological functioning and response to alterations in the hydrological cycle, predicting the generation of likely goods and services under different environmental circumstances is all the more complex. However, stakeholders may be in a position to correlate local knowledge of the causal effects of certain streamflow characteristics. For example, a certain river level may be associated with the migration of valuable fish species or of the arrival of the annual flood. Thus, there has been a move in ecosystem management in recent years to combine local knowledge and value judgments of affected stakeholders in resource planning, including EFA.

Facilitating stakeholder involvement in EFA requires an awareness of the water, sediment and energy flows and nutrient pathways that connect the landscape with the streamflow regime (*c.f.* Figure 1.1). The concept of hydrological connectivity has a major role to play in assisting people with the understanding of how seemingly distant areas are interlinked with the streamflow regime by hydrological processes. The importance of hydrological connectivity is revisited in Chapter 4 of this thesis. In the meantime, the relevance of participatory approaches in facilitating stakeholders to make relevant, scientifically-based, value judgments is briefly reviewed in Section 4.5 of this Chapter.

4.5 Participatory Approaches

Fraser *et al.* (2005) present an outline of the "Wellbeing Assessment" designed by Prescott-Allen (2001) for use in participatory processes for the identification of sustainability indicators. The outline of the "Wellbeing Assessment" is reproduced in Figure 3.2 and is applied in this sub-section to illustrate how the identification of different components of the streamflow regime by stakeholders can promote both ecological and human well-being.

The methodology presented in Figure 3.2 amalgamates different categories of social and environmental indicators into a single goal-oriented format to provide a measure of well-being at a variety of spatial scales, ranging from local communities to regions or even for nationwide application (Fraser *et al.*, 2005). The methodology was envisaged by Prescott-Allen (2001) to combine five general human well-being indicators (health and population, wealth, knowledge and culture, community and equity) and five general ecosystem indicators (land, water, air, species and genes and resource use), but it is equally applicable for specific ecosystem types, such as aquatic systems.

It is not necessary to elaborate on each of the steps of the outline since this sub-section is concerned only with initiating participatory approaches. It is sufficient to note that after the identification of a need for EFA (*i.e.* Step 1), stakeholder participation should be engaged early in the process (Step 2) so that the goals for water resource, or rather ecosystem, management are clearly identified. Fraser *et al.* (2005) highlight the benefits of stakeholder involvement in the choice of indicators (Step 3), even where the indicators selected cannot be measured (either from a lack of data or from difficulty in quantifying the attribute), since in such instances knowledge gaps in the data base can be identified.

Applying the methodology to EFA it could be suggested that, from a hydrological perspective, indicators of intra- and inter-annual streamflow variability and predictability are strong contenders as indicators of ecosystem well-being. However, the selection of indices representing appropriate streamflow characteristics can be complicated by the plethora of temporal hydrological indices available (Olden and Poff, 2003) and by the identification of a representative spatial resolution. For example, there may be distinct differences between two streamflows regimes as a result of different rainfall regimes, topography or geology despite their being only a few hundred kilometres apart. These concerns are the focus of Chapter 5 of this thesis.

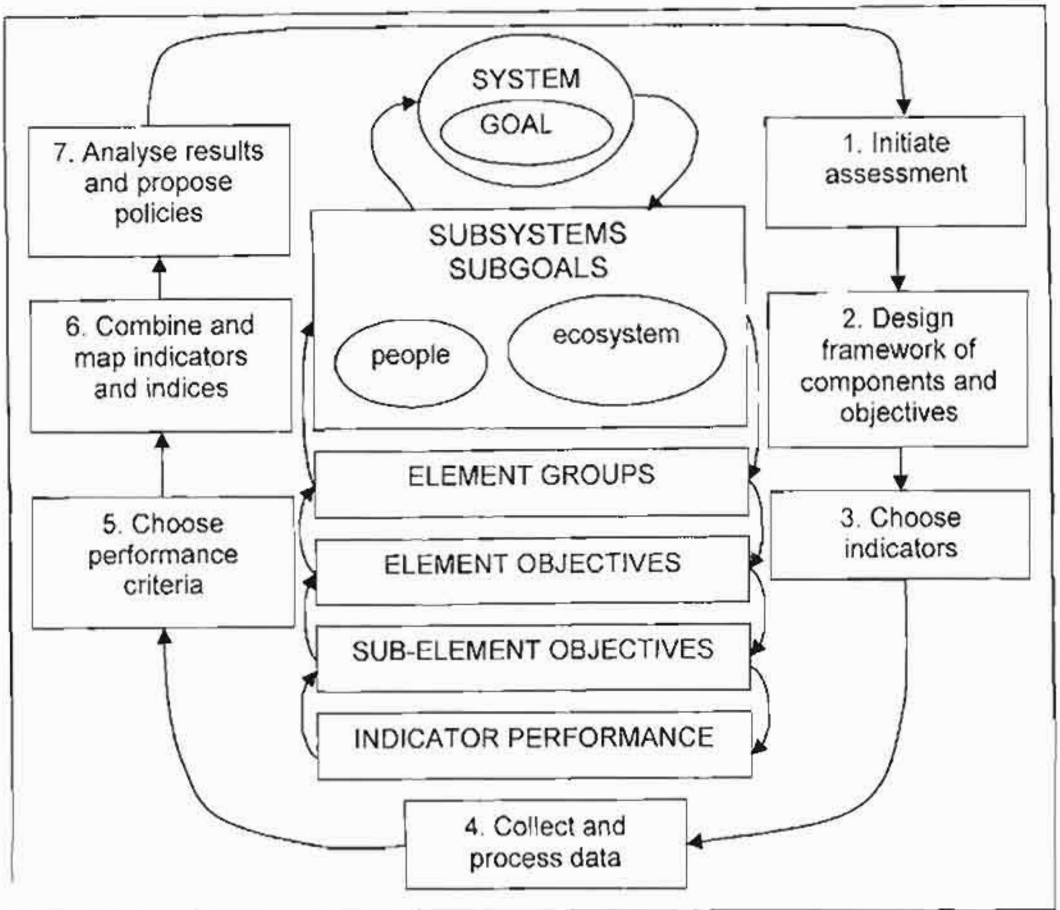

Figure 3.2 Structure of "Well-being Assessment" for integrating societal values and ecological functioning. Steps 1 to 7 identify steps in community participation process. The linkages in the inner portion of the diagram show how indicators of different ecosystem components are integrated into an overall assessment, (modified after Fraser *et al.*, 2005, which was adapted from Prescott-Allen, 2001)

However, the main benefit of participatory methods, operated in a stakeholder forum, is that the process facilitates community empowerment in a way that more traditional development approaches fail to achieve (Fraser *et al.*, 2005). Fraser *et al.* (2005) demonstrate this with the comparison of bottom-up and top-down approaches in environmental management projects. Bottom-up and top-down approaches to EFAs were described in Section 2 of this Chapter, whereby the emphasis was on the form and function of the modified streamflow regime. However, the distinction between these two approaches is also useful for the participatory processes in EFAs. This is all the more relevant where EFAs are to achieve acceptability and "buy-in" from the portions of society most directly affected by any modifications to the streamflow regime. However, whilst top-down approaches are considered to be advantageous to EFA (lest an important ecological component be omitted in a bottom-up approach; Tharme, 2003), Fraser *et al.* (2005) emphasise the benefits of bottom-up approaches over top-down approaches to providing community empowerment and sustainable environmental management. In the past, water resource planning projects have risked failure as a result of top-down approaches which underestimate the importance of stakeholder input in critical ecological objectives. The research by Fraser *et al.* (2005) emphasises a growing need for the integration of local knowledge, scientific research and policy making in environmental assessment studies to be approached from a bottom-up basis, *i.e.* giving precedence to indicators selected by stakeholders.

4.6 Summary

Freshwater management requires an ability to integrate ecological and social resilience in order to achieve sustainable development. Ecosystem management is advocated as an integrator of both human and ecological well-being. Adaptive freshwater management, one of the main themes of ecosystem management, is the vehicle which allows water managers to deal with uncertainty in the scientific understanding of ecosystem response to anthropogenic and environmental change. Predicting ecosystem response to change is complex, but stakeholder participation can facilitate useful sustainability indicators, empower local communities and encourage buy-in in EFAs.

5 DISCUSSION

One of the main threads interlinking the Chapters of this thesis is that humans and their societies are integral parts of ecosystems. Given the evolution of EFAs, from defining the flow requirements of individual, non-human species to addressing the needs of whole ecosystems, the next phase of EFA methods are likely to incorporate greater recognition of societal systems. Combinations of top-down and bottom-up approaches in holistic methods of EFA are already in use in water resources management (Tharme, 2003), yet there is a growing need for greater inclusion of economic considerations.

Assessing the value of ecosystem goods and services involves identifying the ethical or philosophical basis of value as well as the techniques for measurement thereof (Goulder and Kennedy, 1997). The economic value of aquatic ecosystems through the provisioning of market goods including food, freshwater and biological products is readily appreciated by water users. However, the economic value of aquatic ecosystems through the intangible services of environmental process regulation, biological organisation and life enrichment (*c.f.* Chapter 2, Box 3) are often overlooked, even by the main beneficiaries, *viz.*, people and society. Even attempts to measure the full consumption value may underestimate the benefits to human welfare, social equity and ecological sustainability.

Ecosystem functioning, including ecosystem resilience, needs to be conserved in order to provide a reliable supply of life sustaining services. Heal (2000) maintains that the most important economic factor in the conservation of ecosystems that support human societies lies in making conservation more attractive than any other use. This involves correlating some of the societal importance attached to those services into livelihoods or income and ensuring that stakeholders derive economic benefits in return for any stewardship of the ecosystem (Heal, 2000).

The success of future "combination" approaches to EFA will also need to take greater cognisance of issues of scale. Typically, EFAs are applied in-channel and at the micro scale, generally in the range of tens of metres. This scale may be sufficient for identification of ecological functioning, but society operates beyond the channel and at a far larger scale. While chemical, biotic, geomorphological and hydrological indicators of river health do convey some information regarding human well-being, it is necessary to look beyond the channel scale to explore the hydrological indicators of social systems. The difficulties associated with spatial and organisational scale in ecosystems were highlighted in Chapter 2 and will be revisited in Chapters 4, 5 and 6 of this thesis.

Society's ability to identify an adequate quantity and quality of water to maintain the ecosystem functioning needed to generate desired goods and services requires an understanding of ecosystem response to present and future utilisation of water resources. However, given the scientific uncertainties associated with ecological functioning and response to alterations in the hydrological cycle, there has been a move to embrace the concepts of resilience and sustainability through "ecosystem management frameworks".

Ecosystem management is an evolving concept which balances ecosystem functioning and human interaction in a stewardship approach, with the emphasis on ecologically sustainable resource use (Yibarbuk *et al.*, 2001). Thus ecosystem management is based on the inherent value and interconnectivity of ecosystem functioning and human requirements (Fraser *et al.*, 2005) and can be envisaged as an umbrella strategy in water resources management for a variety of management tools, including EFA and adaptive management. In adaptive management, scientific knowledge and understanding is assumed to be "provisional", which "allows water managers to remain flexible and adaptable to uncertainty" (Grumbine, 1994). This is an appropriate springboard for the development of more inclusive participatory approaches to EFA, at scientific, managerial and stakeholder levels. Where EFA is conducted at a relatively large spatial or organisational scale, top-down approaches to the selection of ecological indicators may be adequate until the feedback processes of ecological and societal functioning and response are available. At finer spatial or organisational scales, bottom-up approaches may be more suitable since the feedback processes benefit from local knowledge (Fraser *et al.*, 2005). Nonetheless, in the absence of local knowledge or in desktop studies, top-down approaches may be appropriate even at the finer spatial or organisational scale since the basic premise of adaptive management is that ecologically sustainable water resources management is achievable in the long term (Richter *et al.*, 2003).

The capacity of humans to degrade aquatic ecosystems, either by increasing headcount or by ecological footprint, has led to changing attitudes to water resources problems. Different public attitudes and values have led to different perspectives on the ways in which society should interact with their environment. Some researchers support a preservationist and precautionary approach to environmental protection, maintaining that once that the limit of natural ecological resilience has been breached, ecosystem functioning is disrupted and cannot be retrieved, whereas others advocate a more inclusive, ecosystem approach to resource management as the best way to protect the environment and ensure sustainable development. Whilst preservationist strategies seek to minimise human-caused change, ecosystem management strategies seek to maximise the long-term benefits to society while conserving biodiversity.

Howarth and Farber (2002) propose that society's willingness to pay for ecological conservation and rehabilitation projects often outweighs the associated costs, thereby confirming the importance and value that people attach to their environment. Despite the undoubted priceless value or worth of the many conditions and processes associated with natural ecosystems that benefit humans, there is a price attached to maintaining or restoring ecosystems so that the delivery of goods and services can be sustained. In water scarce regions such as South Africa, this "cost" has been perceived as being less water available for allocation to artificial systems and, consequently, reduced economic productivity or more judicious (and often more expensive) management practices to protect the resource from degradation. Focusing on the benefits of long-term planning for natural resources is a tenuous operation in societal systems where meeting the present basic needs of people is a hardship and the expenditure of public funds is "difficult to justify in the absence of sound economic evaluation" (van Wilgen *et al.*, 1996).

All things considered, monetary values cannot be ascribed effectively to the essential services that aquatic ecosystems provide and it may be that trading of other drivers of non-market environmental services, such as hydrological functioning, could be an acceptable alternative.

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CHAPTER 4 THE ROLE OF THE RESERVE IN SOUTH AFRICAN WATER RESOURCES MANAGEMENT

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TABLES

STRUCTURE OF CHAPTER 4 THE ROLE OF THE RESERVE IN SOUTH AFRICAN WATER RESOURCES MANAGEMENT

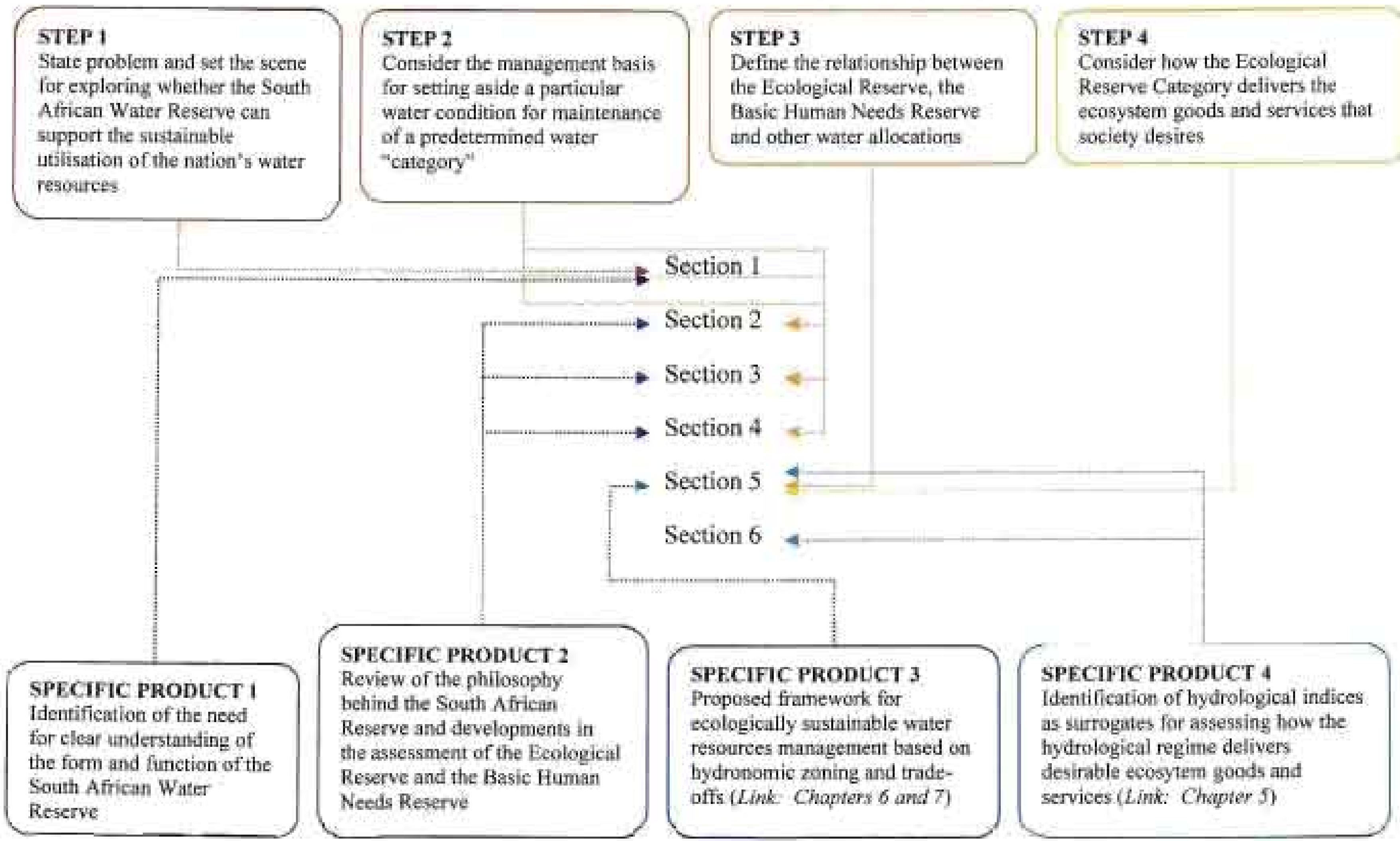

Figure 4.1 Structure of Chapter 4: The role of the Reserve in South African water resources management

CHAPTER 4 THE ROLE OF THE RESERVE IN SOUTH AFRICAN WATER RESOURCES MANAGEMENT

1 INTRODUCTION

There has been increased awareness over the past three decades of the fragile state of the natural environment, and in particular the balance between hydrological and ecological systems in arid or semi-arid regions. In both developed and developing nations, shifts in environmental policy are beginning to encompass a more responsible, or stewardship, approach to water resources management, where water allocation and "commodity-sharing" are being replaced with environmental governance and "benefits-sharing".

A stewardship approach designed to protect the aquatic ecosystem goods and services upon which society depends and, where appropriate, the linkages to terrestrial ecosystems, is envisaged to ensure the sustainable and equitable allocation of the South Africa's limited water resources, since healthy ecosystems are better able to support use than degraded ones (DWAF, 2004). South African water law and policy was radically overhauled in the 1990s to address the inequitable allocation of the nation's water and unsustainable water resource developments which were threatening to degrade the resource base (MacKay, 2000). Several water law policy documents addressing the concerns of sustainability and equity were formulated, with the key environmental principle being that "the quantity, quality and reliability of water required to maintain the ecological functions on which humans depend shall be *reserved* so that the human use of water does not individually or cumulatively compromise the long-term sustainability of aquatic and associated ecosystems" (Palmer, 1999). These innovative changes to the nation's water law resulted in the principles of aquatic resource protection and sustainability being embodied in the South African National Water Act (NWA) of 1998, in the form of two legal water rights, *viz*:

(a) a human right to safe, accessible water for drinking and sanitation (defined as the World Health Organisation's designated minimum of 25 litres per person per day) and

(b) an environmental right to the quantity, quality and reliability of water required by aquatic ecosystems to ensure sustainable use.

The water required to meet these rights is referred to as the Reserve; the water required to meet the human right being known as the Basic Human Needs Reserve (BHNR) and the water required to meet the environmental right being known as the Ecological Reserve (ER). Supporting the BHNR, as well as any other water use, depends on the maintenance of aquatic ecosystem health, since water supply and water quality as well as wide array of ecosystem goods and services require an acceptable state of ecosystem functioning, which is essentially the basis of the ER.

While by the time of writing seven years have lapsed since the legal requirement for a Reserve was promulgated in 1998, many of the issues around its form and function, as well as its implementation, monitoring and, where appropriate, re-assessment are still unclear. Despite numerous and well-intentioned initiatives by the DWAF in rising to the challenge of disseminating the Reserve, there remain large knowledge gaps, even in the interpretation of the purpose of the Reserve (van Wyk *et al.*, 2006). Basic misunderstandings of the role of the Reserve in ecologically sustainable water resources management threaten to undermine meaningful assessments of both ecological and societal freshwater needs. Addressing this confusion requires re-examination of the relationships among the ER, the BHNR and other water uses and allocations. This is all the more pertinent since concern has been voiced that the protection of aquatic resources, and, in turn, the management of the Reserve, could be implemented by a system which simply controlled the number of water use licences issued (Rogers *et al.*, 2000). In its Draft Position Paper for Water Allocation Reform, the DWAF has stated that "the water allocation process must allow for the sustainable use of water resources and must promote the efficient and non-wasteful use of water" (DWAF, 2005). Thus, new ways of approaching the compromise between ecological and societal needs for freshwater water are required. This requires that the focus of freshwater ecosystems be extended beyond the aquatic resource, so that societal activities on the catchment are linked to the protection of instream flows. In this way the full benefits of ecosystem goods and services can be assessed.

This Chapter focuses on the role of the Reserve in maintaining the level of aquatic ecosystem functioning required for sustainable water resources development in South Africa. First, a brief overview of the philosophy behind the Reserve and its function in the South African National Water Resources Strategy is provided (Section 2). Thereafter, the Reserve Determination processes are examined for both the ER (Section 3) and the BHNR

(Section 4). The relationships among the ER, the BHNR and other water uses operate at a catchment-based scale. Thus, an ecosystem-based approach which recognises the hydrological connectivity of the catchment landscape in linking aquatic and terrestrial systems is proposed for ecologically sustainable water resources management (Section 5).

2 THE PHILOSOPHY BEHIND THE SOUTH AFRICAN WATER RESERVE

2.1 Introduction

Water law reform in South Africa in the 1990s led to the Resource Directed Measures of the National Water Resource Strategy (DWAF, 2004) to sustain aquatic ecological functioning, *including the setting aside of a Reserve of water* which is defined as the "quality, as well as the quantity and assurance, of water required to protect basic human needs and aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource" before all other water uses are considered (NWA, 1998). Thus, the function of the Reserve in the South African water resources management process is to ensure the ecologically sustainable use of the nation's water.

2.2 Conceptual Overview

The Reserve has its foundation in the Constitution of the Republic of South Africa (Act Number 108 of 1996, known as the Bill of Rights), which was formulated to ensure a democratic and transparent society and which also aspires to enrich the quality of life of all citizens of South Africa. Chapter 2 of the Act stipulates that "everyone has the right to an environment that is not harmful to their health and well-being, to have the environment protected for the benefit of present and future generations" (Section 24) "and to have access to sufficient food and water" (Section 27) (DWAF, 2003). This legislation set the scene for the development of the White Paper on a National Water Policy for South Africa (the NWA, 1998) and comprised the following (DWAF, 2003):

(a) The Water Law Principles of 1996 (focusing on integrating catchment management issues, DWAF, 1996);

(b) The Water Services Act of 1997, (outlining the minimum water services to which people are entitled, WSA, 1997);

(c) The National Environment Management Act of 1998 (providing for the protection of aquatic ecosystems); and finally

(d) The National Water Act of 1998, NWA (with its emphasis on shifting water resources development to ecologically sustainable water resources management).

Thus the move towards ecologically sustainable water resource management in South African water policy paved the way for the Reserve, *i.e.* that part of a water resource which is set aside to ensure resource protection and sustainable resource use.

The principles of sustainability and equity are raised in the NWA (Box 1) through the concepts of the Reserve and the licensed use of "spare" water (DWAF, 1999; Pollard *et al.*, 2002). The setting aside of a Reserve of water to provide for basic human needs and ecosystem protection before all other water users, and the equitable allocation of surplus water for the sustainable management of the nation's scarce water resources (Box 2) is anticipated to break the deadlock among social equity, ecological integrity and economic growth (DWAF, 2005). To achieve this, the participation of all stakeholders (water users) is paramount in the formulation of a "vision for the resource" with goal-oriented frameworks for the management of catchment water resources.

Box 1 Principles of the NWA (NWA, 1998)

"Sustainability and equity are identified as central guiding principles...recognising the basic human needs of present and future generations, the need to protect water resources, the need to share some water resources with other countries, the need to promote social and economic development through the use of water ..."

Box 2 Department of Water Affairs and Forestry's definition of a water resource

A water resource is an ecosystem, including the physical or structural aquatic habitats (both instream and riparian), the water, the aquatic biota, and the physical, chemical and ecological processes which link habitats, water and biota (DWAF, 1999).

2.3 Vision for the Water Resource

The NWA provides a framework for the development of policies and tools to use in water resources management and, accordingly, the DWAF has developed a process for water resources management which addresses the principles of sustainability and equity (Figure 4.2). This process can be initiated from several of the steps shown in Figure 4.2, although a common staring point is the "vision for the resource". The process includes:

(a) Resource Directed Measures (RDMs) which identify a desired level of protection for a water resource (the management class) and set quantitative or qualitative goals (resource quality objectives, RQOs) to ensure the protection of the water resource, including a catchment Reserve comprising the ER and the BHNR,

(b) demand management, with the development of an allocation plan to keep utilisation within the limits of protection,

(c) source directed controls, designed to control impacts on water resources through the registration and licensing of water use in order to achieve the resource quality objectives and

(d) monitoring the status of water resources (MacKay, 2000).

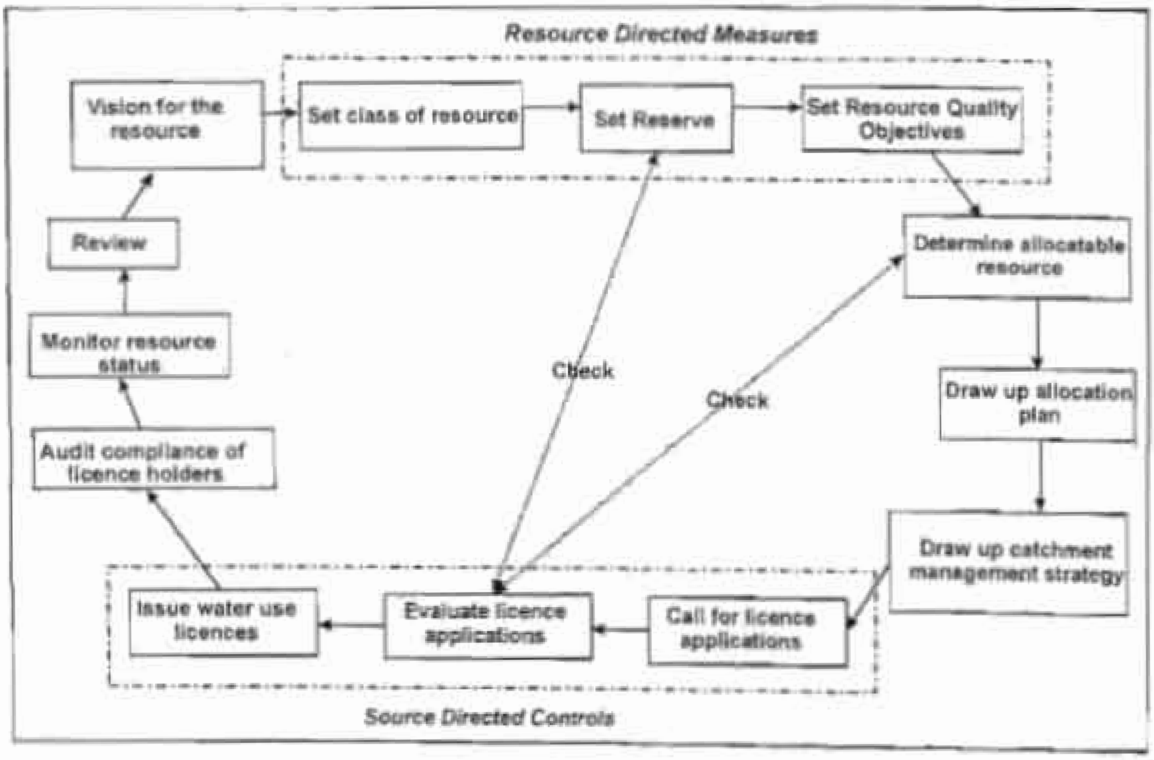

Figure 4.2 Schematic of the Water Resource Management process in South Africa (DWAF, 1999)

2.4 The Water Resource, Ecosystem Goods and Services and the Reserve

The water resource provides a range of instream and off-stream ecosystem goods and services to society, including water supply and abstraction, waste treatment, food production, nature and biodiversity conservation as well as sites for cultural needs; (Palmer *et al.*,2005; *c.f.* Chapter 2, Section 2.2). The crux of the DWAF water policy is that water resources should be managed as renewable resources to sustain a level of ecosystem goods and services that is beneficial to society (MacKay, 2001; DWAF, 2005). Sustaining ecosystem goods and services depends on maintenance of aquatic ecosystem health. The need to identify and select an acceptable state of ecological health so that ecosystems can continue to supply society with these ecosystem goods and services forms the basis for Reserve determinations. In essence, the Reserve is legislated as a water allocation to protect aquatic ecosystems, in order that,

(a) a desired suite of ecosystem goods and services can continue, and so that

(b) an equitable balance can be reached between the needs of instream and off-stream water users. The process for this water allocation is addressed by the Resource Directed Measures.

2.5 Resource Directed Measures

The full set of RDMs (*i.e.* the system for classifying water resources, the determination of the Reserve and the RQOs for the resource in accordance with its class) at different levels of confidence (ranging from low to high) is still being developed (DWAF, 2004). Increasing levels of confidence require greater commitment in expert knowledge and time and, consequently, comprehensive EFAs incur greater associated costs than simple methods of assessment (*c.f.* Section 3.2 of this Chapter). The confidence level associated with RDM depends on the ecological, social and economic importance and vulnerability of the water resource as well as the extent of the impacts of any probable water use (DWAF, 2001). Determining the importance of the water resource is a critical step in the quantification of the ecosystem goods and services that can be expected, particularly in dry seasons or periods of drought.

A consistent classification system, applicable to all water resources, is required which accounts for the different characteristics of rivers, wetlands, impoundments and estuaries, as components of surface water resources, as well as the unique features of groundwater resources and their linkages to surface water resources. More specifically, the classification system should describe different levels of ecosystem health and, from these, the levels of tolerable stress or risk to ecosystem health (thresholds) as well as the level of acceptable use (range of each level of ecosystem health) of the water resource (IWR Environmental, 2003).

Three management classes, *viz.*, Natural or Protected, Good and Fair, have been proposed for the generic national water resources classification system (Table 4.1), representing three conditions of use ranging from "pristine" to "workhorse" (DWAF, 1999) and integrating the requirements of

(a) aquatic ecosystem protection for ecological integrity (represented by Ecological Reserve Categories, ERCs, A to D; King *et al.*, 2000),

(b) basic human needs for social equity (represented by 5 classes assuming basic treatment set out in the Assessment Guide (DWAF, 1998), and

(c) water users for economic growth (representing agriculture, industry, bulk domestic supply, recreation), according to the water quality ranges given in the South African Water Quality Guidelines (DWAF, 1996).

A fourth classification, *viz.* Poor, exists for resources, or parts of resources, that as a result of over-exploitation are already in a condition that are unacceptably ecologically degraded. The classification system provides for the development of RQOs for the rehabilitation of such resources, but the process is not included in Reserve determinations (DWAF, 2004).

Each class should be managed within a range of chemical and physico-chemical, biological and hydro-geomorphological characteristics, with thresholds representing observable or measurable biotic response to change in the value of the characteristics designated to each class (DWAF, 2004). Thus, the class of a resource, the Reserve and the resource quality objectives are inter-linked within the vision for the water resource, since

(a) the Reserve describes the quantity and quality of water to meet basic human needs and to protect ecosystems, and

(b) RQOs describe measurable biological, chemical and physical goals that define a resource according to particular management class. For example, RQOs may be defined for the quantity, pattern and timing of instream flow; water quality; the ecological integrity of both the riparian habitat and aquatic biota (DWAF, 2004). Furthermore,

(c) RQOs must be directed to the needs of water users as well as accounting for the class of the resource (DWAF, 2004).

Table 4.1 Classification system for the management of South Africa's water resources (after DWAF, 2004)

Class	**Condition of Water Resource**
Natural or Protected	• Human activity has caused *no or minimal changes* to the historically natural structure and functioning of biological communities, to the hydrological characteristics, or to the bed, banks and channel of the resource. • Chemical concentrations are not significantly different from background concentration levels or ranges for naturally occurring substances. • Concentration levels of artificial substances do not exceed detection limits of advanced analytical methodologies. *All of the above conditions must be satisfied.* The Protected class is a reference condition. Resources in other classes will be defined in terms of the degree of alteration from conditions of *no or minimal changes*. Criteria for classification as Protected include: • National or international heritage site or areas • Biodiversity • The Ramsar Wetlands Convention • Economic (tourism, medicinal plants) and social / cultural considerations • Areas designated as Protected under any other legislation
Good	The condition of the resource is slightly-to-moderately altered from the Protected Reference conditions.
Fair	The condition of the resource has been significantly changed from Protected conditions.
Poor	The condition of the resource is so severely modified that rehabilitation to anything approaching a natural ecosystem would be impossible or prohibitively expensive.

Thus RQOs refer to sets of rules designed to protect the water resource quality at a level of risk defined by the catchment stakeholders based on the resource management class (DWAF, 2004). RQOs for a water resource are a statement of the vision for a water resource, relating to, *inter alia*, the Reserve of that water resource and, as such, should be measurable. The concept of a vision for the water resource is innovative in that the Minister of Water Affairs and Forestry will only consider the RDMs if they have been reached through stakeholder participation. Essentially, the vision for the water resource is based on the level of risk that stakeholders are prepared to accept for using the water resource; a balance must therefore be sought between the need to protect and sustain the water resource and the need to develop and use the resource. A resource with high ecological significance should incur less risk than a resource which is either already substantially modified or is more resilient to utilisation (MacKay, 2000).

2.6 Summary

South African water law promotes an enabling approach to the management and utilisation of the water resource with goals of sustainability and equity to meet societal and ecological needs. This approach is initiated through Resource Directed Measures, a strategy for resource protection through the classification of the water resource, the establishment of the Reserve, and the setting of Resource Quality Objectives. Classification of the water resource describes its current and potential utilisation. The Reserve describes the quantity and quality of water to meet basic human needs and to protect ecosystems. Resource Quality Objectives are the management approaches to maintain the ecostatus of water resource at a desired level of classification.

An important component of the RDMs is the "setting", or determination, of the Reserve (*c.f.* Figure 4.2). The following two Sections of this Chapter provide a review of the development of the current methods of Reserve Determination in South Africa, dealing first with the determination of the ER (Section 3) before the determination of the BHNR (Section 4).

3 DETERMINING THE ECOLOGICAL RESERVE

3.1 Introduction

The Ecological Reserve is defined as the "quantity and quality of water required…to protect aquatic ecosystems in order to secure ecologically sustainable development and use of the relevant water resource" (DWAF, 1998). The methods for the determination of the Reserve are still being developed. Nevertheless, the approach is evolving to incorporate methods which address the inter-connectivity of different components of the aquatic resource as well as the links to terrestrial ecosystems and society which inhabits them.

Protecting aquatic ecosystems, in order to maintain aquatic ecosystem health and an acceptable state of ecosystem functioning, requires an understanding and assessment of their water requirements, not only to protect the intrinsic value of the ecosystem itself, but also to protect the value of ecosystem goods and services for people (MacKay and Moloi, 2003). However, the determination of the streamflows required to sustain the physical and biological processes of river systems, groundwater, wetlands and estuaries, which, in turn provide society with a wide range of ecosystem goods and services, presents major challenges to scientists, stakeholders and water managers.

3.2 The Process of Determining the Ecological Reserve

Although the NWA (NWA, 1998) provides for a Reserve for all aquatic ecosystems, quantification methods in ER assessments are still best developed for the water requirements of river systems. Moreover, as a result of earlier environmental flow assessments in South Africa, which focused mainly on the instream flow requirements (IFRs) of river systems, the determination process for the water quantity component of ERs is better developed than it is for the water quality component. A method for quantifying environmental water quality requirements in an ER assessment has been proposed by Palmer *et al.* (2005). Nonetheless, this Section deals with the development of methods for quantifying the environmental flow component of an ER assessment.

Reserve determinations (both ER and BHNR) are to be performed at a *resource unit* scale within catchments, where each resource unit is sufficiently different to warrant its own Reserve determination and RQOs (DWAF, 1999). Resource units are essentially a finer resolution of the "national level eco-regions", where (currently 30) areas of relative homogeneity in physiographic, geographic, hydrological, and biological features have been mapped for the whole of South Africa (IWQS, 2001). The breakdown of resource units from eco-regions may be based on:

(a) the integrity of the instream and riparian habitat, based on ecological indicators,

(b) the physical character of the stream, based on geomorphic features, and

(c) river network system operation, thereby ensuring that the resource units are compatible with management requirements (DWAF, 2003).

The scale of resource units is anticipated to be in the order of several kilometres of a river reach in the case of surface water resources (DWAF, 2002), but may be as wide as a catchment or as deep as groundwater (DWAF, 1999). In a similar process, groundwater regions based on the homogeneous biophysical attributes of aquifers (*e.g.* the Groundwater Regions of South Africa; Vegter, 1995) are discretised into groundwater *response zones* (the equivalent of the surface water *resource units*) to represent groundwater management units (Xu *et al.*, 2002).

At the time of writing (2006) there are four levels of confidence of assessment for the preliminary determination of the ER, ranging from a desktop exercise (low confidence level) taking only a few hours to complete, to a comprehensive study (high confidence level) involving the integrated expertise of ecologists, hydraulicians, geomorphologists, hydrologists and social scientists and taking several years to complete. The selection of ER determination approach depends on the ecological importance or significance of the water resource, the extent of water use in the catchment and any proposed infrastructural development, with greater data requirements at both temporal and spatial scales for comprehensive studies. Moreover, even within the comprehensive level of ER determination, confidence levels are set for the following;

(a) the data sets of each of the different biophysical IFR components evaluated,

(b) the hydraulic / habitat relationship, and

(c) a final integrated evaluation of the IFR components and the hydraulics.

Notwithstanding the scope for innovative research and the assimilation of new tools in any potential methods, the DWAF can be challenged to compensate for reductions in any water use when issuing licences. Consequently, any EFA determination must be scientifically based and legally defensible (MacKay, 2001). Moreover, since water resources within many South African catchments are stressed, there has been increasing focus on the links between the ecological water requirements of river systems (with regard to the streamflow regime), or IFRs, with the socio-economic implications of different streamflow regimes in relation to the generation of ecosystem goods and services for human systems. As a result, the methods adopted in the full ER determination process have evolved rapidly since the late 1990s. However, a comprehensive level ER determination for a river comprises the following elements (DWAF, 1999; IWR, Environmental, 2003):

(a) identifying river reaches (resource units) and research study sites therein,

(b) understanding the natural or historical state (reference conditions) of the river,

(c) understanding the present state (with respect to quality and quantity for both societal and ecological systems) of the river,

(d) understanding the reasons for changes in the river (*i.e.* flow related or non-flow related activities),

(e) determining the trajectory of any anthropogenic changes to the present state of the river, if present conditions in the catchment continue, in the short-term (*e.g.* less than 5 years) as well as the longer-term (*e.g.* more than 20 years),

(f) determining how stakeholders would like to see the river system functioning in the future (on the basis of ecological and social importance of the river),

(g) determining the impact this would have on the river system, and

(h) making recommendations on the Reserve after assessing the ecological, social and economic implications of different scenarios (relating to different ERCs).

In current Reserve Determination methods, these elements are addressed by three main components of the ER determination process, *viz.*

(a) resource classification (from both societal and ecological perspectives),

(b) determination of environmental flow requirements (EFRs), or instream flow requirements (IFRs), for a range of ecological protection levels and

(c) examination of options and consequences (both societal and ecological) through flow-related scenarios.

The three components are described further in the following sub-sections.

3.2.1 Resource Classification

As described in Section 2.5 of this Chapter, the NWA (NWA, 1998) provides for a water resource classification system. A preliminary classification system, aimed at optimising sustainable water resource use, is currently applied in ER determinations (DWAF, 2004) and provides a basis for identifying and selecting different levels of ecological health and "for setting qualitative and quantitative resource quality objectives, RQOs", (Palmer *et al.*, 2005). The function of the RQOs (*c.f.* Section 2.5 of this Chapter) is that they should equate to specific management objectives or targets for each of four ecological reserve categories (ERCs) of desired or acceptable ecological integrity, *i.e.* A through to D (*c.f.* Table 4.2), and "provide a means for defining management thresholds for the water resource" (Godfrey and Todd, 2001). Environmental Flow Requirements (EFRs) are then set to achieve these endpoints. The attributes of the different ERCs, applied to describe the ecostatus of aquatic biota (*i.e.* fish, invertebrates and riparian vegetation), river processes (hydrological and geomorphological functioning) and water quality are summarised in Table 4.2. The volume and quality of water allocated to the ER is correlated with the status of the ERC. It is important to note that the ecostatus of a water resource, *i.e.* the ERC, is related to the different management classes proposed for the national water resources classification system in Table 4.1 (*c.f.* Figure 4.3).

Table 4.2 Descriptions of Ecological Reserve Categories based on definitions of generic present ecological state categories (DWAF, 1999, Volume 3 cited in IWR Environmental, 2003)

Class	**Description of Ecological Reserve Category**
A	Natural; • The resource base has not been decreased. • The resource capacity has not been exploited.
B	Largely natural with few modifications; • The resource base has been reduced to a small extent. • A small change in natural habitats and biota may have taken place but the ecosystem functions are essentially unchanged.
C	Moderately modified; • The resource base has been decreased to a moderate extent. • A change of natural habitat and biota has occurred, but the basic ecosystem functions are still predominantly unchanged.
D	Largely modified; • The resource base has been decreased to a large extent. • Large changes in natural habitat, biota and basic ecosystem functions have occurred.
E	Seriously modified; • The resource base has been seriously decreased. • The loss of natural habitat, biota and basic ecosystem functions is extensive.
F	Critically modified; • The resource base has been critically decreased. • Modifications have reached a critical level and the resource has been modified completely with an almost loss of natural habitat and biota. In the worst instances the basic ecosystem functions have been destroyed and the changes are irreversible.

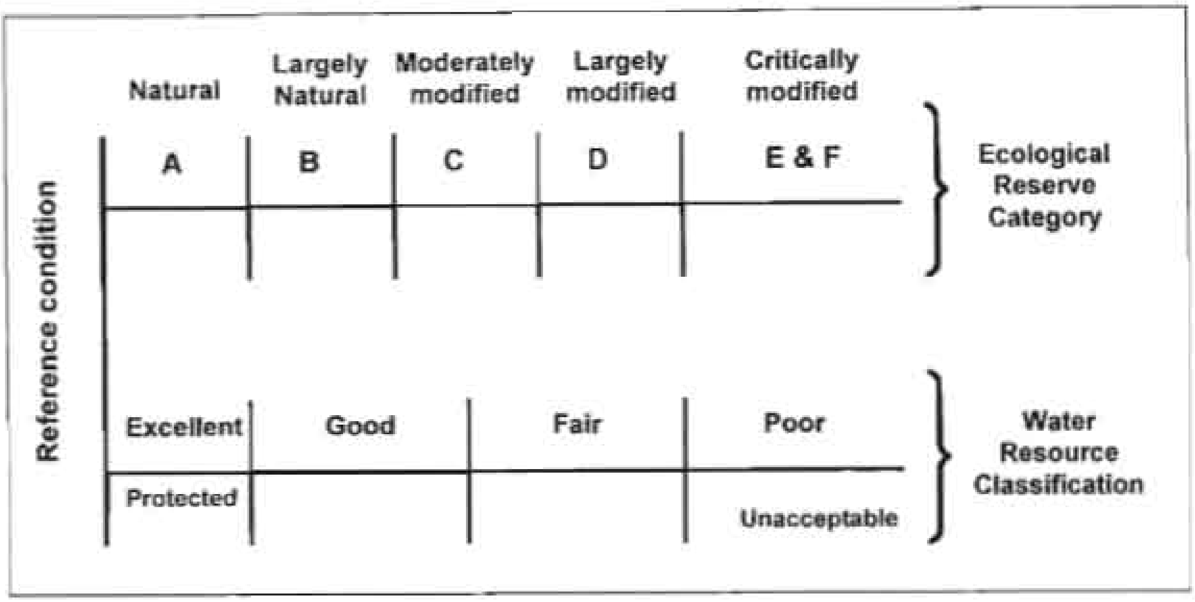

Figure 4.3 A diagram of the alternative classification systems currently in use in the Ecological Reserve determinations (after Hughes, 2005)

In general, an ERC A represents virtually unmodified, natural conditions (generally referred to as reference conditions) and ERC D represents a high degree of modification from natural conditions (King *et al.*, 2000). ERC D is used to describe the lowest level of ecological modification that is generally acceptable. Two further ERCs, E and F, are used to describe resources which are critically modified and generally unacceptable since loss of ecological functioning is extensive. In the worst cases of ERC F, ecological functioning is impaired beyond rehabilitation. Recommendations of ERCs E and F for the future state of river systems are undesirable. However, the recommended ERC must be attainable, even if only in the long-term, and ERCs of E, E/F and even F may be necessary when considering the degree of difficulty, or attainability, and associated measures required to mitigate the causes of degraded water resources.

3.2.2 Determination of Environmental Flow Requirements

ER determinations require quantification of the water resource under different flow-related scenarios so that stakeholders can select a "recommended scenario" (*c.f.* Section 3.2.4.) which meets the resource classification (*c.f.* Section 3.2.1). Thus, a major aim of an ER determination is to provide descriptive and quantitative information regarding the pattern and reliability of environmental flows, including information on frequency, magnitude, duration, timing and rate of change, so that a modified flow regime can be provided (King *et al.*, 2000). This biophysical assessment of the flow component of the ER should assess environmental flow requirements (EFRs) in order to manage the health of the river bank, groundwater, wetlands, floodplains and estuary in addition to the river channel, for a range of ecological protection levels (*c.f.* Section 3.2.1).

In South African water resources management, managing aquatic health and meeting EFRs for ecosystem functioning is strongly linked with the assessment of instream flow requirements (IFRs). While there are several definitions of IFRs in the literature, and many methods to characterise them, the definition of IFRs provided by MacKay and Moloi (2003) is adopted in this Chapter, *viz*:

(a) "the *quantity and quality of water*, expressed in terms of the magnitude, duration, timing, frequency and rate of change of specific flows, and

(b) the *quality of water*, expressed in terms of the range, frequency and duration of occurrence of concentrations of key water quality variables that are required to maintain a desired level of aquatic ecosystem health".

There is an increasing recognition in water resources management of the importance of the connectivity of the hydrological cycle and the role of Environmental Flow Assessment (EFA) as a management tool for the sustainable use of water resources. Notwithstanding the links among different components of water resources, (*i.e.* river channel / river bank / groundwater / wetland / floodplain / estuary) as well as the links among ecological, social and economic systems (*c.f.* Section 3.2.4), determining the flow requirements for South Africa's water resources *largely remains in the realms of assessing water quantity, streamflow patterns and assurance to meet the IFR*. However, since the term IFR in the South African context is used to define the water required to protect aquatic ecosystem form and function at some agreed level (the ERC), which in turn delivers a level of ecosystem goods and services, the link to human systems is implied. Thus the determination of the IFRs is perceived as being critical in protecting the wide range of benefits provided by the ecological functions of healthy aquatic ecosystems.

Protecting the health of an aquatic ecosystem requires that four components of the ecosystem be addressed, which can be summarised as follows (MacKay and Moloi, 2003):

(a) An adequate *volume of water* must be provided, which should be distributed in a pattern which reflects natural conditions.

(b) *Water quality* must be sufficient to *protect* the biota and their associated *ecological processes.*

(c) The *characteristics* of the aquatic habitat (form, extent and ecostatus) must be maintained to *provide sufficient habitat* of appropriate quality to sustain viable biotic populations.

(d) The *characteristics* of the biotic populations (composition, distribution and ecostatus) must be preserved to *maintain the ecological processes* that depend on these populations.

MacKay and Moloi (2003) highlight two primary approaches to the assessment of the IFR:

(a) assessment of the volume and quality of water that is left in a water resource (*i.e.* river system), at specific sites in the resource, to maintain a certain desired level of aquatic ecosystem functioning and health (*i.e.* "*how much water must be left in the ecosystem?*"), or

(b) assessment of the volume of water that may be abstracted from specific sites in a water resource and the permissible extent of change in water quality in the resource for aquatic ecosystem functioning and health to be maintained at a certain desired level (*i.e.* "*how much water can be taken out of the ecosystem?*").

Both approaches have different applications, advantages and disadvantages, the salient points of which can be summarised from the paper by MacKay and Moloi (2003) as provided in Table 4.3.

Differences in spatial scale and resolution among various components of the catchment and river system (from headwaters to estuary) have led to the determination of IFRs in South Africa being assessed at a river reach level, whereby catchments are segmented into resource units, with each river reach requiring separate specifications of its IFR (Tharme and King, 1998). The most useful IFR determination to water resources managers, particularly where releases are to made from impoundments to meet the quantity, quality, patterns and assurance of flows required for a desired level of ecosystem protection, need to describe ecologically relevant streamflow components relating to magnitude, timing, frequency, duration and the rate of change of flows as well as account for inter-annual and intra-annual variability of flows (MacKay and Moloi, 2003).

Table 4.3 Summary of two different approaches applied to determine the Instream Flow Requirements of South African River Systems (source information from MacKay and Moloi, 2003)

Attribute	**Approach to IFR Determination**	
	How much water must be left in the ecosystem?	**How much water can be taken out of the ecosystem?**
Application	• Water utilisation is already high or over utilised • System is regulated with major abstractions made from impoundments • Limits to utilisation: • Total abstraction = total virgin flow - IFR	• Impact of human activities is relatively insignificant • Little or no large-scale development has taken place • Systems display high level of natural variability in rainfall, runoff and water chemistry, (difficult to assess IFR)

Advantages	• Similar to Strategic Environmental Assessment (SEA) approach (*i.e.*
	  • water use and projects contained within limits, or
	  • water demand measures can be implemented)
		• Similar to Environmental Impact Assessment (EIA) approach (*i.e.*
		  • scenarios developed and tested and
		  • relative change is assessed and translated into limits)

Disadvantages	• If IFR is initially set too low, difficulties arise when attempting to:
	  • reduce already authorised water use or
	  • reverse impacts of over utilisation of water use
		• Unless regulation and auditing of water uses are effective, problems arise regarding:
		  • cumulative impacts of "individual" projects which can lead to
		  • gradual degradation of ecosystem condition

Example Method	The Building Block Method, BBM, (King and Tharme, 1994)	The Downstream Response to Flow Transformation, DRIFT (Brown and King, 2000)

The ecologically important flows are as follows.

(a) *Dry season low flows:* These maintain essential ecological processes as well as habitats and refugia for biota during the low-flow season.

(b) *Wet season low flows:* These maintain important wet season ecological processes and habitats (King *et al.*, 2000).

Together, dry and wet season low flows, or baseflow, conditions characterise periods of flow between high rainfall events. Baseflows define the quantity of water in the channel, the extent of wetted perimeter and influence habitat availability for instream biotic species as well as "the depth to saturated soils for riparian species" (Baron *et al.*, 2002).

(c) *Intra-annual floods or freshettes:* These higher flows act as environmental cues for biological processes such as spawning and migration. Flushing flows dilute poor quality water and break up small debris that accumulates during the dry season (King *et al.*, 2000). Moderately high flows connect the channel to the river bank and riparian terrace, thereby maintaining riparian vegetation.

(d) *Larger floods:* These maintain channel geomorphological characteristics, scour and transport accumulated sediment and reshape the channel (King *et al.*, 2000). Extreme floods inundate the floodplain and support biological productivity through nutrient exchange (Sparks, 1995).

(e) *Intra-annual variation,* or *seasonality,* of flows: These are essential for maintaining the different life cycle strategies of native species.

(f) *Inter-annual variation of flows:* These are critical for maintaining aquatic ecosystem biodiversity.

In the determination of the ER, IFRs should be set which address each of these streamflow components. The baseflow components (*i.e.* both dry season and wet season low flows) form a critical role in hydrological connectivity, linking the catchment landscape with surface aquatic ecosystems and groundwater systems. The high flow components (*i.e.* annual flood pulses and the larger inter-annual floods) form an important role in connecting aquatic ecosystems with terrestrial ecosystems. Determinations of IFRs which account for these flow components are vital to maintain the dynamic characteristics of aquatic habitats.

There are currently three main methods of environmental flow requirement assessment in South African water resources management, *viz.*, the,

(a) Building Block Methodology (King and Louw, 1998)

(b) Downstream Response to Imposed Transformation (Brown and King, 2000)

(c) Flow-Stress Response Method (O'Keeffe *et al.*, 2002)

The basis and characteristics of these three methods are outlined briefly in the remainder of this sub-section.

The Building Block Methodology

The assessment of IFRs in South African water resources management has benefited from the application and experience derived from a well established biophysical assessment process known as the Building Block Methodology, BBM (King and Louw, 1998). The BBM is a "how much must be left in?" approach (*c.f.* Table 4.3), designed to construct a streamflow regime for maintaining a river in a predetermined condition (King *et al.*, 2000) in years of average, or maintenance, water conditions and in years where ecosystems are stressed, as they might be expected to experience under hydrological drought conditions. The IFR of either target species only, or of all the riverine ecosystem components (*i.e.* vegetation, invertebrates, fish, geomorphology, water quality and hydrology) of the aquatic ecosystem can be addressed and, as such, Tharme (2003) has appraised the BBM as being an holistic approach to the assessment of streamflows.

In general, the BBM addresses three "building blocks" of the modified regime:

(a) The first "block" addresses the magnitude of baseflows in wet and dry seasons and "defines the required perenniality or non-perenniality of the river, as well as the timing of wet and dry seasons" (King *et al.*, 2000).

(b) The second "block" addresses the intra-annual freshes in the wet season that are required to initiate scouring of the channel bed and cleansing of the river.

(c) The third "block" determines the magnitude, timing and duration of higher floods in the wet season (Tharme and King, 1998; King *et al.*, 2000).

These baseflows and essential higher flows are recommended for both maintenance years (*i.e.* years of "normal", climatic conditions, when the full suite of ecological functions and processes might be expected to occur) and drought periods (*i.e.* flows are designed to allow biotic survival and important ecological processes; King *et al.*, 2000).

Required flows are identified on a month by month basis, for both the baseflows and the higher flows, based on expert knowledge and experience of the hydraulic habitat preferences of different species. Many studies in hydraulic stream ecology have shown that simple measurements of average flow velocity, flow depth as well as bed and channel roughness can be correlated with the distribution of flow-dependent biota (O'Keeffe and Hughes, 2005). The biological implications of any flows are described in terms of depth, wetted perimeter, velocity or areas inundated, using habitat and hydraulic surveys and plots of hydraulic relationships (King *et al.*, 2000). Thus, the BBM, approach utilises the relationship between flow, hydraulic habitat and biotic response in its approach to environmental flow assessment and expert knowledge of species habitat preference is converted to streamflow values.

The output from early BBM workshops was an IFR table of the magnitude component of monthly requirements for the maintenance and drought flows. However, this output did not provide information on the frequency (or percentage of time) of droughts, how often flows should be between droughts and maintenance flows, and in wet years, how often flows should be above maintenance requirements (IWR, Environmental, 1999). More recently, the South African IFR process was further developed to include the frequency component in environmental flow requirement assessments, in order to establish assurance rules that combine the maintenance and drought requirements, based on a percentage exceedence of flows, on a month by month basis.

A basic assumption of the BBM is that the "supply" of the flows specified in the IFR table should be linked to the natural flow regime or to a natural climatic trigger. Specialists at a comprehensive BBM workshop define a set of low flow and flood "operating rules" for use with climatic cues. The operating rules represent threshold values, which are compared in a [Daily IFR Design, Hughes, 2005] model with daily values of the prevailing climatic cues (derived by examining the daily flows within a "reference flow" time series) to determine the likely day to day pattern of flow rates (IWR Environmental, 1999). Thereafter, "the operating rules are calibrated until an acceptable pattern of time series of modified flows is achieved" which meets the specialists' understanding of the effects of the modified flow regime on the river ecosystem functioning (IWR Environmental, 1999). The final (modelled) IFR output can be summarised statistically as the percentage of time that the modified flow regime is at, or above, maintenance, between maintenance and drought or at drought levels, for each calendar month. In essence, these statistics represent the recommended assurance levels of the different flows and can be illustrated by a flow duration curve (IWR Environmental, 1999), as shown in Figure 4.4. In this way, the output of a BBM IFR table is changed to the output of the Daily IFR design model to generate a time series of daily Reserve requirements.

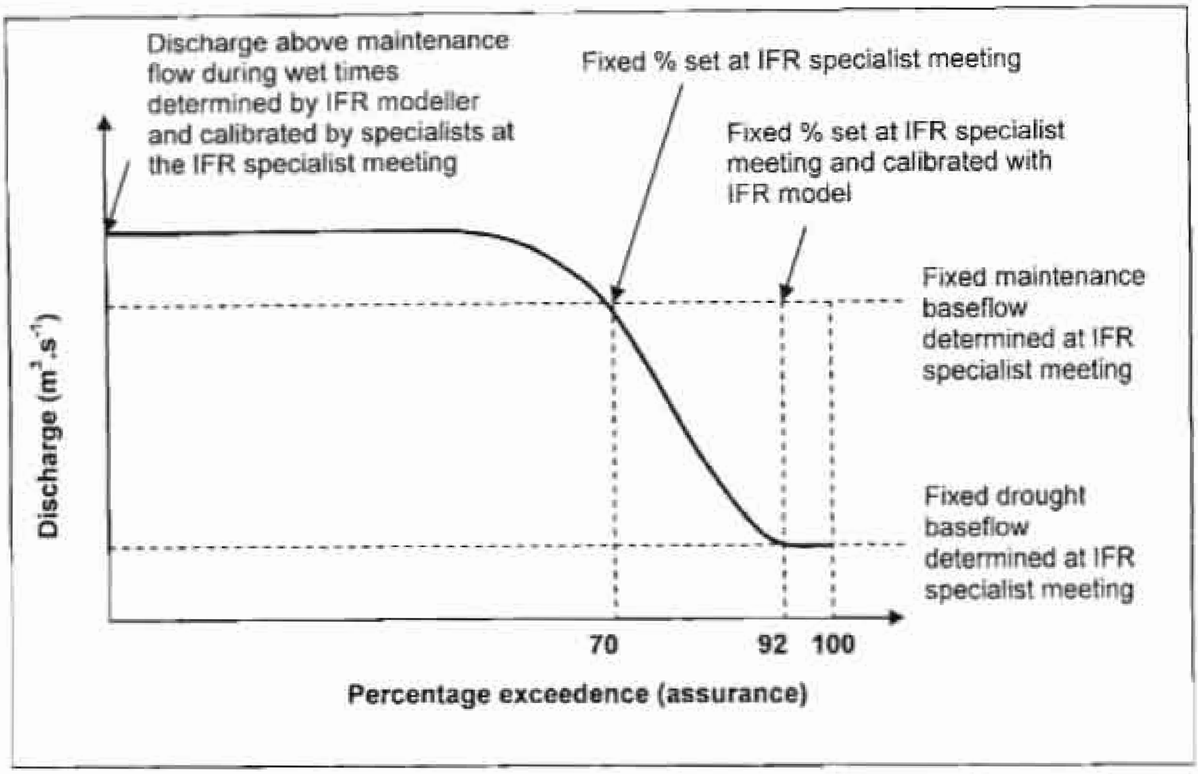

Figure 4.4 Output of an IFR specialist meeting illustrated as a Flow Duration Curve for any calendar month. NB all numbers on axis are hypothetical and differ from river to river (after IWR Environmental, 1999).

Alternatively, preliminary and relatively low confidence Ecological Reserve determinations have been conducted for each of South Africa's 1946 Quaternary Catchments for each of the different ERCs, A through D, and incorporated in a Desktop Reserve model (Hughes and Hannart, 2003). These nationwide ER determinations are based on the results of much more comprehensive IFR studies, performed for fewer river sites and involving groups of specialists applying the BBM. Reserve requirement assurance tables are generated by the Desktop Reserve model which includes parameters for 21 regionalised assurance curves. Examples of these curves, from the work of Hughes and Hannart (2003), are shown for two different climatic regions in Figure 4.5. Hughes and Hannart (2003) explain that "the regionalisation was based on the natural flow duration curve characteristics of the 1946 quaternary catchments in South Africa". The y-axis represents the combined baseflow and high flow requirement, represented here as a percentage of the maintenance requirement for January, whereas the x-axis represents the percentage assurance that the flows would be expected in the modified regime (Hughes and Hannart, 2003). The numerical methods and regional parameter values for the assurance curves can be found in Hughes and Munster (2000). However, it is evident from the illustration of Hughes and Hannart (2003), reproduced in Figure 4.5, that in drier regions, the maintenance flows are expected with lower levels of frequency (*e.g.* 27% assurance) than the less variable flow regimes in wetter regions (*e.g.* 60% assurance), whereas flows considerably greater than maintenance are required infrequently (*e.g.* 170% of the maintenance requirement at between 0 and 10% assurance in Figure 4.5; Hughes and Hannart, 2003).

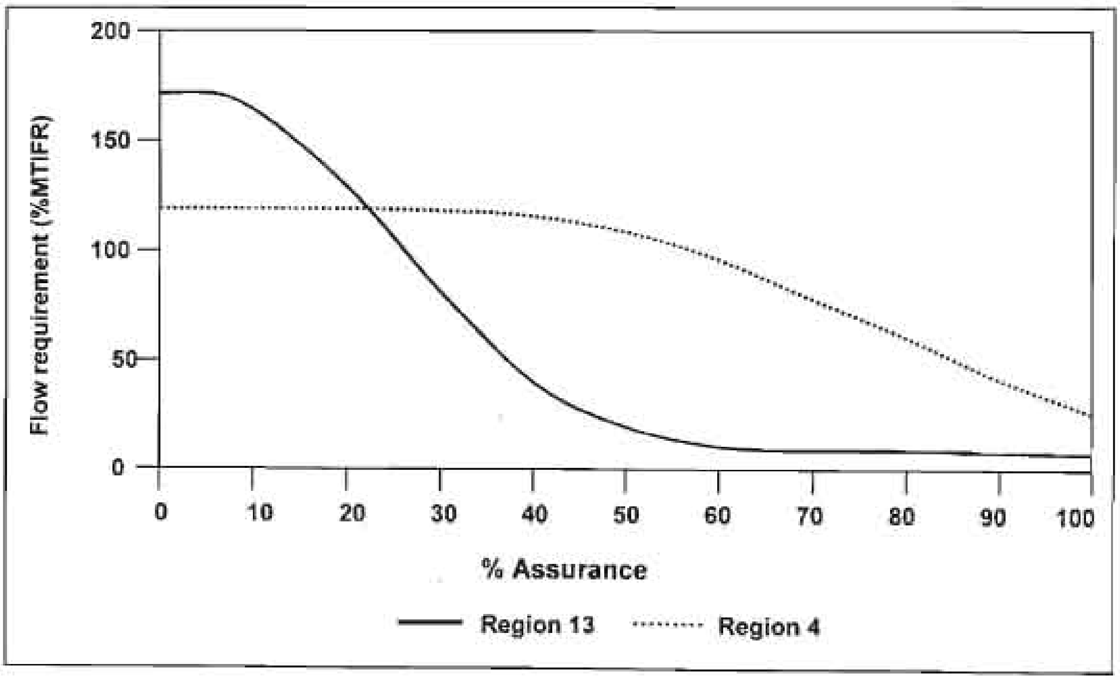

Figure 4.5 Examples of generic assurance rule curves for the month of January for a wet region in the area of the Drakensberg Mountains, Region 13 and a dry region in the Eastern Karoo, Region 4 (after Hughes and Hannart, 2003)

Thus, the final output from a BBM based Reserve determination is a table of flows for each calendar month for a set of assurance percentages, which are considered by the ecological specialists as being likely to meet predetermined ecological objectives (*c.f.* Section 3.2.1 of this Chapter) (Hughes, 2005).

In current ER Determinations, regardless of whether the Daily IFR model or the Desktop Reserve model is used to generate a time series of Reserve requirements, several determinations are undertaken in order that the DWAF is provided with a range of options and consequences from which to select the one which best meets the overall resource quality objectives for the water resource (Hughes, 2005). Thus, an interactive process is initiated to develop alternative IFR, or flow-related, scenarios (*c.f.* Section 3.2.4).

The Downstream Response to Imposed Transformations

In a separate southern African development, the Downstream Response to Imposed Transformations, DRIFT (Brown and King, 2000; *c.f.* Chapter 3, Section 2.5), arose as a scenario-based assessment of the socio-economic impacts of progressive reductions in river discharge from reference conditions on the biophysical functioning of the resource (Tharme, 2003). Thus, the DRIFT method, a "how much can be taken out" approach (*c.f.* Table 4.3), considers streamflow responses to different water-management activities (Brown and King, 2002) and has received increasing recognition in the South African EFAs. Overall, DRIFT comprises four modules (biophysical, societal use, scenario development and compensation economics). It is not necessary to apply all the modules in an EFA (Brown *et al.*, 2006) and often only the biophysical and scenario development modules are used. In DRIFT, the biophysical assessment is approached in a workshop forum by separating the present-day long term daily flow data for each site of interest into a number of flow classes which address both low flows and floods. Thereafter, specialists predict the impacts of several levels of change from present conditions in each flow class for different biophysical components of the river system (Brown *et al.*, 2006). In this way a database of information relating to the flow, hydraulic habitat and biotic response relationship is assimilated for a range of possible flow changes, which can be used to examine management-related issues and to provide a range of scenarios linking flow and ecological condition. Each consequence of change is accompanied by an integrity rating, which, together with the database, is used to create summary plots of the predicted results using two MS Excel-based programmes, DRIFT SOLVER and DRIFT CATEGORY (Brown and Joubert, 2003). DRIFT SOLVER "comprises an integer linear programming multicriteria analysis method, which generates optimally distributed flow regime scenarios for different total annual volumes of water", whereas DRIFT CATEGORY "facilitates evaluation of these flow regime scenarios in terms of river condition" (Brown *et al.*, 2006).

The integrity ratings for each ecosystem component are combined for each level of flow change in wet season low flows, dry season low flows and the different class floods. Thereafter, flow regime scenarios for different total annual volumes of water are created by plotting the DRIFT integrity score against the volume of water associated with each change level investigated (Brown and Joubert, 2003). The optimal flow regime scenario is calculated from the maximised DRIFT scores for different annual water volumes, distributed over the year in such a way as to provide most benefit to the river ecosystem, given the limits of the flow changes assessed (Brown *et al.*, 2006).

Similarly to the BBM, the output of the DRIFT process of environmental flow assessment is an annual summary of the magnitude component of a recommended flow regime for maintaining a predetermined ecological condition, although in the DRIFT process volumes are specified for every year rather than for maintenance years and drought years. Nonetheless, as in the BBM, these flow volumes, presented in a table with flow rates for low flows and flood flows for different calendar months represent an *estimate* of the flows required and the actual volumes depend on the climate (Brown *et al.*, 2006). Thus, in an ER determination, the information provided in a DRIFT output table is supported with IFR assurance rules, based on exceedence data, which facilitates the operationalisation of the ER (based on climatic cues) and also the incorporation of the ER into the DWAF Water Resources Yield Model, WRYM, for evaluation of the ecological, social and economic implications of a range of flow-related scenarios (*c.f.* Section 3.2.4), as represented by the different ERCs (Brown *et al.*, 2006).

The Flow-Stress Response Method

A relatively recent development in EFA for South African water resources management is the application of the Flow-Stress Response (FS-R) method (O'Keeffe *et al.*, 2002). The FS-R was developed "as a tool to guide the evaluation of the ecological consequences of modified *low* flow regimes, based on the principles of ecological risk, using an index of flow-related stress" (O'Keeffe and Hughes, 2005). The FS-R is still in the course of development, and while a suitable approach to applying the principles of the flow-stress relationship (see below) to high flows has been considered, to date the method has been applied only to the response of instream biota at low flows. At the time of writing (2006), there is very little published material from which to review the FS-R. Consequently, the following review is sourced mainly from the material presented in O'Keeffe and Hughes (2005).

Similarly to the BBM and DRIFT, the FS-R arose in recognition of the relationship between flow, hydraulic habitat and the response of instream biota. The founding principle of the FS-R is that, given that species' habitat preference can be characterised and quantified (*c.f.* Page 4-20), it is, in turn, possible to predict the effects of habitat loss, as a result of an altered flow regime, on the distribution of flow-dependent biota. It is further assumed in the FS-R that with progressive reductions in low flows, habitat suitability for flow dependent communities is also reduced which, in turn, results in reductions in the biotic abundance. Thus, similarly to the BBM and DRIFT, the FS-R relies on specialist knowledge and information regarding the hydraulic habitat preferences of a variety of flow-related species. This knowledge of the relationship between flow, hydraulic habitat and instream biotic response is used to construct an index of increasing stress, in accordance with a generic table, from which a site-specific flow-stressor response relationship can be established. Thus, the "flow-stress" relationship describes the hydraulic preference of any particular species or community and a stress curve can be constructed to describe the relationship between changing flow and stress for any flow regime (present day, natural or any other selected scenario).

Whether the assessment relates to flow requirements or to flow-stress levels, a main concern regarding EFAs, is that the biophysical assessment of individual ecosystem components can be perceived to be separatist rather than holistic. The developers of the FS-R have approached this anomaly by adopting a "best estimate of ecosystem requirements". This is achieved in the FS-R through identification of a critical stress curve representing a combined flow-stress relationship and which describes the highest stress level among all the ecosystem components (of vegetation, invertebrates and fish) for any given flow.

The FS-R was designed for use *within* methodologies such as the BBM and DRIFT as a means of reliably capturing specialist knowledge regarding the flow-response relationship. However, the method can be used in the development of flow-related scenarios. In keeping with any other EFA, the ecologists involved in an FS-R assessment are required to identify objectives or conditions which will result in their component being associated with one or more ERCs, from A to D. Consequently, the resource objectives must be at a sufficient level of detail to be related to stress levels, durations and frequencies. The critical stress curve forming the flow-stressor response relationship is applied to transform the natural, present day or any other scenario flow time series into a stress regime which can be analysed in terms of the magnitude, frequency and duration of stress levels experienced by specific biota for any scenario. A stress duration curve can then be constructed for any flow regime scenario. Thereafter the stresses specified in the objectives are applied to the curves to identify the most appropriate ERC.

The FS-R has not been applied extensively, or even for the full range of ecologically relevant streamflow components, and it may be more useful in assessing sensitive or vulnerable river systems where ecosystem "stress" is a more frequently occurring natural phenomenon (*i.e.* as in ephemeral systems) than in more permanent, or perennial systems. This does not detract from its use in ER determinations. For example, in the comprehensive assessment of the ER for the Thukela Catchment in KwaZulu-Natal, the FS-R was applied to assess the low flow component of the IFRs for each of the (22) RUs identified, whereas a method adjusted from the more traditional BBM and from the DRIFT method was applied to assess the high flow component (IWR Environmental, 2003). Consequently, after the low flows and high flows have been incorporated into an integrated flow regime, the final output of a "partial FS-R approach" to an ER determination is that IFR rules, presented as duration tables, are provided from either the Daily IFR Design Model or the Desktop Reserve Model.

3.2.3 Summary of the assessment of the flow requirements of the Ecological Reserve

Figure 4.6 summarises the main attributes of assessing the requirements of the ER. While the ecological state, or ERC, of an Resource Unit (RU) is not solely streamflow related, it is recognised that the streamflow regime drives variation in many other ecological components and that the natural variability of the hydrological condition is the key component in ER determinations. In order to reflect ecological biodiversity, ER determinations focus on the natural intra-annual and inter-annual variation within the streamflow regime. Thus, the maintenance conditions referred to in Figure 4.6 relate to streamflow patterns that could be expected in years of average flow conditions, whereas the drought or stress conditions refer to streamflow patterns that could be expected to disturb the ecological functioning of the resource in times of water scarcity. IFR assurance rules are required to make a frequency link between maintenance and drought conditions. Management objectives which focus on maintaining or enhancing the ERC of a RU can be either streamflow or non-streamflow related. However, the impact of providing increased ecological assurance together with increased habitat integrity is that the quantity and quality of water required for the ER increases.

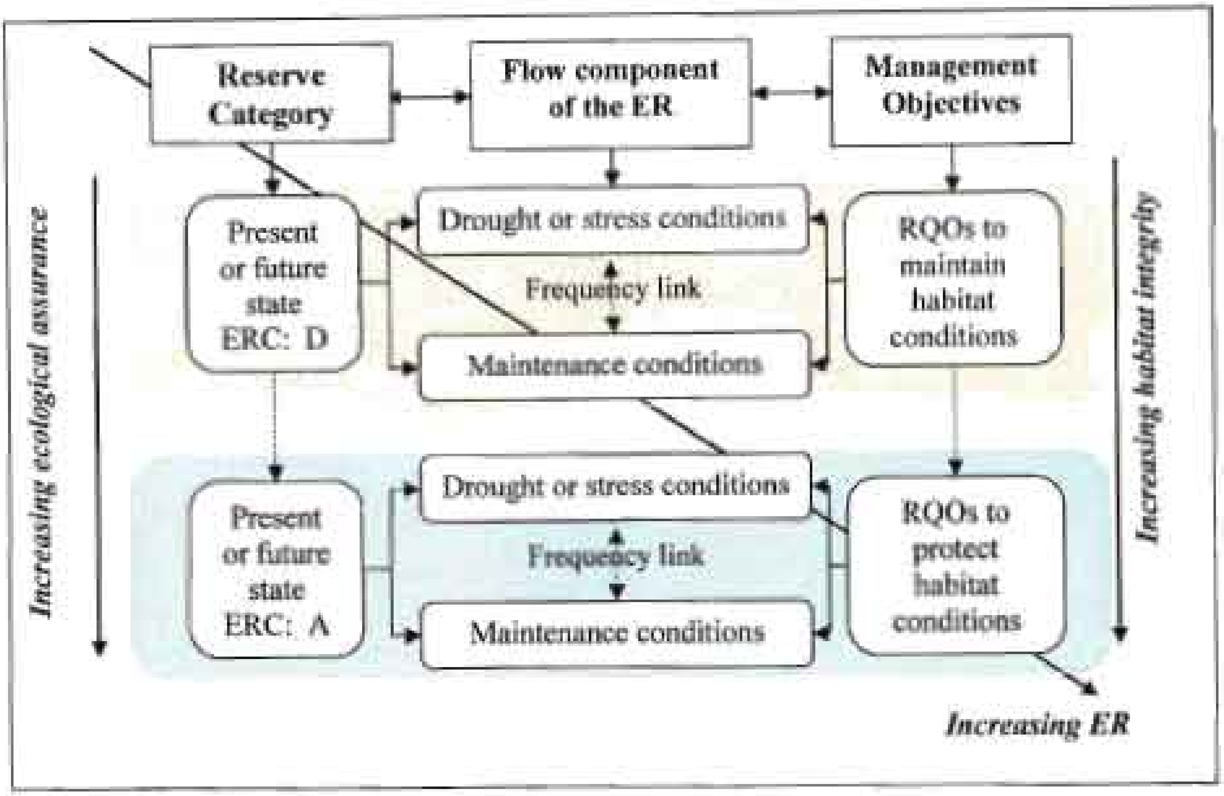

Figure 4.6 Summary of the assessment of the flow requirements of the Ecological Reserve

3.2.4 Flow-related scenarios

In the most advanced ER determinations, the ecological impact and associated societal and economic impacts of different flow regime scenarios are evaluated in order to provide decision-makers with a full array of the associated options and consequences. Essentially this requires assessment of the reference and present day flow regimes as well as any other recommended or alternative flow regime scenarios on present and future water users and of the impacts of future water users on the ER.

In current Reserve Determinations, the time series output from either the Daily IFR model or the Desktop Reserve model, together with a set of operating rules (*c.f.* Section 3.2.2) for Reserve releases from reservoirs, can be used as input to either a daily or monthly rainfall-runoff model to determine the existing, natural or future allocable yield (Hughes, 2005).

For relatively uncomplicated systems, where each reservoir is operated independently, the allocable yield is assessed through iterations of rainfall-runoff model runs with different abstractions, dam size or operating rule scenarios until a specified allocation or demand is achieved at the required assurance, which, in turn, is determined through graphical comparison of the time series of required draft with the achieved draft, or by examining the time series of reservoir level (Hughes, 2005). For multiple reservoir systems, where more comprehensive operating rules and priorities for various abstractions are required, it is necessary to apply a water resources system model such as the WRYM to assess allocable yield. The WRYM was developed to determine whether planned water resource developments can meet expected abstractions as well as the IFR release requirement, with or without the IFR accommodated as a priority (IWR Environmental, 1999). The WRYM uses input data at the monthly time step for a set of percentage exceedence flows. Consequently, the time series output from the Daily IFR requires some manipulation to generate flow duration curve data prior to application (Hughes, 2005). The output from the WRYM is typically a time series of monthly flows at different points within the reservoir system for a range of scenario options. These time series can be converted for comparison with various stress curves generated by the FS-R model (Hughes, 2005).

Where no reservoir storage exists, the Reserve Licensing Model (RML) can be used to assess the impacts of the ER on present water users or, alternatively, the impact of additional water uses on the ER (Hughes, 2005). However, the water uses in the RML are restricted to those relating to afforestation, abstractions from small distributed dams and run-of-river abstractions. Consequently, many other streamflow altering activities are overlooked. Again, the RML scenarios are preformed at a monthly time-step, with the natural flow monthly time series typically being provided by the WR90 database (Midgley *et al.*, 1994) or generated by the monthly Pitman model (Pitman, 1973). In addition, the flow-related scenarios require a table of Reserve assurance rules from the Desktop Reserve model as well as some information regarding present day and future water use. The resultant evaluations are compared with the Reserve requirements using graphical comparisons of the flow duration curves (Hughes, 2005).

In the most advanced Reserve determination studies, the ecological impact and associated societal and economic impacts of different flow scenarios are evaluated with the objective that decision makers are equipped with a complete set of information and comparable results (DWAF, 2003). The optimal scenario balances ecological sustainability with user requirements.

Thus, while the approach to the actual environmental flow assessment of DRIFT differs to that of the BBM and the FS-R, all these methods result in flow-related scenario assessment. As the RDM processes evolve, the determination of the ER for South Africa's water resources has incorporated the benefits of the different approaches (*c.f.* Table 4.2) to EFA. In addition to combining different methods of flow assessment, different quality assurance techniques are being combined, even across different levels of confidence. For example, the IFRs for different ERC scenarios derived from the Desktop Reserve Model (generally low confidence level Reserve estimates, extrapolated from earlier comprehensive determinations) were applied in the Thukela Reserve Determination to generate low flow stress curves using the FS-R method for comparison with natural stress profiles and present-day stress profiles. Moreover, the Desktop Reserve Model was further applied in the final IFR rules (combination of low flow and high flow requirements) in the production of IFR assurance rules. Thus the evolution of Reserve determinations has benefited from the practicalities of combining different approaches and levels of confidence in the methods.

Notwithstanding the different approaches to the determination of the ER for South Africa's water resources, there is a common need for useful tools to assist in the EFAs. Clearly credible hydrological models are indispensable for anticipating the impacts of various ERC scenarios on present and future water users. Most of the facilities required to facilitate the ER assessments described in this Section have been incorporated in an integrating framework, known as SPATSIM, which was developed for ER determination and implementation (Hughes, 2005).

3.3 Complexities to the Determination of the Ecological Reserve

As a result of experience in assessing the IFRs of several of South Africa's key river systems (*e.g.* Thukela in 1995 and 2002; Marite, Sabie and Luvuvhu in 1996; Mkomazi in 1998; and Mhlatuze in 2000), determining the ER initially focused on the ecological requirements of the surface streamflow component of water resources. However, determining the ER requires assessment of the relationship of the main water resource components, *i.e.* among groundwater and surface water bodies, comprising rivers, wetlands, lakes and estuaries.

Discussions at a RDM workshop held in 2001 to review the progress of the DWAF's RDM Revision Project on consolidating the Reserve Methodology raised a number of salient points (DWAF, 2001) which are still challenges to the RDM process, *viz*:

(a) Many of the RDM concepts (and consequently the assessment of the flow requirements of the ER as well as the RQOs) are founded on experience on assessing surface streamflows, notably the IFR processes, which are not always appropriate to the other components of the water resource.

(b) The different spatial scales of the different water resource components, particularly the three-dimensional attributes of groundwater and the intermittently discontinuous attributes of wetlands, need to be addressed in the Reserve determination methodologies.

(c) A Reserve determination method in which wetlands are conceptualised as a water balance between direct precipitation, surface water, groundwater and evaporation would be more appropriate since wetlands represent essentially "circular" rather than "linear" systems.

(d) The inter-connectivity of ecological functioning of the different components of water resources requires attention in the available methodologies.

(e) An estuary may be ecologically or economically sensitive, yet with current methodologies an upstream IFR for, say, a degraded section of the river could have been set for a lower ecological category. There is a need for better matching of flow requirements between rivers and estuaries.

(f) There is a need for greater understanding of the relationships between groundwater and other water resource components, particularly the processes of groundwater discharge to surface streamflows, and the dependency of terrestrial ecosystems on groundwater. Baseflow-dependent ecosystems are important for biodiversity, providing a habitat niche for a variety of organisms, *e.g.* riparian ecosystems play important roles in maintaining the biodiversity and functioning of the adjoining terrestrial ecosystems in the Sabie-Sand River system in the Kruger National Park (Jewitt *et al.*, 1998).

(g) The surface water Reserve is spatially referenced; it may be more appropriate to use spatially referenced water levels (within buffer zones or areas) rather than volumetric or gradient units to express the groundwater Reserve.

(h) Accounting for temporal variability of the groundwater Reserve is problematic as a result of the complexities associated with the determination of the contribution of groundwater storage to surface water IFRs.

3.4 The Groundwater Link

The determination of the ER entails examination of the relationship among the major interlinked components of the water cycle, *viz.*, surface water and groundwater (Smakhtin, 2001). For the sake of compatibility, it is important that parallel processes to those described in Section 3.2 of this Chapter will apply to different components of aquatic resources. For example, determining the ER for the groundwater component should incorporate understanding of the following (Xu *et al.*, 2002):

(a) the hydrogeological characteristics (groundwater response zone),

(b) current and potential future uses of groundwater,

(c) potential hazards to groundwater use (ecological vulnerability; contamination; drought),

(d) importance of groundwater use to stakeholders (social and economic importance),

(e) the level of protection for that function desired by stakeholders (RDM class),

(f) measurable attributes of the resource that maintain its desired functioning (RQOs), and

(g) the area where these attributes need to be managed to safeguard that function (management units).

As a result of the high ecological and social significance of aquifers nationwide, it is generally accepted that groundwater plays a major role in both components (ER and BHNR) of the Reserve. This is summarised as follows by Xu *et al.* (2002):

(a) Where there is a direct connection between groundwater and surface water processes, the role of groundwater in terms of water level, volume and water quality in supporting the aquatic ecosystem and human populations (both the ER and BHNR) needs to be ascertained.

(b) Where aquifers have limited connection with aquatic ecosystems, the groundwater component of the Reserve may constitute the Basic Human Needs component for those people using a local groundwater supply.

In catchments where groundwater is strongly linked to surface water, the impacts of supplementing the BHNR with groundwater resources could impact on the ecological functioning of surface water resources (Pollard *et al.*, 2002). Groundwater resources recover slowly from over-utilisation, and their vulnerability in terms of (a) ecological dependence on the wider environment, (b) contamination from land-based activities and (c) drought imply that they may require different RDMs (system of resource classification, Reserve determination and RQOs) to surface water resources (Xu *et al.*, 2002). Moreover, RQOs for the groundwater component of the Reserve are viewed as integral to South African Water Resources Management since they are not restricted to aquatic ecosystems and therefore highlight the role of aquifers to the wider environment (Xu *et al.*, 2002).

Research towards the RDMs of the groundwater component of the Reserve recommends that a generic approach, based on the *functionality* of the aquifer resource (*i.e.* as a sink or source for water; as a sink or source for nutrients; habitats), measurable *objectives* (RQOs based on the risk of impacting on the functionality) and key *indicators* (*e.g.* hydraulic gradients; hydrochemical conditions and biotic indicators) are selected to ensure that aquifers are fulfilling their functions in the environment (Xu *et al.*, 2002). While this approach is similar to that proposed for the surface water component of the Reserve, some of the RDMs for the groundwater component of the Reserve will be inherently unique, DWAF, 2004). The consequence of these differences, together with the differences in spatial resolution (*e.g.* where areas of aquifer recharge and / or discharge are located in different surface water RUs) and temporal resolution (*e.g.* the runoff responses may be relatively "quick" for surface water, but the transmission responses may be much "slower" for groundwater, where aquifer storage has been over-utilised) will further complicate the determination of the Reserve. Moreover, while surface streamflow is naturally variable, groundwater flows are much less variable, yet adding a dimension of temporal variability has been viewed as being imperative to the determination of the groundwater Reserve (Smakhtin, 2001), particularly where surface water and groundwater are interlinked.

Quantification of the groundwater component of the Ecological Reserve

While a framework for the quantification of the groundwater component of a comprehensive Ecological Reserve was devised by Braune *et al.* (1999), Smakhtin (2001) recommends that the *geohydrological time series of baseflow, groundwater storage* and *recharge* should form part of any proposed methodology. Smakhtin (2001) developed a simple baseflow separation technique for estimating continuous monthly baseflow time series from streamflow records which are widely available in South Africa. This technique was subsequently revised by Hughes *et al.* (2003) for time series of both daily and monthly streamflow values and applied in the Desktop Reserve Model referred to in Section 3.2 of this Chapter. Smakhtin (2001) further proposes that the result of applying this technique, whether to simulated streamflow time series (*e.g.* WR90 monthly time series of natural, or reference conditions, as available from Midgely *et al.* (1994), to DWAF observed monthly streamflow data or to an IFR time series, is a *baseflow time series* which "may be considered to be the outflow from groundwater storage". However, this definition of baseflow is contentious, since Xu *et al.* (2002) as well as Hughes *et al.* (2003) stipulate that baseflow derived by separating the direct runoff from the hydrograph is not indicative of any specific streamflow generation mechanism or origins of water sources. For example, Xu *et al.* (2002) contend that baseflow derived in this way "may still contain some interflow component". Moreover, Smakhtin (2001) states that continuous baseflow separation techniques "are not always appropriate for the identification of the origin of baseflow". Notwithstanding this discrepancy, Smakhtin (2001) suggests that the baseflow time series may be applied to derive a *groundwater storage time series* from the baseflow for each month using a simple "linear reservoir" approach, which, in turn, assumes that groundwater storage is proportional to baseflow with a retention constant estimated on a regional basis from the low flow studies of Smakhtin and Watkins (1997). Alternatively, a multi-storage reservoir model may be necessary if distinction among the different types of baseflow and the distribution of the subsurface storage is required, thereby adding complexity to this particular methodology of a time-series based groundwater component of the Reserve (Smakhtin, 2001). Smakhtin (2001) suggests that a low confidence estimate of *groundwater recharge* could be based on the difference between existing storage and that remaining from the previous month in the time series; although he does concede that alternative approaches may be required. Using Smakhtin's baseflow separation algorithm (Smakhtin, 2001 and see Chapter 5, Section 2.4.3), in conjunction with the WR90 simulated streamflow values, the geohydrological time series of baseflow and groundwater storage represents natural, or reference, conditions for quantification of the groundwater Reserve (Smakhtin, 2001).

Quantification of groundwater contributions to the Reserve and, in particular, to the river Reserve, should be matched with the IFR time series (Xu *et al.*, 2002). However, Xu *et al.* (2002) consider that it is difficult to regulate the groundwater contribution to meet IFRs on a "month-by-month" basis as they often are for surface water (see description of the BBM table on Page 4-23). Nonetheless, according to Smakhtin (2001), when applying the geohydrological monthly time series to the monthly IFR (surface water) time series (see Section 3.2.2 of this Chapter), the resultant groundwater storage time series represents the groundwater component of the Reserve. Where surface water and groundwater are connected, the groundwater and surface water Reserve time series are inter-dependent. In such instances, the surface water Reserve has a direct impact on groundwater resources, since IFRs determine the groundwater storages which are essential to maintain them (Smakhtin, 2001). Thus, Smakhtin (2001) refers to these groundwater storages as being "IFR driven". According to Smakhtin (2001), the time series of the differences between reference or present day storages and the IFR driven storages represents a time series of allocable groundwater. This time series could be used to derive the groundwater yield in a similar approach to the assessment of allocable surface water using a WRYM, whereas the groundwater storage time series could be applied to assess the impacts of meeting the ER on groundwater yield (Smakhtin, 2001).

In general, methods for determining the groundwater component of the Reserve require further research: However, alternatives to the techniques employed by Smakhtin (2001) to generate groundwater discharge time series can be found in the literature. While Xu *et al.* (2002) suggest the Smakhtin (2001) method for hydrograph separation for low to medium confidence ER determinations, they recommend a chemical separation method and algorithm such as that presented by Freeze and Cherry (1979), which uses data relating to chemical concentrations for direct runoff, groundwater and surface water as being suitable for medium to high confidence ER determinations. Moreover, Xu *et al.* (2002) suggest a hydro-geomorphological approach to the quantification of the groundwater contribution to surface water which incorporates qualitative knowledge of the nature of the surface water / groundwater interaction (*e.g.* influent; intermittent; interflow stream) as well as the geomorphology of the stream (*i.e.* interactive type) to be used in combination with hydrograph analysis.

Despite any proposed methods for the determination of the groundwater component of the ER, a major obstacle is that the interaction of groundwater with surface water bodies in South Africa is poorly understood and the links between the surface water Reserve and the groundwater Reserve also need further research. However, Xu *et al.* (2002) suggest that, where groundwater "leakage" or discharge to aquatic ecosystems or surface water bodies occurs, such discharge should be treated as the groundwater component of the ER and as such should not be greatly modified by allocation for other uses. While this may be considered to be conservative, where there is interaction between surface water and groundwater and where the management emphasis is to protect ecosystem functioning from human alterations, it does provide a simple solution to the determination of the groundwater component of the ER.

3.6 Summary

The ER is intended to protect the ecological functioning of water resources so that societal use of water and any potential water resource developments are carried out in an ecologically sustainable manner. The most highly developed methods for determining the ER relate to surface water resources, although the methods should be generic for all water resources. Determining the ER requires knowledge of the natural characteristics of hydrological regimes, including the magnitude, duration, timing, frequency and rate of change of streamflows, water quality, aquatic habitat and biotic characteristics. Current methods of the ER determination process include (a) resource classification, (b) the biophysical assessment of environmental flow requirements and (c) flow-related scenarios. There are three main approaches to the biophysical assessment of the ER which culminate in scenario assessment. Complexities to the determination of the ER emanate from need to integrate the major water resource components, taking cognisance of the relationships among groundwater and surface water bodies, comprising rivers, wetlands, lakes and estuaries.

4 DETERMINING THE BASIC HUMAN NEEDS RESERVE

4.1 Introduction

The priority afforded to domestic water for basic human needs and the assurance of supply for the Reserve, which includes the Basic Human Needs Reserve, *together with* the re-allocation of catchment water through licensing (Figure 4.2) is anticipated to redress some of the issues of poverty, alienation from natural resources and environmental degradation in rural areas (Pollard *et al.*, 2002). While the processes to determine the ER are well developed, less attention has focused on determining the BHNR despite the priority given to this "human right" in the National Water Act. In simplest terms, the BHNR is obtained by the product of the population and 25 litres per person per day, according to the Water Service Act of 1997 (WSA, 1997). Notwithstanding the simple concept of the BHNR, there are many challenges to meeting and delivering this Reserve, at both temporal and spatial resolutions.

4.2 Water for Basic Human Needs

The NWA (NWA, 1998) defines the BHNR as "the essential needs of individuals served by a water resource and includes water for drinking, for food preparation and for personal hygiene", but does not specify a quantity or assurance level of delivery. The Water Services Act (WSA, 1997) outlines the Reconstruction and Development Programme's (RDP) minimum service level to which people have a right as being 25 litres of water of acceptable quality per person per day, within 200 metres of their homes (WSA, 1997). Consequently, this quantity is used in water balance and allocation assessments and in water policy contexts as being representative of a short-term goal for the BHNR. A number of studies performed in Southern Africa to ascertain the daily water consumption of rural communities (*e.g.* Meigh, 2000; Peres de Mendiguren and Mabelane, 2001) demonstrate that the RDP recommended daily minimum of 25 litres may be adequate water for basic human life sustaining functions, since many communities survive on less.

Nonetheless, access to water to sustain livelihoods can also be construed as a basic human need in social systems, particularly in rural areas where communities depend on water for subsistence agriculture and small-scale enterprise. Yet, as formulated in the provisions of the NWA (NWA, 1998), any water required for additional household use, subsistence agriculture or small-scale productive use has to be met from the allocable catchment water resources (Pollard *et al.*, 2002).

4.3 Initial Models for the Basic Human Needs Reserve

In contrast to the sophistication of the methods and tools available to determine the ER, the processes to determine the BHNR are still in their infancy and there have been few attempts to meaningfully express the BHNR other than the simple estimation of volumes of water consumed per month on the basis of the RDP minimum recommendation of 25 litres per person per day. However, this volume has been translated by DWAF into the 6 kilolitres of "Free Basic Water" delivered each month to those households that are connected to a domestic bulk water supply system.

Nonetheless, this quantity of water is not strictly in keeping with the thrust of the NWA, which advocates sustainability and equity as central guiding principles (NWA, 1998). Moreover, the priority of basic human needs can be anticipated to intensify as the expectations for "essential needs" increase in the longer term *and* as the population being supplied with domestic bulk water increases (DWAF, 2001). In the meantime, the ER is determined according to the ERC (*c.f.* Section 3.2 of this Chapter) recommended to sustain a desired ecostatus, or level of aquatic ecosystem functioning, determined by the stakeholders of the water resource. While the ER (both in quality and quantity) may vary according to the ERC, there is no quantitative variation attached to the BHNR (Pollard *et al.*, 2002). In addition, while the ER follows natural intra- and inter-annual variability in climatic and streamflow conditions across a diversity of streamflow regime types, the BHNR currently does not possess any temporal (*i.e.* changes within or between years) or spatial (*i.e.* dependent on geographical location) variability. While differences in the quantification of the BHNR due to population changes are relatively easy to account for in water resources system models, it has been suggested that determination of the BHNR should account for temporal and spatial variability (Pollard *et al.*, 2002). Humans generally require greater quantities of water for survival in warmer than cooler climatic conditions, and greater assurance of supply in areas where there is within and between year variability. The source of the BHNR and point of delivery can be far removed and spatial variability of the BHNR refers to the need to account for service delivery losses from the source to the point of use. Moreover, the quantity of water has not been stipulated in the Act and many specialists have suggested quantities of water greater than the initial target of 25 litres per person per day. Thus, while 25 litres per person per day is perceived as a first target for the quantification of the BHNR, second and third targets of 60 and 100 litres have been discussed (Huggins, 2001, pers. comm.).

There is little literature relating to "pilot" or "preliminary" determinations of the BHNR for any of South Africa's Water Management Areas. However, assessment of the quantitative component of the BHNR has been included in the determination of the Reserve for the Thukela Catchment in KwaZulu-Natal by IWR Environmental (Pollard *et al.*, 2001), even though this determination focused mainly on the ER. Although the human population is widely distributed throughout the Thukela Catchment, the BHNR was calculated at the same spatial scale of the ER RUs, which focus on surface water streamflows. The spatial discrepancies between the delineation of the RUs and the distribution of societal communities were partially tackled by accounting for the population living within 5 km of the Thukela "run-of-river" for each Resource Unit. Larger settlements that are connected to a water supply scheme were also included in the model. In this preliminary BHNR model, the calculation of water for basic human needs within the downstream RU (at present or predicted consumption rates) was added to the quantification of the ER determined at each IFR site. However, linking the BHNR to the ER in this way raises a number of issues, which include the following:

(a) The Thukela ER is determined for instream ecological functioning (quantified as the IFR) whereas the BHNR is determined for off-stream human consumption. Releasing a plug of water downstream, *in addition to* the IFR is unlikely to result in the range of flows designed to maintain the diversity of ecosystem functioning required by stakeholders (Pollard *et al.*, 2002).

(b) More than half the present Thukela population abstract water for daily needs from boreholes, streams and springs beyond the 5 km buffer strip and upstream of many of the IFR sites. These abstractions and their impacts (relating to water quality and water level) on the ER (surface water and groundwater) have still to be addressed.

4.4 A Proposed System for Assessing the Basic Human Needs Reserve

The concept of the BHNR, as part of the Reserve, encompasses "the *quality*, as well as the *quantity* and *assurance*, of water required for the protection of basic human needs". The importance of these three characteristics for the BHNR can be represented by the system shown in Figure 4.7, where the terms "household reserve category" (HRC) and "household resource quality objectives", or "household RQOs" are introduced and can be compared with the "ecological reserve category", or ERC and "ecological RQOs" in the ER process (Section 3 of this Chapter). This system shows the relationship among the household reserve category (HRC), the BHNR determination and the household RQOs *for different levels of BHNR*. The relevance of this relationship is that the BHNR should not be regarded as an invariable quantity, quality and assurance of supply, since climatic as well as anthropogenic (socio-economic) conditions shape the BHNR. Although assurance of supply, water quality and water quantity are interlinked, it useful to focus on the unique characteristics of these components of the BHNR. As far as possible, parallels have been drawn between the system in Figure 4.7 and the summary for the assessment of the ER (*c.f.* Figure 4.6). However, the assurance of supply, water quality and water quantity of the BHNR possess different attributes to those of the ER.

First, the *assurance* of the BHNR for human or household use is expected to be most attainable where the goal is to meet only basic water rights for human survival. While such minimal conditions are not desirable for human systems, basic water conditions for human survival can be compared with the "hydrological drought" or "system stress" conditions of the ER (*c.f.* Section 3.2 of this Chapter). Water allocations which exceed the threshold required for human survival contribute towards the maintenance of "normal" quality of life conditions for people. Further increases in water allocations can contribute towards the "equitable" distribution of resources and are linked to increased household prosperity, in terms of both "quality of life" and socio-economic *functioning;* since improved quality of life can enable people to function better in a socio-economic sphere (*i.e.* improved sanitation improves general human health). Future populations potentially require, overall, increased water allocation for both "hydrological drought" conditions and "normal" quality of life conditions to the BHNR. Preferred conditions (or to use the terminology of the ER, the "desired future state") would necessitate the greatest demands on water resources to meet the household water requirements. Meeting these water requirements could be expected to result in increased equity assurance, increased socio-economic functioning and a high BHNR determination.

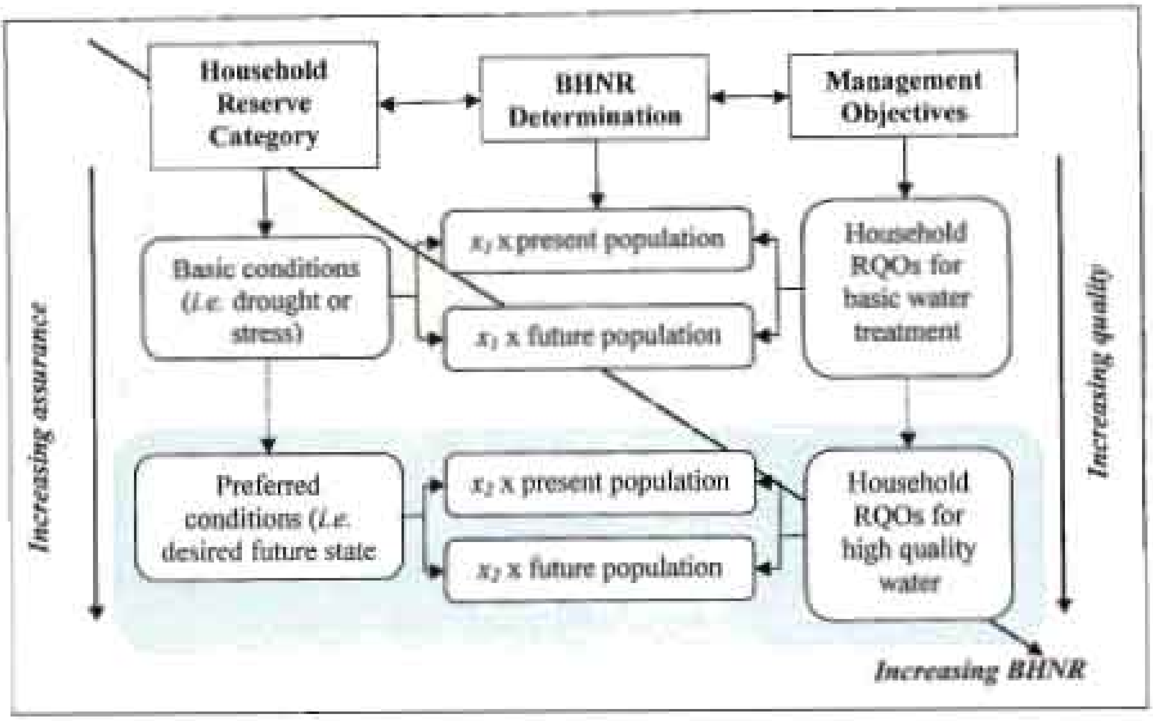

Figure 4.7 A proposed system for the assessment of the BHNR. x_1 refers to the minimum volume of water supplied (*i.e.* 25 litres per day for basic drought conditions). x_2 refers to a preferred water delivery.

Second, the *quality* of the BHNR will increase with household RQOs which address improved infrastructure and systems to abstract, treat and convey water for household supply. The minimum acceptable requirements for the quality of the BHNR necessitate basic water treatment, rendering it safe to drink "at the tap". This requirement is equally applicable for surface was well as groundwater sources. There are many difficulties associated with the infrastructure required to deliver the BHNR. While it is recognised that the infrastructure required for household water requirements is pivotal to the quality of the BHNR, any further discussion relating to the infrastructure required for delivery of the BHNR is beyond the scope this Chapter, but is discussed by Moriarty *et al.* (2004). Nonetheless, in many regions of South Africa, water from groundwater sources may be safe to drink with basic treatment, yet is often tainted by brackish water. Thus, in addition to improved water services delivery, the highest BHNR quality would be achieved with improved water palatability and enhanced removal of colloidal particles to improve water clarity and to reduce water conduit maintenance.

Third the *quantity* of the BHNR increases with the shift towards the desired future state of equity assurance, desired water quality and the expectations of present and potential future populations (*i.e.* a high HRC). While 25 litres per person per day may be sufficient for human survival conditions, this quantity should be regarded as the minimum BHNR since it represents the least desirable conditions for socio-economic functioning. These socio-economic conditions can be compared with the hydrological drought conditions of ecological systems which have an ERC of D (*i.e.* systems which possess poor ecological functioning). Linking the second and third targets of 60 litres and 100 litres per person per day (*c.f.* Section 4.3 of this Chapter), or any more appropriate allocation, with present and potential future populations could be considered to enhance the integrity of socio-economic functioning and could be compared with the desired future state of ecological systems which have an ERC of B or A (*i.e.* systems which possess a high degree of ecological functioning). In the same way as high ecostatus and assurance of the ERCs incur high allocations to the ER, such determinations of these household requirements would incur the high allocations to the BHNR.

While undesirable, it is necessary to consider human requirements for water under drought or stress conditions since water shortages do exist, either as a result of climatic conditions or as human induced water scarcity. A high level of BHNR may not be achievable if other ecological and socio-economic requirements are to be met, and a lower BHNR may be necessary. Thus, the worst case scenario for assessments of the BHNR can be represented by the minimum volume of water supplied (x_1 in Figure 4.7). The best case scenario for assessments of the BHNR occurs where satisfactory quantities of water are delivered for potable use (x_2 in Figure 4.7). These different "life" conditions introduce variability into the quantity of water required to meet the BHNR. Further variability in the magnitude of the BHNR results from supplying a growing number of people with water. Thus, the highest quantities of the BHNR (x_2 x future population) promote human health, just as high quantities of the ER promote ecological health. However, the state of the BHNR, and its variability, is not directly related to the streamflow regime, but rather to the infrastructure required to deliver it (Moriarty *et al.*, 2004). Management objectives which focus on increasing the quality of the water supply are also non-streamflow related. However, the impact of providing increased infrastructure together with increased service is that the volume of water required for the BHNR increases.

4.5 Constraints to the Determination of the Basic Human Needs Reserve

Unlike the determination of the ER, temporal variation does not present any major difficulties to quantification of the BHNR, apart from consideration of "when" to store water. However, the spatial distribution of the population presents many challenges in linking people with the water resources on which they depend, particularly if it is to be determined at the same resolution as the surface water *resource units* and / or groundwater *response zones* described in Section 3 of this Chapter.

The BHNR will be sourced from surface water streamflows, groundwater or a combination of both, particularly in water stressed catchments or when there is drought. All of these resources can be highly variable, both spatially and temporally. Some of the complexities for the determination of the Reserve (both the ER and BHNR) resulting from the connectivity and interrelationships of groundwater and surface water resources have already been mentioned in Section 3 of this Chapter. However, additional constraints to the determination of the BHNR relate to differences among the spatial distribution of people, supply schemes and resource availability as highlighted by the Resource, Infrastructure, Demands and entitlements (RIDe) model described by Moriarty *et al.* (2004).

Unlike the ER, which is a non-consumptive instream requirement, the BHNR is a consumptive off-stream water use. Consequently, it is not sufficient to merely allocate the BHNR at a point in a river, dam or aquifer within a catchment some distance from the intended population (Pollard *et al.*, 2002). Determining the BHNR should also anticipate evaporation, transmission and transportation losses between the point of its source and of its delivery.

4.6 Summary

In comparison to the determination of the ER, determining the BHNR, as defined by the NWA, is relatively straightforward involving a headcount and a pre-determined quantity of "survival" water. However, this seemingly straightforward assessment is not absolute; both the headcount and the quantity of survival water are flexible. It is anticipated that the BHNR will increase with time in terms of population demographics and increasing expectations of the basic human, or household, right to water. Most of the complexities of BHNR determinations relate to the spatial scale of the assessment, since the resource and point of use are often widely separated.

5 LINKING THE RESERVE WITH HYDROLOGICAL PROCESSES AND THE GENERATION OF ECOSYSTEM GOODS AND SERVICES

5.1 Introduction

As a consequence of the legislation relating to the Reserve, described in Section 2 of this Chapter, there has been increased cooperation among aquatic researchers, water managers and users in South Africa. This has resulted in significant progress in raising awareness of "ecological water rights" and "household water rights" among the major stakeholders. However, awareness alone is not sufficient to ensure judicious resource use or environmental security, since water users need to be equipped with the means to judge the impacts of different water uses on the water resource or, more pertinently, what any ERC scenario represents in terms of ecosystem goods and services, instream as well as off-stream. Moreover, while implementing the legislation on the Reserve implies a need for accurate assessments of the quantities, patterns, and quality of the flow requirements of the ER and of the BHNR, some of the greatest challenges to implementing the Reserve lie in understanding the relationships among the ER, the BHNR and other water users. This is all the more pertinent in light of the DWAF's goal to promote the beneficial use of water for all South Africa's citizens and its recent project (2005) to review existing and develop alternative approaches to water allocation in South Africa (DWAF, 2005).

In this Section, a framework is proposed for ecologically sustainable water resources management, linking the ER with the BHNR and other water users to the ecosystem goods and services that can be expected under different Reserve Categories. In light of the links between hydrological connectivity and ecological functioning (*c.f.* Section 3.2 of this Chapter) the proposed framework is structured around the concept that protecting water resources instream depends on maintaining hydrological functioning through protecting hydrological processes (*e.g.* precipitation, evaporation, transpiration, surface runoff, water infiltrability, diffusion, drainage and subsurface flow) which operate within catchments at large. Thus, the proposed framework addresses the disruption of the hydrological cycle, and any consequent alteration to hydrological processes, as a result of societal activities both on-stream and off-stream.

5.2 The Relationships among the Ecological Reserve, the Basic Human Needs Reserve and Water Users

While the form of the Reserve is well documented and promoted, its functions are less well understood. Concern has been raised that misinterpretations of the role of the Reserve in water resources management may contribute to obstacles in the implementation of the NWA (van Wyk *et al.*, 2006). The concept of the BHNR and "household water rights" is simple; in times of water scarcity, human needs for basic water consumption take priority over all other water allocation. However, the parallel undertaking of "ecological water rights", in the form of the ER, has raised skepticism, particularly in water stressed regions. Nonetheless, the vision for the ER is that it becomes a management tool for conserving the quantity, quality and patterns of flow required to maintain (a) the aquatic ecosystem functioning which society needs and (b) the sustainable use of water resources. To implement this vision will mean that the relationships among the BHNR, the ER and other water users need to be clearly defined. In many ways the nature of the relationships among the BHNR, the ER and other water users have been precipitated by the imminent allocation of licences to registered and potential water users (*c.f.* Figure 4.2). However, the information presented in Figure 4.2 can be misleading; giving rise to the perception that once the Reserve requirements (typically, though incorrectly, only the ER is addressed) have been determined, or "set", at a particular ERC decided by the stakeholders, any "spare" water in the resource is wholly allocable for economic use. Proceeding to allocate water use licences in such a way as to fully utilise the water resource, so that catchment water conditions become "closed", does not provide for any "reserve" in the real sense of its intended purpose, *viz.* securing long-term benefits of sustainability for the catchment (*c.f.* Figure 4.8). As in Seckler (1996), cited by Molden *et al.*, (2001), the narration of "closed" water conditions in this thesis refers to situations, either catchments or zones, where water resources are fully allocated to environmental and other water users.

Figure 4.8 summarises the current assessment of the allocable water and in many respects combines the attributes of Figures 4.6 and 4.7. This is understandable since Reserve determinations (*c.f.* Figures 4.6 and 4.7) and the assumption that there is spare water in aquatic ecosystems (*c.f.* Figures 4.2 and 4.8) has formed the foundation of South African EFAs (O'Keeffe, 2000). However, rather than the link between increasing habitat integrity and increasing ER in Figure 4.6 and the parallel link between increasing services and increasing BHNR in Figure 4.7, the pathway between a "Closed Catchment" and a "Protected Catchment" is much more tenuous, requiring a complete paradigm shift from a water supply management to a water demand management focused approach.

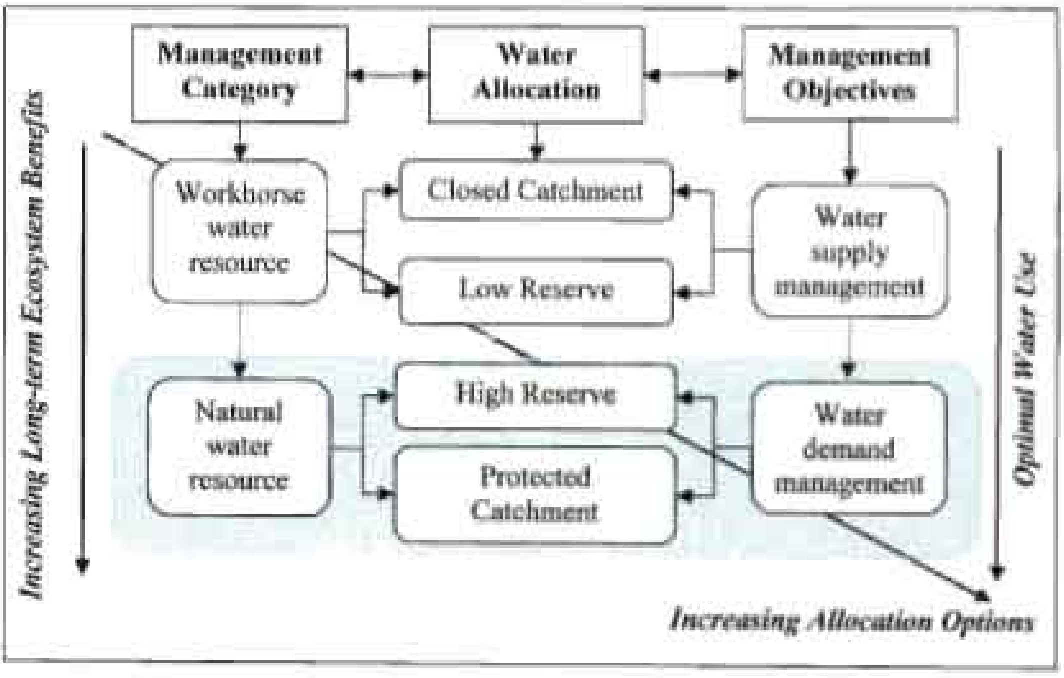

Figure 4.8 Summary of the current assessment of allocable water

As described in Section 3 of this Chapter, the most advanced approaches to the assessment of environmental flow requirements reflect the compositional and structural diversity as well as the natural functioning of aquatic ecosystems. Since it is recognised that the hydrological regime drives variation in many other ecological components, the natural variability of the hydrological regime is the key component in this biophysical assessment. Unlike the ER, the BHNR and water allocable for economic use are not directly streamflow related, yet they are dependent on the benefits provided by hydrological functioning. While current methods account for societal water use in a system yield and scenario approach, a framework for ecologically sustainable water resources management is proposed, using an ecosystem approach and adaptive management to link ecological requirements and societal dependence on the benefits provided by aquatic ecosystems.

5.3 Proposed Framework for Ecologically Sustainable Water Resources Management

The framework for ecologically sustainable water resources management (*c.f.* Figure 4.9) presented in this Section is based to some extent on a similar framework proposed by Richter *et al.* (2003). The framework in Figure 4.9 incorporates adaptive management techniques and, similarly to the other such approaches (for example see Richter *et al* 2003, but more pertinently the process outlined in Figure 4.2), can be initiated from several stages within the framework structure. However, rather than focusing on "spare water" which is allocable for societal use, as outlined in the process for water resources management shown in Figure 4.2, the concept behind the framework for ecologically sustainable water resources management proposed in this Section is that *societal well-being and biodiversity are inextricably linked* (Figure 4.9). The structure of the proposed framework in Figure 4.9 is similar to any process involving environmental impact assessment and incorporates the essential elements of Scoping or Problem Definition (shown as Assessment), Assimilation, Resolution (dealt with here in an Ecosystem Approach (*c.f.* Chapter 3) and the classic Adaptive Management and Implementation Phase. However, the fundamental basis of the formulation of the framework proposed in Figure 4.9 is the catchment to river system relationship.

5.3.1 Catchment to river system relationship

The ecostatus of a river system is not solely streamflow related. River systems are strongly linked to terrestrial environments rather than being disconnected bodies or simply drainage systems (Baron *et al.*, 2002). Moreover, the ecostatus of a river system is largely a reflection of the land care management operating at a catchment-wide basis. Several societal activities across the catchment have the potential to degrade river systems by impacting on the quality of the water resource in terms of changes in environmental factors such as temperature, suspended solids or dissolved oxygen content. The hydrological cycle links the different aquatic ecosystems of rivers, wetlands, lakes and groundwater, with terrestrial ecosystems beyond the stream channel and floodplain through its energy,

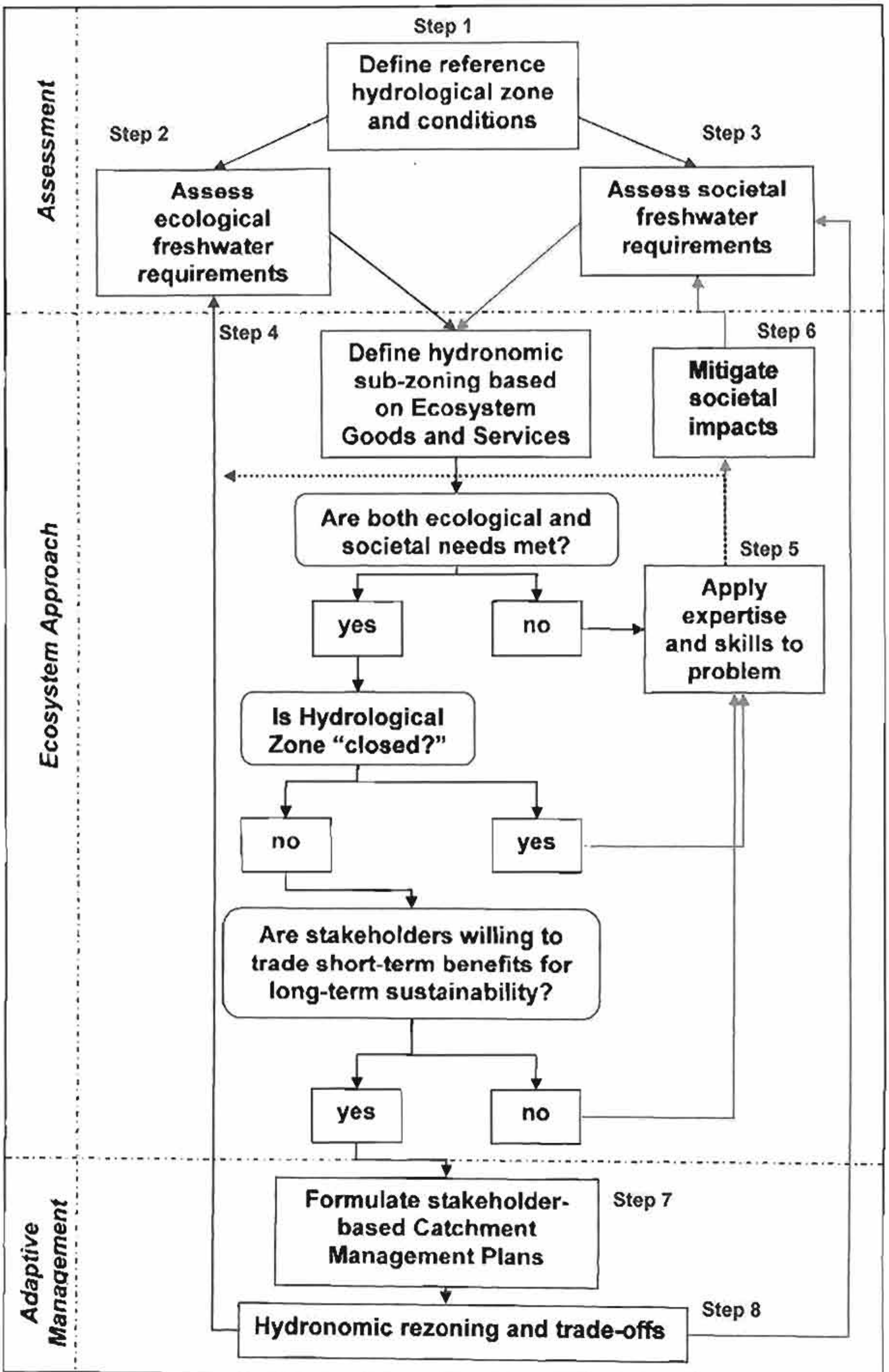

Figure 4.9 A proposed framework for ecologically sustainable water resources management (much modified from Richter *et al.*, 2003)

sediment and water flows. However, while freshwater assessments have focused predominantly on societal appropriation of liquid water flow in rivers, lakes and reservoirs, society also depends on water vapour flow for the generation of ecosystem goods and services (Jansson *et al.*, 1999). Every societal need for freshwater, whether it is off-stream or instream, can be described in terms of its interaction with, and disruption of, the hydrological cycle. However, it is relevant to note that at the catchment scale, precipitation is the basic water resource (Falkenmark, 2003). Figure 4.10 indicates the relationship among precipitation, terrestrial ecosystems and aquatic ecosystems.

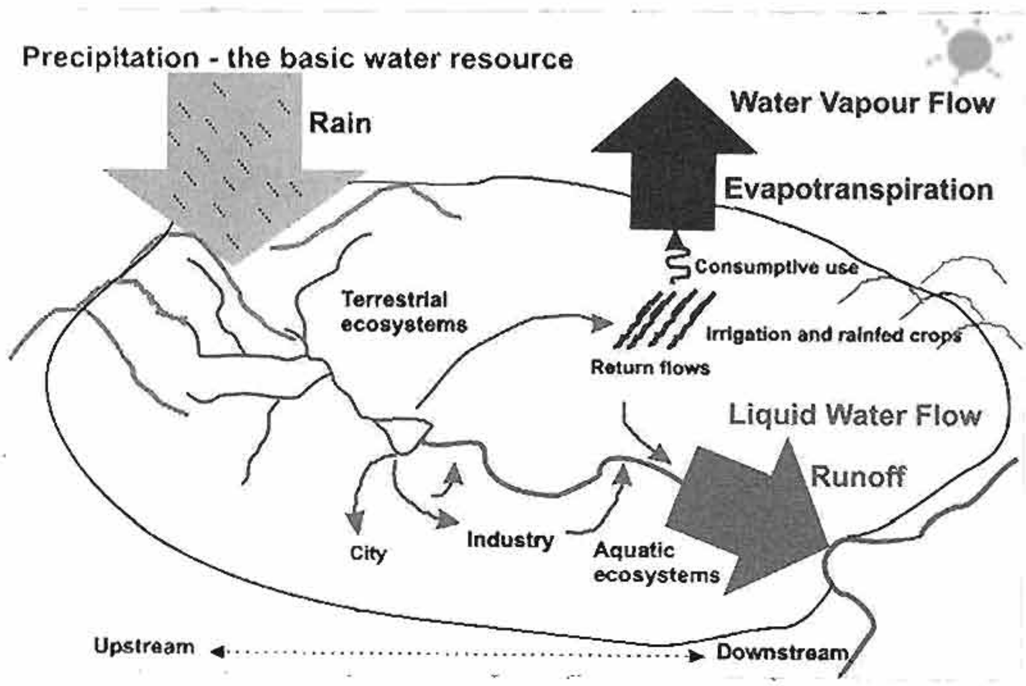

Figure 4.10 The relationship among precipitation, terrestrial ecosystems and aquatic ecosystems (after Falkenmark, 2003)

5.3.2 Reference zones for managing hydrological response

In addition to the societal needs for freshwater, either through the appropriation of liquid water flow or its dependence on water vapour flow, humans and society act as agents of dispersal within and across ecosystems. Thus, in the proposed framework it is recognised from the outset that humans and society form an integral component of any terrestrial or aquatic ecosystem. One of the principal misconceptions among stakeholders regarding the protection of ecological functioning of a water resource is that the ER is perceived to be in direct competition with society for water (van Wyk *et al.*, 2006). In the proposed framework, this misunderstanding is addressed by attempting to match the spatial scales of ecosystems with societal systems.

Stakeholders require information on ecosystem functioning under natural, or reference, conditions in order to ascertain the likely ecosystem response to alterations of the streamflow regime as a result of any societal water use and water development. However, as discussed in Chapter 2 of this thesis, defining reference conditions in EFAs, including the South African Ecological Reserve determinations, tends to focus on the biophysical delineation of water resource units (*i.e.* the DWAF Resource Units, *c.f.* Sections 2 and 3 of this Chapter) from "ecoregions" to hydrological response units within the catchment. By and large, the delineation of resource units is defaulted to the Quaternary Catchment scale, particularly in the lower confidence methods of Reserve determination. This biophysical delineation overlooks the sensitivity of land uses, resource management practices and socio-economic conditions which operate within catchments. However, the RDM guidelines (DWAF, 1999) advise a breakdown of ecoregions into resource units which are suitable for management requirements, which could be on the basis of the major land uses. This approach is applied in this study, since delineating catchments in such a manner that societal influences can be ascertained in addition to reference biophysical conditions should facilitate the resolution of any conflict between ecological and societal needs for freshwater.

5.3.3 Hydronomic zones at a fine organisational scale

The merits of the hydronomic zones (*hydro* = water and *nomus* = management) described by Molden *et al.* (2001) as a tool for ecologically sustainable water resources management were discussed in Chapter 2, Section 4.5. However, adopting a management programme which focuses on the *fate of water outflow from the "unit"* as proposed by Molden *et al.* (2001) is complex from a spatial perspective in terms of defining the "hydrological response (resource) unit" or "hydronomic zone", since within any "unit" there can be several societal activities, each of which interrupts the hydrological cycle in different ways.

The hydronomic [sub-] zones proposed in Figure 4.9 could build on the set of six hydronomic zones developed by Molden *et al.* (2001), *viz.*, the water source zone; natural recapture zone; regulated recapture zone; final use zone; stagnation zone and environmentally sensitive zone. However, while Molden *et al.* (2001) focus on whether or not the water outflow from a zone is recoverable for downstream use, *an additional attribute of the zoning presented here* would be to reflect the optimal use of instream flows and stored water in the upstream zone.

Similarly to the DWAF's Resource Unit, the scale of a hydronomic zone could be as wide as a catchment or as deep as groundwater. Likewise, catchment areas, ecoregions or river reaches could be identified and managed as water conservation resources, "workhorse" resources, or those of intermediate water utilisation, depending on the societal value attributed to the short-term economic benefits and the long-term sustainability of the water resource.

In this framework for ecologically sustainable water management, it is proposed to focus on hydronomic sub-zones at an *organisational scale* which is finer than the resolution of reference hydrological response units, or zones, such as Quaternary Catchments. In this Chapter, the term *organisational scale* is used to define catchment landscape units which are governed by a societal activity which disrupts the hydrological processes in a different way to the societal activity in an adjacent unit.

The fundamental principles of the current proposal are that *societal well-being and biodiversity are inextricably linked* and that *the ecosystem goods and services generated by healthy ecosystems justify the call for the protection and stewardship of the water resource*. Defining hydronomic sub-zones to represent water use in a catchment, based not only on "biophysical resource units", but also on the potential generation of ecosystem goods and services, is proposed as a useful spatial and organisational scale for managing ecosystems and maintaining ecological and societal needs for freshwater.

As illustrated in Figure 4.10, ecologically sustainable management of catchment water resources should address the freshwater requirements of both ecological and societal systems. In the proposed framework, hydronomic "sub-zones" are defined *within* the reference hydrological response units, or zones, since this finer resolution represents the scale at which stakeholders are most likely to trade short-term benefits for longer-term sustainability. Consideration is given to the impacts of different societal activities on the hydrological cycle and is directed to the *relationship among precipitation, terrestrial* *ecosystems and aquatic ecosystems*. To illustrate this concept, eight different types of hydronomic sub-zones are suggested for application in the proposed framework: ***Conservation sub-Zone, Streamflow Reduction sub-Zone, Supplementary sub-Zone, Transmission sub-Zone, Recession sub-Zone, Succession sub-Zone, Diversion sub-Zone and Abstraction sub-Zone***. Table 4.4 provides definitions of the attributes of these zone types, based on the impacts of societal disruption of the hydrological processes linking precipitation, terrestrial and aquatic ecosystems. Examples of the application of these zone types is described below with reference to the main societal activities and conditions practiced within catchments in South Africa.

5.3.4 Applying the hydronomic sub-zone types to the proposed framework

Commercial forestry in South Africa is known to intercept precipitation, increase evapotranspiration and reduce streamflow generation. This relationship is well documented elsewhere and need not be repeated here. The relevant feature is that the increased biomass of commercial forestry (when compared to the grassland it replaces) results in disturbance to the relationship between hillslope processes and hydrological partitioning. Thus, commercial forestry disrupts the energy and nutrient pathways of terrestrial ecosystems. However, the focus of proposed framework is the impact of commercial forestry on the hydrological partitioning of precipitation and hillslope processes (*e.g.* surface runoff and subsurface flow *c.f.* Figure 4.11), since a change in these features can manifest as a change in the streamflow characteristics which are important for maintaining biodiversity. Projecting such a change could be expected to result in reduced aquatic ecosystem resilience to disturbance in the flow regime which, in turn, could lead to reduced ecosystem goods and services.

Commercial agriculture largely comprises rain-fed as well as irrigated crops and livestock grazing for economic profit, with the bulk of the agricultural produce being consumed far from the water source. ***Subsistence agriculture*** largely comprises rain-fed crops for local human and livestock consumption. Similarly to commercial forestry, these practices utilise the relationship between precipitation, evapotranspiration and, to a lesser extent, runoff. While in some instances precipitation made be collected from rooftops for supplementary use (see below), this practice generally amounts only to small additional quantities of water. Thus, for the purposes of the proposed framework, commercial and subsistence

Table 4.4 Definitions of the hydronomic sub-zones suggested for application in the proposed framework for ecologically sustainable water resources management

Sub-Zone Type	Location of Zone	Societal Activity	Disruption to hydrological processes
Conservation	Typically, but not uniquely, in more remote areas Areas of scientific or special interest	Minimal anthropogenic impact Stewardship approach to ecosytem management of long-term benefits	Minimal disruption to hydrological processes
Streamflow Reduction	Either on hill slopes or lower, more fertile land Catchment wide	Planting of rain-fed crops for harvesting of short-term ecosystem benefits	Increased interception of precipitation Increased evapotranspiration Decreased infiltration Reduced streamflow generation
Supplementary	Typically on lower, flat land Catchment wide	Planting of crops which require supplementary water for optimal harvesting of short-term ecosystem benefits	Increased interception of precipitation Increased evapotranspiration Reduced streamflow generation Increased percolation and stormflow generation in downstream zones
Transmission	Lowest area of the hillslope or catchment	Clearance of natural vegetation Mismanagement of water systems Conveyance of materials and transport	Increased interception of precipitation Increased evapotranspiration Increased transmission of liquid water flow from hill slopes, channel bank and water table storage to adjacent (transmission) zone Reduced streamflow generation Alteration of streamflow regime
Recession	Catchment wide	Clearance of natural vegetation Mismanagement of water and soil systems	Reduced interception of precipitation Increased infiltrability Reduced evapotranspiration Reduced subsurface flows Increased stormflow generation
Succession	Catchment wide	Mismanagement of water and soil systems, leading to Opportunistic biomass production	Increased interception of precipitation Increased evapotranspiration Reduced streamflow generation
Diversion	Catchment wide	Regulation and transfer of streamflows	Streamflow diverted from natural pathway
Abstraction	Catchment wide	Regulation and / or *in situ* abstraction of streamflows	Streamflow diverted from natural pathway

agriculture are also considered to disrupt the hydrological cycle through changes in interception, evapotranspiration and runoff. This is in keeping with current South African water legislation (NWA, 1998) which recognises both these practices as being either StreamFlow Reduction Activities (SFRA) or potentially SFRA.

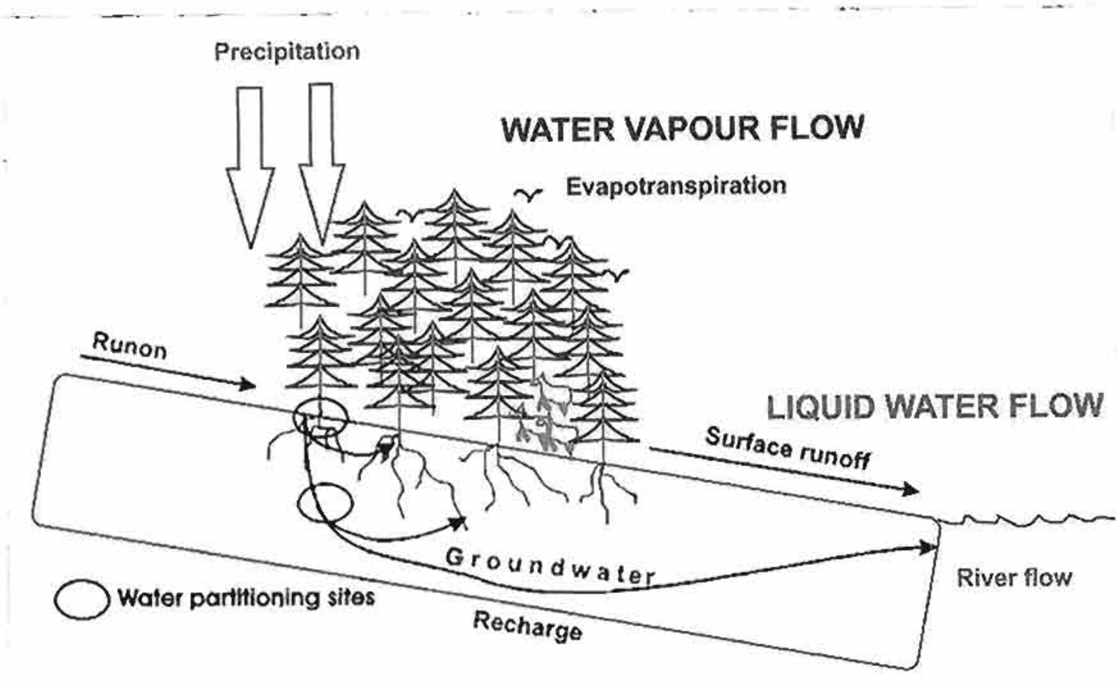

Figure 4.11 The relationship between precipitation, water vapour flow (evapotranspiration) and liquid water flow (runoff) in a forest ecosystem (after Jansson *et al.*, 1999)

Consequently, commercial forestry, rain-fed commercial agriculture and subsistence agriculture can all be allocated to the ***Streamflow Reduction sub-Zone***. However, commercial forestry is allocated to a further sub-zone of *Deep-rooted Streamflow Reduction Zone*, since tree roots occupy greater depths and volumes of the soil and sub-soil profile where they can continue the process of soil water extraction throughout the year, providing there is available subsurface water. Rain-fed commercial agriculture and subsistence agriculture are allocated to a different sub-zone of *Shallow-rooted Streamflow Reduction Zone*. The roots of rain-fed crops occupy much shallower depths since they are generally seasonal crops, performing the process of soil water extraction only during their growing season, when precipitation is available.

Commercial irrigation also intercepts precipitation, increases evapotranspiration and generally reduces streamflow generation although its impact on streamflow generation downstream can be complicated by additional deep percolation and additional surface runoff. Nonetheless, the defining attribute of commercial irrigation is its requirement of the appropriation of ***supplementary*** liquid water (runoff and groundwater) for increased biomass throughout the plant's growing season. The amount of supplementary liquid water required by the plant to ensure optimal yield depends on soil medium properties and local climate conditions. The major sources for supplementary liquid water are reservoir and farm dams which capture high flows resulting from rainfall events as well as direct abstractions from rivers. In this way, commercial irrigation supplements precipitation through river regulation and abstraction of streamflows. Thus, commercial irrigation is allocated to the ***Supplementary sub-Zone***.

Alien riparian vegetation directly intercepts precipitation, increases evapotranspiration, reduces streamflow generation and additionally supplements the precipitation it intercepts through channel and hillslope ***transmission*** processes. Elmore and Beschta (1987) define riparian areas as the "narrow strips of land" that border stream channels or wetland areas. Moreover, they state that while these areas may only occupy a small portion of the overall catchment area, they perform very important roles in connecting the stream channel to the overall landscape. Under natural conditions, riparian vegetation is quite distinct from that of the adjacent slopes and uplands as a result of its proximity to water storage in the channel banks and water table (Elmore and Beschta, 1987). The hydrological functioning of this water storage is disrupted by alien riparian vegetation, which generally results in increased *transmission* of liquid flow from the channel bank and water table storage to the plant through increased root osmosis, photosynthesis as well as evapotranspiration, and reduction of subsurface flows towards the stream.

A river channel and its banks provide a longitudinal connectivity in addition to the lateral and vertical dimensions described above. It is this longitudinal dimension which is used opportunistically by (a) society to convey a variety of materials and transport as well as to facilitate recreational activities, (b) animals as pathways to facilitate different life cycles and as refugia and (c) plants through seed dispersal. River channels and the streamflow regime provide the main agents of alien invasive species (both plant and animal). Thus, for the purposes of the proposed framework, alien invasive riparian vegetation is allocated to the ***Transmission sub-Zone***.

When precipitation falls on ***degraded land*** (or ***urban*** areas), the natural vegetative interception processes are replaced by reduced interception, reduced infiltration, reduced evapotranspiration, reduced subsurface flows but increased stormflow generation. In this way, the natural hydrological processes can be described as being in decline or *receding*. The extent to which this unnatural disturbance in hydrological processes can be tolerated by terrestrial and aquatic ecosystems is a measure of their inherent resilience. This emphasises the need for stewardship management practices at a sub-zone level. While it cannot always be expected that the impacts of urban areas can be mitigated (*e.g.* delayed storm water techniques can be engineered), the impacts of degraded land conditions can be ameliorated with restorative catchment management practices. Thus, for the purpose of the proposed framework, degraded land is allocated to a ***Recession sub-Zone*** in recognition that such receding attributes to the hydrological processes could be rehabilitated.

A different motive for stewardship approaches to sub-zone management arises when considering the impacts of ***encroaching thicket and bushland*** replacing natural grassland. Natural succession of vegetation is a fundamental ecosystem process, whereby species composition and organisation within an ecosystem changes over time. Some species within the ecosystem may become less abundant, or may be out-competed and disappear. Other species may become more abundant or encroach the system from adjacent ecosystems. While succession is a natural process, human and societal impacts often act as catalysts, with people acting as dispersal agents. Notwithstanding the causes of succession, management of the process may be required if conservation issues (see below) are a high priority, since thicket and bushland encroachment increases precipitation interception, evapotranspiration and reduces streamflow generation at higher rates than the grassland it replaces. Thus, thicket and bushland is allocated to a ***Succession sub-Zone*** for the purposes of the proposed framework.

Judicious catchment management plans should incorporate areas to be conserved as well as areas to be protected. It is important to note that the terms conservation and protection are not strictly interchangeable; conservation implies wise or good use, whereas protection implies the provision of a safe habitat for biota. However for the purposes of the proposed framework both of these practices are referred as "conservation" activities. Conservation areas are traditionally perceived to be located in more remote parts of a catchment, where a minimum of societal influence is experienced. While conservation areas can be managed at different spatial scales from a catchment-wide to local scale, the management aim should be to introduce only a minimal anthropogenic impact to the hydrological processes of precipitation interception, evapotranspiration and streamflow generation. The presence of conservation or protected areas, even at the sub-zone level, is the key to biodiversity. Worldwide, there are instances where conservation management provides direct benefits to local people and society. For example, maintaining conservation corridors adjacent to forest plantations provides a diverse habitat for pollinating insects, including bees, which in turn produce honey as a food and economic resource for local communities. Thus, ***Conservation sub-Zones*** should be maintained within the main hydronomic zones (*i.e.* the water management zones which match the spatial scale of the reference hydrological zones).

The requirements of the BHNR impact directly on liquid water flow, but consumption of the BHNR has the potential to impact indirectly on water vapour flow. In addition, the assessment of the location of a zone which describes the influence of the BHNR on hydrological processes is complex, as the point of delivery is generally disconnected from the source of its impact on the interception of both precipitation and streamflow. Where the BHNR is consumed outside of the hydrological zone from which it is extracted, this household water right could be could be allocated to a ***Diversion sub-Zone.*** Alternatively, where the BHNR is consumed locally it could be allocated to an ***Abstraction sub-Zone***.

Additional zones could be designed to describe the impacts of other societal activities on the streamflow regime, particularly where water quality impacts have an influence on the ecostatus of a river system. For example, mineral extraction could be allocated to a ***Reuse sub-Zone,*** since mining frequently *reuses* rather consumes liquid water flow, whereas the practice of floodplain agriculture could be allocated to a ***Deposition sub-Zone***, since this floodplain area benefits from the deposition of the nutrient and sediment flows of the flood regime.

5.4 Components of the Proposed Framework

Returning to Figure 4.9 it can be seen that Steps 1, 2 and 3 together form the Assessment component, Steps 4, 5 and 6 together with relevant questions, form the Ecosystem Approach, and Steps 7 and 8 together form the Adaptive Management and Implementation Phase. This Section addresses each of these steps.

Step 1 Define reference hydrological zone and conditions

While reference conditions in EFAs clearly include conditions relating to ecological and geomorphological functioning, the main aims of this thesis relate to the hydrological basis for the protection of water resources. Thus, while it is recognised that hydrological reference conditions are only one component applied in determining the ecostatus of water resources, the hydrological regime is known to be the key driver of variability in aquatic ecosystem functioning. As a consequence, defining reference hydrological conditions is the cornerstone of EFA.

Understanding ecological functioning, and to a lesser degree societal functioning, requires high resolution information of the relative processes, both temporally and spatially. In addition to the reference hydrological zones being based on the biophysical region in which each zone is located, reference hydrological conditions, which should take cognisance of natural hydrological characteristics such as annual rainfall, seasonality and variability of streamflow, are required. Hydrological indices of intra- and inter-annual variability which describe the dominant patterns of variability of streamflow in varying climatic and geological conditions in "natural", or minimally impacted, catchments are appropriate for ecologically relevant streamflow classification (*c.f.* Chapter 5) and for defining reference hydrological conditions. Reliable, long-term records of streamflows, at a minimum of a daily step, are required for the extraction of ecologically relevant hydrological indices. Where natural streamflow conditions have been compromised, or where the streamflow record is inadequate in length or quality, the hydrological indices can be determined by applying hydrological simulation modelling to generate time series of daily streamflows under recognised "baseline," or reference, land cover conditions such as those represented by Acocks Veld Types (Acocks, 1988).

Step 2 Assessing ecological freshwater requirements

The requirements of aquatic ecosystems are often viewed as being at odds with human activities (Baron *et al.*, 2002; van Wyk *et al.*, 2006). In common with the philosophy driving the water resources management process shown in Figure 4.2, Step 2 of this framework views ecosystem functioning as the foundation of long-term water resources sustainability for society. Functioning ecosystems have ecological needs for freshwater, defined as ecological flow requirements (*c.f.* Section 3.2 of this Chapter). *Fully* functioning ecosystems deliver a high level of biodiversity, thereby supporting a high volume of ecosystem goods and services. Thus, assessment of the hydrological status, with regard to the ecological flow requirements of an aquatic ecosystem, should determine the key characteristics of the natural streamflow regime in question, and the natural range of variability in each of these characteristics, to maintain the functionality of the ecosystem (Richter *et al.*, 2003). This is a departure from the assessment of the streamflow requirements of several different biotic components of the ecosystem applied in many EFR methods, including those applied in South African Reserve determinations outlined in Section 3.2 of this Chapter. Current South African Reserve determinations of the quantity component of the flow requirements of the ER assess each ecological component (and often for a limited number of biota) separately before integrating these requirements into a consolidated IFR and then reconciling an IFR time series with water resources through yield modelling and scenario assessment.

Furthermore, it is not reasonable to expect that aquatic scientists will be able to provide comprehensive or exact values and details of the ecological flow requirements of each species, aquatic and riparian communities or entire ecosystems (Richter *et al.*, 2003). Nonetheless, ecological flow requirements and any objectives designed to meet them (Step 7) need to be quantitatively defined in order to integrate them with other components of water management objectives (Rogers and Bestier, 1997). Ecological flow requirements can be expressed as a numerical range within which key streamflow characteristics should be maintained or, alternatively, a target window with upper and lower boundaries can be prescribed for a specified frequency of a streamflow characteristic (Richter *et al.*, 2003). For example, it may be acceptable for a streamflow characteristic to be attained in a certain range of its natural frequency in only half the years (Richter *et al.*, 1998).

An important consideration in the assessment of ecological flow requirements is that initial determinations should be made without consideration of whether the requirements are attainable, either presently or even in the short-term. The premise of the South African Reserve determinations is that ecological sustainability should be presumed to be attainable in the longer term in some *desired future ecological state*. This sentiment has been confirmed for different frameworks for ecologically sustainable water resources management in different global regions, for example in the USA (Richter *et al.*, 2003).

Step 3 Assessing societal freshwater requirements

Societal needs for freshwater, in the strictest sense, constitutes the BHNR in South African water resources management. Also, in the strictest sense, the BHNR represents the water needs of merely another species having its niche in the ecosystem, yet, as described in Section 4 of this Chapter, this freshwater need is not streamflow related, either temporally or spatially. However, pursuing the argument presented in Section 4 of this Chapter, the BHNR should be a variable entity, reflecting the infrastructure and delivery status required to meet a variable demand (*i.e.* varying consumption rates; varying population estimates) for the service.

In the broader sense, societal needs for freshwater for many different societal activities also take place across the catchment and, as such, are also not directly streamflow related, either temporally or spatially. In addition, the variability associated with the water required for industrial, agricultural, urban, household and recreational activities is governed by the water management status (*c.f.* Figure 4.8) rather than environmental variability or consumption rates.

Although onerous, the assessment of societal freshwater requirements is generally less complex than that of ecological flow requirements. Nonetheless, complexities in the assessment arise when predicting future requirements, since the uncertainties of societal systems lie in population demographics, economic developments, expectations and resilience as well as vulnerability in terms of adaptability of society to environmental change.

The zoning approach to meeting ecological and societal freshwater needs, presented in this Chapter, represents a different approach to the allocation of water among the BHNR, the

ER and other water users, whereby user requirements are determined in much the same way as the ER and BHNR. Managing water demand through conservative use is inherently preferable to supplying water resources on demand. Thus, rather than allocating water use licences based solely on Reserve Determinations, in terms of either *"how much water must be left in the ecosystem?" or "how much water can be taken out of the ecosystem?"* (*c.f.* Section 3.2 of this Chapter), the amount of water that is *just sufficient for any particular economic use* could be also be determined. This approach opens the way for innovative technologies to be developed to ensure efficient water use while stimulating economic growth at all societal scales (*c.f.* Step 6).

Step 4 Define hydronomic sub-zoning based on ecosystem goods and services

Using a hydronomic zoning approach, based on the ecosystem goods and services that meet both ecological and societal needs for freshwater, would identify resource units (or hydronomic sub-zones) where water should be conserved; where protection is required and where genuine water use efficiency can be implemented to enhance water quality and quantity for downstream use (Molden *et al.*, 2001). The strength of the hydronomic sub-zones (*c.f.* Sections 5.3.3 and 5.3.4 of this Chapter) in the proposed framework is that they reflect the dynamic relationship between societal values and ecosystem functioning. Consequently, it is envisaged that defining hydronomic sub-zones will integrate the freshwater requirements of both ecological and societal systems and, in addition, highlight any areas of incompatibility or conflict where the limits of ecosystem functioning are breached by human influences. A series of three questions concerning the main hydronomic zones (*i.e.* the water management zones which match the spatial scale of the reference hydrological zones) and the hydronomic sub-zones are included in the proposed framework.

- **Can both ecological and social needs for freshwater can be met?**

Increasingly, hydrological records, either observed or simulated, are used as an ecological resource to analyse the impacts of societal development on different characteristics of the streamflow regime. Long-term hydrological records are extremely useful for water management studies since they incorporate any human induced change and can be used to simulate scenarios of environmental, catchment or water management change. Thus the hydrological records of key streamflow characteristics can be analysed to evaluate whether both ecological and societal needs can be met within and among years, and under a variety of proposed water management practices, for any particular hydronomic zone or sub-zone. Areas of conflict between the ecological and societal needs for freshwater can be identified by comparing ecological freshwater requirements (Step 2) with the flow regime resulting from the societal needs (Step 3) using the comparison of pre- and post-change in the hydrograph as demonstrated by the IHA method (Richter *et al.*, 1996). If areas of conflict are found, the process follows through to Step 5, where water resource managers and stakeholders may have to draw on the expertise and skills of scientists and researchers to address the problem. If there is no conflict between the ecological requirements and societal needs, then two additional questions must be considered before the process proceeds to Step 7.

- **Is the main hydronomic zone closed?**

Whether a basin is "closing", "closed" or "open" forms part of the water management formulating strategies adopted by IWMI (see Molden *et al.*, 2001). The closed hydronomic zone condition (*c.f.* Section 5.2), or state, represents a zone where utilisation has imposed severe restrictions for managing future benefits from the water resource (*c.f.* Figure 4.8). In such cases modified ecosystem functioning may be sufficient to deliver a relatively high volume of short-term socio-economic benefits, such as the artificial services associated with reservoirs or irrigated fields. However, the long-term environmental costs incurred by reduced biodiversity or increased pollution may be serious threats to the long-term sustainability of those benefits. The shift towards the closed hydronomic zone state is a possibility for all main hydronomic zones (*i.e.* the water management zones which match the spatial scale of the reference hydrological zones), although the least desirable instance would be for this state to impinge on any water source zone. This scenario would severely limit natural ecosystem functioning and biodiversity in the downstream zone(s), with the effect that societal well-being of downstream users could be compromised by the activities of upstream use. The aim of ecologically sustainable water resources management is to avoid the closed hydronomic zone state where possible, since optimal hydronomic zones maintain the relationship between the trade-off between short-term socio-economic benefits and the long-term benefits of fully functioning ecosystems (*c.f.* Chapter 2, Section 2.2 and Figure 2.2). If the closed hydronomic state becomes a reality, Step 5 of the proposed framework is invoked. However, so long as options are maintained, another question must be considered in the proposed framework. This is posed below.

- **Are stakeholders willing to trade short-term benefits for long-term sustainability?**

Arguably the most important consideration for truly integrating ecological and societal needs for freshwater is whether stakeholders are willing to trade short-term economic benefits for long-term sustainability (*c.f.* Chapter 2, Figure 2.2). Rather than dealing with the two systems in a piecemeal manner, an Ecosystem Approach is applied in the proposed framework to maximise the total benefits provided by ecosystem functioning, whilst conserving biodiversity.

The provision and maintenance of ecosystem goods and services are cited as justification for the protection of catchment resources. The value of ecosystem goods and services, in both the short-term and the long-term, is the basis for ecosystem approaches to water management. However, stakeholders often lose sight of the long-term benefits (*e.g.* biodiversity and sustainability in favour of the short-term benefits such as timber and food production. There is a defined need for the benefits of ecosystem approaches to freshwater resources to be clearly demonstrated.

One way in which this could be achieved would be to model the impacts of each major activity (represented by the sub-zones described in Sections 5.3.3 and 5.3.4 of this Chapter) on the dominant hydrological indices which characterise a particular streamflow type in its reference hydrological condition (*c.f.* Step 1). Highlighting the differences among the hydrological impacts of different societal activities at this fine organisational scale would equip stakeholders with improved information relating to their short-term and longer term water use within the catchment (*c.f.* the Mkomazi Catchment Case Study presented in Chapter 6).

Step 5 Apply expertise and skills to the problem

Step 5 of the proposed framework, whereby resource managers and stakeholders may have to draw on the expertise and skills of scientists and researchers to find ways in which to alleviate the problem(s), is invoked in the following instances.

(a) Incompatibility between the ecological and societal needs for freshwater is identified.

(b) A closed hydronomic state becomes a reality.

(c) The benefits provided by the short-term economic value of utilising the liquid water flows and / or the water vapour flows are perceived to outweigh the costs of securing long-term sustainability.

Once these problems have been identified, both spatially and temporally, negotiations can begin among all the interested parties to find ways in which to mitigate the harmful influences of societal activities. In order to seek acceptable solutions, all three instances require collaboration and clear dialogue among the affected stakeholders, water managers, researchers and scientists. This is one of the major challenges to the successful implementation of the ER in South African catchments, where many stakeholders view the ER as being in direct competition with people for water (van Wyk *et al.*, 2006). If there is an absence of "environmental will" among stakeholders, then it is unlikely that the misconceptions regarding the intent of the ER will be rectified. Constructive debate among all the interested and affected parties is anticipated to clarify the role and function of the ER in the sustainable use and protection of the diversity of the ecosystem benefits to society. Collaborative dialogue in the search for mitigation of the negative trade-offs of over-utilisation of water resources benefits from local knowledge as much as from technical skills and expertise.

Step 6 Mitigate societal impacts

Incompatibility between ecological and societal systems (instance 'a' in Step 5) may be alleviated when stakeholders accept that their needs may not be met fully every year in all areas of the catchment and that some compromise is required in order to avoid the main hydronomic zone becoming closed (instance 'b' in Step 5). In developing catchments the incentive to increase the productivity, or harvest, of an ecosystem is considerable, particularly where economic forces are a factor and short-term benefits are needed (instance 'c' in Step 5). However, water resource management objectives should also evaluate the negative trade-offs, including any unforeseen impacts, that may arise from any disruption to the hydrological regime. In all water management approaches, but particularly in situations when any of the three instances highlighted above prevail, it is necessary to formulate mutually agreeable goals which are incorporated into stakeholder-based Catchment Management Plans (Step 7). However, first, water management goals and objectives must be formulated to mitigate the societal impacts and negatives trade-offs among ecological and societal systems.

Step 7 Formulate stakeholder-based Catchment Management Plans

The zoning approach described by Molden *et al.* (2001), and modified in this study, provides a framework to develop sets of water management strategies that are suited to the different conditions that exist within catchments. The formulation of water management strategies into Catchment Management Plans (CMPs) is akin to designing RQOs as described in Section 2.4 of this Chapter. Clearly, it is judicious firstly to protect the water source zones of the catchment, since these are areas where most of the runoff or groundwater recharge for downstream use is generated (Molden *et al.*, 2001). It is also where water management strategies can have catchment-wide impacts. The hydrological connectivity operating within the catchment to the river network system is a powerful incentive for CMPs to include upstream-downstream trade-offs between the off-stream societal activities in different parts of the catchment. This concept could be enacted through stakeholders in headwater or water source zones receiving compensation from downstream communities for not engaging in SFRAs. Stewardship approaches to environment management such as the South African Working for Water and Land Care Programmes are also appropriate contenders in ameliorating any negative impacts on the upstream-downstream relationship. Alternatively, stakeholders may accept that water demand management policies (*c.f.* Figure 4.8) are an acceptable option, if they are prepared to modify their current water use through water saving strategies or the adoption of more efficient technologies.

Step 8 Hydronomic rezoning and trade-offs

The main goal of an ecosystem approach to water resources management is to maximise the generation of ecosystem goods and services while conserving biodiversity (McCartney *et al.*, 1999). Accordingly, the goal of the proposed framework is that after reviewing any incompatibilities between ecological and societal freshwater requirements and mitigating any impediments to resolve any potential conflicts, ecologically sustainable water resources management should be achievable through stakeholder collaboration. The zoning approach incorporated in the proposed framework provides the spatial, temporal and organisational structure for matching any incompatibles between these two interlinked systems (*i.e.* ecological and societal systems). Clearly the desired outcome of the proposed framework is that the optimal use of catchment resources is achieved while maintaining biodiversity for future options. Decisions regarding the optimal use of catchment water resources are best made by those who have a vested interest in the outcome. Undoubtedly there is a need to incorporate an economic consideration in the proposed framework. If stakeholders can assign a monetary value for ecosystem goods and services, they will be better informed to ascertain ERCs.

The discipline of ecological economics is gaining ground in EFA. However, most ecological applications are restricted to assigning an economic value to the tangible goods provided by aquatic ecosystems, rather than assessing the much more valuable, yet unseen, ecosystem services since these are more complex. Moreover, excluding ecosystem goods and services generated by the hydrological cycle through water vapour, or those ecosystem goods and services supplemented by the hydrological regime through the appropriation of liquid water, from the economic valuation in environmental flow assessments fails to realise the true worth of hydrological functioning to stakeholders. Ecological economics and the full assessment of ecosystem goods and services are beyond the scope of this study. However, in the absence of such information an indirect approach to hydronomic rezoning and trade-offs is to define management targets for ecologically relevant sustainability indicators of the hydrological regime. First, however, it is necessary to revisit the management function of the RQOs (*c.f.* Section 2.4 of this Chapter) formulated to meet a particular category of ecological integrity (*i.e.* an ERC, *c.f.* Section 3.2 of this Chapter).

5.5 Matching the Ecological Reserve Category with Ecosystems Goods and Services

Among the most relevant questions posed by the South African water resource classification system employed in current RDMs (*c.f.* Figure 4.2) is "*How does the hydrological regime deliver the ecosystem goods and services associated with the Ecological Reserve Category selected by the various catchment water users*?" In a similar vein, Breen (2001) asked *"How does the ecological state [of a water resource] deliver the goods and services required"*. On the face of it, the answer to this type of question is that the recommended IFR for a site of ecological and / or social importance should deliver the ecosystem goods and services that stakeholders desire. However, as described in Section 3.2 of this Chapter, the route between prescribing a streamflow regime to meet a particular ecostatus and the implementation of the IFR recommendation is complex, requiring hydrological models which take cognisance of existing streamflow conditions and climatic cues (Hughes and Ziervogel, 1998). As highlighted in the proposed framework, the streamflow regime is the result of many different environmental processes, including, but not restricted to, hydrological processes which operate at different spatial, temporal and organisational scales across catchments.

As described in Section 2.3 of this Chapter, the function of the RQOs is that they should equate to specific management objectives or targets for each of the four categories of desired or acceptable ecological integrity (A to D, *c.f.* Table 4.2) whilst providing "a means for defining management thresholds for the water resource" (Godfrey and Todd, 2001). Godfrey and Todd (2001) propose a method to define management thresholds for freshwater indicators (*e.g.* ecologically relevant hydrological indices) using the principles of the NWA (NWA, 1998). The method is based on the concept that for a water resource there is a threshold representing each of the following (Godfrey and Todd, 2001), *viz.*

(a) the desirable or acceptable level of ecosystem functioning lying between the limit of naturalness and the limit of sustainability (the management threshold, *e.g.* possible management levels, or ERCs, of A to D);

(b) the point beyond which the resilience (limit of sustainability) of the ecosystem is exceeded (system threshold); and

(c) the point at which the ecosystem will be unable to recover to its natural equilibrium state or to an acceptable or desirable level of ecosystem functioning (critical threshold).

Godfrey and Todd (2001) illustrate these levels, limits and thresholds in a threshold model, which is shown in Figure 4.12. The limit of naturalness is defined by Godfrey and Todd (2001) as the upper threshold for the realm of reference conditions and as including natural fluctuation or variation within the ecosystem. In addition, Godfrey and Todd (2001) described this limit as being between 80 to 120% of reference conditions (*c.f.* Figure 4.12). It is possible for a river system to have more than 100% of its natural flow if it is, for example, used as a conduit for downstream irrigation, for diluting purposes or for recreational events. However, any of these societal activities could cause changes to the natural ecological functioning and are, thus, not indicative of the "naturalness" of an ecosystem. The desirable or acceptable level of ecosystem functioning is arrived at with the co-operation and deliberation of the various stakeholders in the resource (DWAF, 1999; Rogers and Biggs, 1999). The desired state (or "realm of sustainability") represents a range of water conditions within which the water resource, and provision of ecosystem goods and services, is sustainable (*c.f.* Figure 4.12) and the desired ecological management class lies within this "realm". Although it is generally accepted that the ERCs A to D are not thresholds but a continuum (Hughes, 2005), aquatic researchers (*e.g.* Godfrey and Todd, 2001 and Brown *et al.*, 2006) refer to this management level as management thresholds (point 'a' above). The resilience of the water resource, where ecosystems are capable of regaining their natural state after stress or disturbance, links the dynamic attributes of ecological systems with the institutional attributes of social systems (Raufflet, 2000) and lies between the limit of acceptability and the limit of sustainability (the "realm of intervention", *c.f.* Figure 4.12).

The realm of intervention is where management decisions could be made to prevent a water resource from deteriorating towards unsustainable levels (the system threshold of point 'b' above). Beyond the limit of sustainability, the water resource becomes unsustainable and the ecological equilibrium of the water resource is altered as the critical threshold is reached (point 'c' above). The system "collapses" at the critical threshold, beyond which irreversible damage occurs (Godfrey and Todd, 2001).

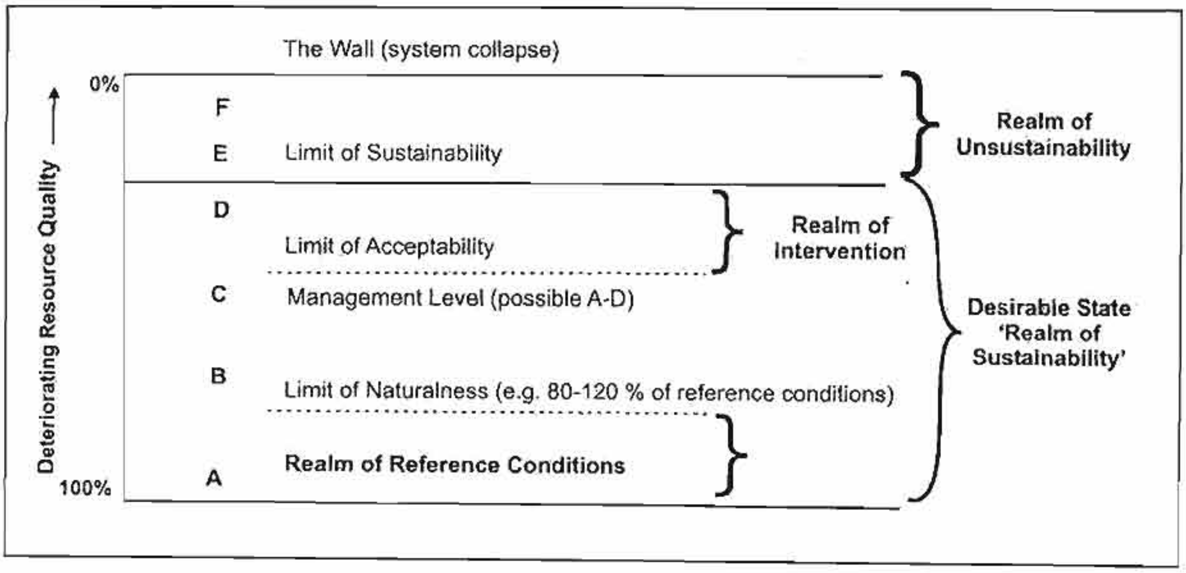

Figure 4.12 Threshold model for sustainability indicators within the context of the South African water resource management (after Godfrey and Todd, 2001)

The method described above assimilates some of the philosophy behind concepts such as "thresholds of probable concern", TPCs (Rogers and Bestbier, 1997) for defining system and management (acceptability and sustainability) thresholds. The method could be useful for assessing the state of an ecosystem indicator or entire system. However, ecosystem, critical and management thresholds are generally unknown and Mackay (2001) advocates that a quantitative risk-based approach to the setting of RQOs is not yet achievable in South Africa because of a lack of both ecological data and expertise (Jooste, *et al.*, 1999). Godfrey and Todd (2001) acknowledge this and suggest that applying a range of percentages of reference, or unimpacted, conditions could suffice to quantify acceptability, sustainability and system thresholds as well as change within a water resource.

While the method could be applied for some components of ecosystems, ecosystems do not become unsustainable nor do they collapse at particular points or levels. However, the method does acknowledge the roles of natural flow variability and hydrological alteration as well as disturbance and resilience in ecosystems. The suggestion by Gordon and Todd (2001) of applying scaled down conditions for the thresholds of ecosystem indicators bears some similarity to the more advanced hydrologically-based environmental flow methods such as the Range of Variability Approach, RVA (Richter *et al.*, 1997) which focuses on the natural range of streamflow variability to produce a recommended flow regime.

Nonetheless, selecting a suite of sustainability or ecosystem indicators and defining management targeted ranges of variation, or thresholds of probable concern, for each ecosystem component requires a high level of understanding of the ecological processes and functioning of the synergy among the hydrological regime, human activities and ecosystem response (Richter *et al.*, 2003). This dilemma contributes to the difficulties inherent in the valuation of the benefits of most ecosystem goods and services associated with catchment water resources.

Defining management targets for river networks

An important component of the proposed framework for ecologically sustainable water resources management described in Section 5.4 of this Chapter is the identification of any potential incompatibility or conflict between the ecological and societal needs for freshwater (Step 4 in Figure 4.9). In the description of Step 4 of the proposed framework it was suggested that these issues could be assessed by comparing ecological flow requirements with the flow regime resulting from meeting societal needs, using the comparison of pre- and post-development change in the hydrograph as demonstrated by the IHA method (Richter *et al.*, 1996). Daily streamflow hydrographs resulting from societally impacted conditions can provide both visual and statistical comparisons between flow requirements for the two interlinked systems (Richter *et al.*, 2003). However, this method is best applied with intra-annual flow parameters which describe short-term hydrological conditions such as extreme low flows or flood events, since these attributes are particularly relevant in ecological terms. Richter *et al.* (1997) developed the Range of Variability Approach (RVA) for defining management or restoration targets prescribed on the basis of the natural variability in the streamflow characteristics of the 33 intra-annual flow parameters comprising the IHA method (Richter *et al.*, 1996). Using the RVA approach, water managers can aim to maintain annual values of the different flow parameters within a pre-set target range which defines a portion, or all, of the natural range of variability in any particular flow parameter. This is entirely in keeping with the threshold model proposed by Godfrey and Todd (2001), where the "desired state or realm of sustainability" is actually a range of conditions for different ecosystem indicators.

Ideally, the RVA management targets should be based on available ecological information. However, there is a paucity of such information in southern Africa and in such instances the developers of the RVA recommend that the target range is based on selected percentile levels or a single multiple of the parameter standard deviations for the natural or pre-development streamflow regime. It is not the intention that the river attain the target range every year, but rather that the target range is attained with the same frequency as occurred in the natural or pre-development streamflow regime (Richter *et al.*, 1998). The developers of the RVA recommend that in the absence of any ecological information, the 25th to 75th percentile range is selected for preliminary targets, since attainment of an RVA target range defined by these percentiles values of any particular parameter would be expected in only 50% of the years. Monitoring of the ecosystem response to the preliminary targets should identify system flow thresholds for components of the river ecosystem and allow subsequent refinement of the flow-based management targets (Richter *et al.*, 1997).

5.6 Determining which Hydrological Components Deliver Desirable Ecosystem Goods and Services

Despite the acknowledged relationship between the hydrological regime and aquatic integrity there is surprisingly little information in the literature regarding the *relationship* between specific ecosystem goods and services (*c.f.* Table 2.1 in Chapter 2) and different aspects of the hydrological regime, far less the *volume* of ecosystem goods and services that can be expected from different hydrological components. Mostly, the information in the literature relates to the *influence* of different hydrological components on aquatic and riparian ecosystems (*e.g.* Table 4.5 and Chapter 2, Sections 3.3 and 3.4). For example, hydrological extremes (in both high flow and low flow disturbances) are recognised for the constraints they impose on both ecological and societal communities. In both ecological and societal systems, periods of hydrological stress drive inherent and natural adaptive processes. In ecological systems stress is an important natural selection process and results in increased resilience and biodiversity in ecosystems. In societal systems stress is an important development process and results in improved management strategies and technological innovation. As described in Section 3.2.2 of this Chapter, in recent years, the process for ER determinations has incorporated a Flow-stress response component (O'Keeffe *et al.*, 2002) for application to low flows to assist aquatic researchers in quantitatively assessing the impact of flow-related *stresses* on biotic *response*, in terms of abundance, life stages and persistence. However, knowledge among stakeholders of which hydrological components actually deliver desirable ecosystem goods and service to society is generally restricted to the value of (a) high flow events in providing increased food production and security, both instream and off-stream and (b) low flows to those who rely on run-of-river abstractions.

Most off-stream societal uses of water (*e.g.* the BHNR, supplementary irrigation and mining) as well as some instream activities (*e.g.* hydroelectric power generation) do not require naturally flowing rivers to operate; the storage or diversion of high flows is generally sufficient to meet these needs. However, the timing or duration of irrigation scheduling may be advanced if there is a hydrological drought. The duration, timing, frequency and rate of change in the streamflow regime are every bit as important as the magnitude of streamflows in delivering desirable ecosystem goods and services and the

Table 4.5 Summary of hydrologic variables used in the Indicators of Hydrological Alteration, and their characteristics (after Richter *et al.*, 1998)

General Group	Regime characteristics	Streamflow parameters	Examples of ecosytem influences
1. Magnitude of monthly discharge conditions	Magnitude, timing	Mean discharge for each calendar month	Habitat availability for aquatic organisms Soil moisture availability for plants Influences water temperature, oxygen levels, photosynthesis in water column
2: Magnitude and duration of annual extreme discharge conditions	Magnitude, duration	Annual maxima one-day means Annual minima one-day means Annual maxima three-day means Annual minima three-day means Annual maxima seven-day means Annual minima seven-day means Annual maxima thirty-day means Annual minima thirty-day means Annual maxima ninety-day means Annual minima ninety-day means Number of zero flow days Seven-day minimum flow divided by mean flow for year (baseflow)	Balance of competitive, ruderal, and stress-tolerant organisms Creation of sites for plant colonisation Structuring of aquatic ecosystems by abiotic vs. biotic factors Structuring of river channel morphology and physical habitat conditions Soil moisture stress in plants Dehydration in plants Anaerobic stress in plants Volume of nutrient exchanges between rivers and floodplains Duration of stressful conditions such as low oxygen and concentrated chemicals in aquatic environments Distribution of plant communities in lakes, ponds, floodplains Duration of high flows from waste disposal, aeration of spawning beds on channel sediments
3. Timing of annual extreme discharge conditions	Timing	Julian date of each annual one-day maximum discharge Julian date of each annual one-day minimum discharge	Compatibility with life cycles of organisms Predictability / avoidability of stress for organisms Access to special habitats during reproduction or to avoid predation Spawning cues for migratory fish Evolution of life history strategies, behavioural mechanisms
4. Frequency and duration of high / low pulses	Magnitude, frequency, duration	Number of high pulses each year Number of low pulses each year Mean duration of high pulses each year Mean duration of low pulses each year	Frequency and magnitude of soil moisture stress for plants Frequency and duration of anaerobic stress for plants Availability of floodplain habitats for aquatic organisms Nutrient and organic matter exchanges between river and floodplain Influences bedload transport, channel sediment textures, and duration of substrate disturbance (high pulses)
5. Rate / frequency of hydrograph changes	Frequency, rate of change	Means of all positive differences between consecutive daily values Means of all negative differences between consecutive daily values Number of flow reversals	Entrapment of organisms on islands, floodplains (rising levels) Drought stress on plants (falling levels) Desiccation stress on low-mobility stream edge organisms

merits of these characteristics are revisited in Chapters 5, and 6 as well as Appendix 6A of this thesis.

Hydrological variables which describe different components of the streamflow regime include indicators of general streamflow variability and predictability as well as indices of high flow disturbance and low flow disturbance (*c.f.* Chapter 5). Both aquatic and terrestrial scientists use a variety of hydrological variables, or indicators, to interpret the relationships among different ecosystem components and the generation of ecosystem goods and services. As described in Chapter 2, Section 2.4.4, hydrological indicators are valuable measures of aquatic integrity.

Several researchers have developed ecologically relevant classifications of naturally flowing rivers based on their geographical distribution (*e.g.* Joubert and Hurly, 1994; Poff, 1996). In addition to describing the spatio-temporal characteristics of streamflow regimes, hydrological indicators are sensitive to human influences such as land use change, reservoir operation, groundwater abstractions and agricultural diversions which disrupt the hydrological cycle. As a result, a plethora of hydrological indices which describe various aspects of streamflow regimes exists for use in hydro-ecological studies. The remainder of this thesis is directed to the investigation of whether hydrological indices can be used as surrogates for assessing how the hydrological regime delivers desirable ecosystem goods and services.

5.7 Summary

Some of the greatest challenges to implementing the Reserve lie in understanding the relationships among the ER, the BHNR and other water allocations. If this understanding is misdirected there is a danger that a low ER will be allocated and that water users will forfeit the long-term benefits of future options and sustainability in favour of the more immediately available economic benefits. A framework for ecologically sustainable water resources management is proposed, advocating the delineation of catchment landscape zones which account for both reference hydrological conditions (hydrological zones) and societal activities and management options (hydronomic sub-zones). The zoning approach presented in the proposed framework considers the disruption of the hydrological cycle through changes to key hydrological processes as a result of societal activities both on-stream and off-stream. The main emphasis of the proposed framework is that societal well-being and biodiversity are interlinked. This focus is expected to resolve some of the misunderstandings concerning the ER. However, there are still knowledge gaps regarding how the Ecological Reserve Category of a river actually delivers the ecosystem goods and services that society desires.

6 DISCUSSION AND CONCLUSIONS

The South African National Water Act (NWA) was promulgated in 1998 with the intent that the nation's finite water resources be used for the benefit of its diverse and growing population (NWA, 1998). The NWA promotes a shift in policy from traditional water impact assessments to strategies which include source directed controls (source protection) and resource directed measures (resource protection). The Resource Directed Measures (classification of the nations water resources, determination of the Reserve and the Resource Quality Objectives) described in Section 2 of this Chapter are just one of several sets of measures (*c.f.* Figure 4.2) outlined in the NWA, which "when implemented together will ensure sustainable, equitable, efficient, optimal use of South Africa's water resources" (Xu *et al.*, 2002).

As described in Section 2 of this Chapter, the basic premise of the Reserve is that sustaining ecosystem goods and services for human use, including water for life-sustaining functions (the Basic Human Needs Reserve), depends on the protection of the resource base for ecological functioning (the Ecological Reserve). This is a bold approach to water resources management, since the water needs of humans and society are frequently viewed as being in competition with natural ecosystems. However, there is growing evidence that water resources management conducted in an ecologically sustainable approach, where ecological flow requirements are recognised, does not need to compromise freshwater ecosystems while providing for human needs (Richter *et al.*, 2003). The move towards ecologically sustainable water resources management has precipitated from recognition that humans and society cannot continue to utilise water in ways which degrade *their* environment.

The concept of the South African Reserve (NWA, 1998) is well known in water management and environmental affairs and refers to two different, yet interconnected Reserves, of the *water resource* which are "set aside (or reserved) before any other water use demand is considered". The term "Ecological Reserve" is understood to be the condition of a water resource required to meet predetermined ecological reserve categories of ecological health, the latter being more frequently referred to as a system's ecostatus. The term "Basic Human Needs Reserve" refers to the basic human, or household, right to water for survival. However, the Reserve is still perceived by some to be some sacrosanct volume of water that is "set-aside" for the environment and which could be more beneficially allocated to other water uses. Such definitions of a "Reserve", based on the concept of "set aside" can be misleading on several levels since they diverge from the commonly held understanding in environmental management that both the quantity and quality of a "reserve" of a finite natural resource can vary according to the development of technological, political and economic mechanisms. First, the South African Reserve should be rather viewed as a water *resource* which sustains future water use options. Second, the Reserve should not be viewed as having absolute dimensions, but rather as a variable entity dependent on environmental and societal conditions. Using the example of oil (also a finite natural resource), "reserves" are measured in the number of barrels or the projected number of years of oil supply that can be delivered using currently viable techniques, which can be expected to change according to technological, political and economic forces. Notwithstanding the ongoing depletion of the (oil) resource, the assessment of the future state of "oil reserves" is not static since technological developments may permit extractions of the resource which are currently unviable, thus adding to the "total reserves". Such a definition of a "reserve" which inherently incorporates the socio-politico-economic influence on the availability of the "resource", may have a use in better defining the "environmental water requirements" of ecosystems (including the human component) and partitioning of water as a limited or finite (*e.g.* in the case of groundwater) resource within the hydrological cycle. Rather than viewing aquatic resource protection as "how much water must be left in the system?" or "how much water can be taken out of the system?" (*c.f.* Table 4.2), it may be more beneficial to adopt a "reserve" concept where the full value of the water resource (including both off-stream and instream) is accounted for in accordance with present and future environmental and social conditions. This approach may assist in linking the benefits of protecting water resources with associated ecosystem goods and services.

Notwithstanding any misinterpretations of the South African Reserve, the main goal should be the long term protection and management of the ecological, societal and economic importance of the Nation's water resources. Each of these attributes is, in turn, dependent on the ability of aquatic ecosystems to maintain their ecological diversity and functioning. Society depends on the ecological functioning of water resources for basic human needs and a wide range of ecosystem goods and services which maintain societal and economic well-being. Thus any change in ecological functioning in an ecosystem results in a change in the goods and services provided by that ecological component.

Any assessment of a change in ecological functioning will require constructive debate around stakeholder expectations of aquatic ecosystems. However, public expectations of aquatic ecosystems may be related to how the resource unit (see Section 2.5 of this Chapter) looks (for example river level or clarity; King *et al.*, 2000; Calder, 2002), whereas scientists and water managers may wish to focus more on the value of ecological functioning (for example, the ecosystem benefits resulting from restoration of natural river flows; Naussauer, 2001). The vision for the ER is that it becomes a management tool for conserving the quantity, quality and patterns of flow required to maintain (a) the aquatic ecosystem functioning which society needs and (b) the sustainable use of water resources. However, assessing environmental flow requirements on all relevant spatial scales, from river reach to catchment, for instream and beyond channel flows, is complicated. The framework for ecologically sustainable water resources management outlined in Section 5 of this Chapter is proposed as a step towards meeting the Department of Water Affairs and Forestry's vision for the ER and goal of beneficial and equitable water use. The principal benefit of ecologically sustainable water resources management is environmental security, where the array of ecosystem goods and services required by inter-generational humans needs is sustained.

The BBM method for assessing environmental flow requirements of the ER (for rivers) to meet the vision for South African water resources has to some extent (*i.e.* currently, low flows only) been superceded by the development of the FS-R method. However, in the FS-R method a similar basis to the BBM method is applied for setting some of the initial flow requirements for various different species. Despite the fact that several potential flow regimes may be determined for each of the ERCs set for each section of a river, and that those flow regimes are tested in a water resources systems yield model, neither method can be described as one which fully integrates ecological and societal water requirements into a sustainable secure environment for present and future generations. In essence, this type of biophysical assessment, which focuses on aquatic habitat, resembles a patchwork of the flow requirements of several different single aquatic species, which is more likely to isolate the landscape rather than integrate it with societies' needs. While the rationale in current methods is that the initial IFR assessments are carried out independently of societal needs because the requirements for a particular level of ecological protection and the ability to meet that level are independent of each other, it is considered that the approach presented in Figure 4.9 facilitates a more *holistic ecosystem* approach. Moreover, the goal of providing habitat for ecosystem functioning may not be enough to ensure ecologically sustainable water resources management. In addition, the current biophysical assessments of environmental flow requirements tend to minimise the impacts of land use beyond the Resource Unit of the mainstream channel and therefore disconnect the hydrological connectivity of hydrological-landscape zones that are so integral to catchments. For example, although the RDM process attempts to reconcile any discrepancies, several different ERC categories can be prescribed along a river reach from the headwater region to the estuary. This highlights the need to assess the ecological and societal freshwater needs along the tributaries which connect moderate water source zones to the mainstream river system.

Nonetheless, the determination of the Ecological Reserve is still in its infancy and while it is important that a "process" to do so is identified, the procedure should be characterised by adaptability, incorporating innovative ways of assessing the dynamic systems in which people live. The emergence of the DRIFT approach, which assesses scenarios of different flow regimes and the likely impacts of "losing" some components of the natural flow patterns, has demonstrated the evolutionary nature of this exciting field of research.

The proposed framework for ecologically sustainable water resources management in Section 5 of this Chapter, presented as a link between societal well-being and the long-term protection of aquatic ecosystems, recognises that the hydro-ecostatus of a river system is not solely streamflow related. Worldwide, ecosystem management by catchment has been recommended as a strategy for water resources management. This is enshrined by the South African Water Law Principles of 1996 which focus on integrating catchment management issues (DWAF, 1996). However, most strategies advise the delineation of catchments based on ecoregions or other physical factors. The integration of hydrological-landscape zones with other more societally influenced activities presented by the proposed framework in Section 5 of this Chapter is a different approach to water resources management since it acknowledges the hydrological connectivity operating in the physical landscape with societal systems. The framework complies with the requirements of the RDM guidelines (DWAF, 1999) for the description of resource units and enhances the delineation by providing a societal link at the outset of any environmental flow assessment. By focusing on how societal activities interact with and disrupt hydrological connectivity, stakeholders are better placed to make decisions about their catchment water use. However, the extent of any disruption and the trajectory of change in that disruption still need to be assessed.

Hydrological indicators, or, more pertinently, hydrological indices have an important role in environmental flow assessments particularly where other ecological data are limited. Hydrological indices are good candidates for measuring the extent of any change in ecosystem functioning. If both long historical hydrological records and ecological records are available, the extraction of ecologically relevant hydrological indices can assist in determining "thresholds of probable concern" which can be used to invoke management options. Alternatively, preliminary management targets can be set to achieve a specified frequency of a hydrological index, which can be monitored in conjunction with ecosystem response. Moreover, in this Chapter it is suggested that the value of hydrological indices in describing the streamflow characteristics which influence the generation of ecosystem goods and services may also be useful for describing how the hydrological regime matches a volume of ecosystem goods and services with a particular ecological reserve category.

Presently, the aquatic environment has a relatively strong voice in South African water resources management as a result of prior development of the instream flow requirement and BBM processes which provided a springboard for ER determinations (Pollard *et al.*, 2002). Nonetheless, first priority is legally given to the BHNR, yet this component of the Reserve still needs to be clearly defined in terms of quantity, quality and assurance of supply. The framework proposed in Section 5 of this Chapter defined the relationship between ecological needs and societal needs for freshwater, including the relationships between the South African ER, BHNR and other water allocations. In many ways, the aquatic resource base described as the South African Water Reserve represents the core environment resource to help cope with the rapidly growing aspirations for equity among the nation's people. Yet, in keeping with any other natural resource its status depends on technological expertise to realise its full potential and efficient management to ensure its optimal usefulness. Minimising water use through innovative techniques has evolved as human and societal pressure on water resources increases. However, it is critical that such developments are implemented if the "ecological footprint" is to be minimised, while the "headcount" grows.

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CHAPTER 5 HYDROLOGICAL INDICES OF ECOLOGICAL WATER REQUIREMENTS OF RIVERS IN SOUTH AFRICA

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STRUCTURE OF CHAPTER 5 HYDROLOGICAL INDICES OF ECOLOGICAL WATER REQUIREMENTS OF RIVERS IN SOUTH AFRICA

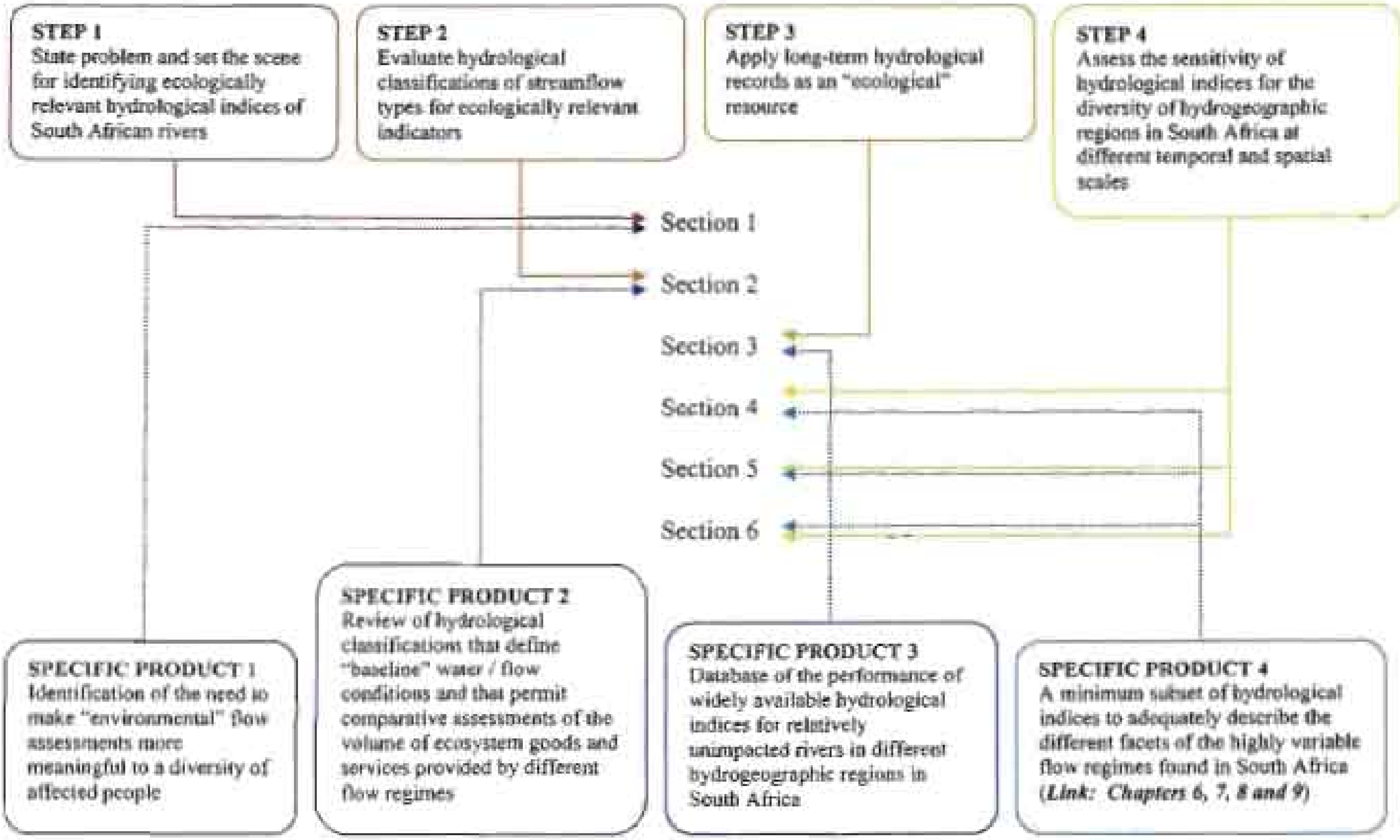

Figure 5.1 Structure of Chapter 5: Hydrological indices of ecological requirements of rivers in South Africa

CHAPTER 5 HYDROLOGICAL INDICES OF ECOLOGICAL WATER REQUIREMENTS OF RIVERS IN SOUTH AFRICA

1 INTRODUCTION

Streamflow determines many of the physiological (*e.g.* gas and temperature exchange) and biological (*e.g.* nutrient cycling) processes in stream channels, banks, floodplains and hyporheic zones (Hynes, 1970; Poff, 1996). Moreover, the streamflow regime links many ecological processes in freshwater ecosystems and plays an important role in determining the structure (*e.g.* channel morphology), composition (*e.g.* occurrence and distribution of aquatic and riparian biota) and functioning (*e.g.* water quality, water temperature, transportation of sediment and organic matter, estuarine inflow and other environmental conditions) of aquatic systems (Junk *et al.*, 1989; Poff *et al.*, 1997; Richter *et al.*, 1996; 1997). Streamflow is naturally variable over time and space. On the temporal scale, the magnitude of a flood may be many times greater than that during a drought; the timing and duration of the annual flood pulse may vary from year to year; the frequency of both droughts and floods may increase or decrease and the rates of change between rising and falling river levels may fluctuate in any given year (Riggs, 1985). Consequently, aquatic ecosystems are in a state of dynamic flux. The present state of an aquatic ecosystem results from adaptation to a range of streamflow conditions that occurred in the past, yet, ecosystems are "reset" by disturbance (either natural or anthropogenic) to evolve to a future state (Carpenter, 2001). Variability among rivers in different climatic, geological and topographic regions results in further streamflow variation and spatial heterogeneity in the ecological organisation of streams (Poff, 1996).

Hydrological variability is considered to be an important driver of the habitat template in stream ecosystems (Minshall, 1988; Poff and Ward, 1990), influencing habitat and food-web biodiversity (Thoms and Sheldon, 1997). Numerous studies have investigated the influence of various streamflow characteristics on species distribution and life cycles, as well as community structure and functioning (Poff and Ward, 1989; Jowett and Duncan, 1990; Poff *et al.*, 1997; Richter *et al.*, 1996; 1997; Clausen and Biggs, 1998). Yet, ecologists still face difficulties in predicting and quantifying biotic response to altered streamflow regimes (Bunn and Arthington, 2002). This predicament is compounded by uncertainties regarding the impacts of land use change on streamflow regimes and a series of untested hypotheses relating to aquatic ecosystem response to altered streamflow regimes (Bunn and Arthington, 2002). For the most part, aquatic ecological studies have been performed in relatively shallow, perennial rivers in temperate climatic regions and extrapolation of any ecological theories such as the River Continuum Concept, proposed by Vannote *et al.* (1980), to the ecological characteristics of intermittent and ephemeral streams, may be unsound and require modifications (Walker *et al.*, 1995).

With the global move towards integrating increased stakeholder participation in environmental flow assessments (EFAs), there is a need to identify hydrological indices[1] that are readily understood by a diverse group of interested parties. Specialists need to be able to convey the attributes of "ecologically relevant hydrological indicators" (Poff *et al.*, 1996; Richter *et al.*, 1996) to non-experts. However, there is diversity in focus among the specialists involved in EFA. For example, geomorphologists and aquatic biologists tend to focus on specific streamflow components or events, but the former group may view a high streamflow event in terms of its scouring potential whereas the latter group may be concerned with its diluting properties. In addition, different ecological, spatial and temporal scales complicate the understanding of such streamflow events. Since the 1990s, there has been considerable debate concerning the natural streamflow regime and addressing the full range of intra- and inter-annual streamflow variability and associated characteristics of magnitude, timing, duration, frequency and rate of change, which are considered critical in sustaining the full natural biodiversity of aquatic ecosystems (Richter *et al.*, 1997). This proposition has strong misgivings, particularly in regions where people strive for economic growth and social well-being under harsh physiographic conditions and where scarce water resources are, additionally, compromised by high climatic variability. In South African water resource assessments it has been expressed that where

[1] The term "hydrological index" is used increasingly in the evolving language of environmental flows to represent a measure of a "hydrological parameter" or "hydrological variable". While, in this Chapter, the terms hydrological parameter(s) and variable(s) are used where they accurately reflect the terminology used by other researchers in the literature, where possible, "hydrological index" or "hydrological indices" is used in preference.

the environment and people compete, allocating the full range of natural streamflows to the environment is not a viable proposition (O'Keeffe, 1998). Nonetheless, measurements of "natural" hydrological variation that are ecologically relevant are perceived as essential "trend indicators" for successful river management (Walker *et al.*, 1995).

Recently, there has been a resurgence of interest internationally in the application of hydrological indices, based on the statistical analyses of certain "ecologically relevant" streamflow characteristics, to predict the likely ecological response to variation in streamflow (Poff *et al.*, 1997; Clausen and Biggs, 1998). This has been coupled with a strong desire among environmental hydrologists to utilise streamflow records for managing streams as an ecological resource (Clausen and Biggs, 2000), since long-term monitoring and assessment of river ecosystem response to different streamflow regimes is rare. As a result, there is a plethora of hydrological indices describing the ecologically important characteristics of the seasonal patterns of streamflows, including the timing of extreme flows; the frequency, predictability and duration of floods, droughts and intermittent flows; daily, seasonal and annual flow variability; and rates of change (Poff *et al.*, 1997). Since no single variable is likely to represent all the processes that support the life stages of instream biota, several researchers have reviewed the choice of hydrological indices in order to provide researchers with a framework for the selection of indices for eco-hydrological studies, while minimising redundancy in any analysis without the loss of information (Clausen and Biggs, 2000; Olden and Poff, 2003).

Understanding the complex ecological functioning of aquatic ecosystems at different scales requires high-resolution information on flow characteristics throughout, and between, years. Analysing hydrological variables derived from daily mean streamflows rather than monthly mean streamflows is more likely to reveal details of low and high flow events, both of which have important roles in structuring aquatic ecosystems (Poff and Ward, 1989). Recently, Olden and Poff (2003) reviewed a comprehensive list of 171 currently available hydrological indices (including the Indicators of Hydrologic Alteration (Richter *et al.*, 1996) developed in the United States of America) using long-term daily hydrological records from 420 sites from across the USA. Their study revealed that many of the ecologically relevant hydrological indices currently available are highly correlated and that it is possible to identify groups of "high information, non-redundant" hydrological indices for different streamflow types in different geographical regions (Olden and Poff, 2003).

However, many of the hydrological indices appraised as being ecologically relevant have been identified on the basis of research carried out in temperate climatic regions, and for perennial rivers. Furthermore, the statistical analysis of long-term daily streamflow records is meaningful only when calculated for a sufficiently long hydrological record. It is generally accepted that longer datasets may be required for arid regions than for wetter regions to adequately represent the streamflow patterns that would occur under natural conditions (Joubert and Hurly, 1994). Consequently, those indices identified as relevant in temperate climatic regions may not be as appropriate to describe the variability associated with less "predictable" streamflow regimes associated with more arid climates. Different indices may therefore be more useful for describing the streamflow characteristics associated with more arid conditions.

As discussed in Chapter 4, EFAs in South Africa range from comprehensive specialist studies, *e.g.* the Building Block Methodology (BBM) described by King and Tharme (1994), the Flow-Stress Response (FS-R) approach developed by O'Keeffe *et al.* (2002) and Downstream Response to Imposed Flow Transformations (DRIFT) developed by Brown and King (2000), to "desktop" estimates of the flow requirements of unsurveyed rivers based on the results of the more detailed BBM studies. With the pressure to define the Ecological Reserve for South Africa's river systems prior to the issues of licenses to water users, there is a case for exploring whether desktop determinations of environmental flow requirements (EFRs) can be enhanced and, consequently, provide increased usefulness to a greater diversity of stakeholders.

In this Chapter, a minimum subset (or subsets) of hydrological indices which adequately represents the different facets of the streamflow regimes found in South Africa is reviewed. Statistical analysis is applied to long-term daily streamflow records held by the Department of Water Affairs and Forestry (DWAF) using a multivariable approach to investigate the inter-relationships among 74 streamflow variables. The suitability of these indices for application in eco-hydrological studies and for assisting in determining environmental flow requirements is examined for river systems in different hydro-geographical regions in South Africa. In addition, the length of record necessary to obtain consistent hydrological indices, with minimal influence of climatic variation, is investigated.

2 CLASSIFYING RIVER SYSTEMS

2.1 Introduction

The variability associated with streamflow regimes is utilised by many stakeholders to operate and develop a range of economic activities and societal welfare. It is, therefore, pertinent that stakeholders are aware of the impacts of a variety of activities and utilisation of freshwater ecosystem goods and services. First, a frame of reference of the "natural" streamflow patterns and ecologically relevant indices of the different kinds of river systems in South Africa is required to measure the impacts of human activities on streamflow regimes and for sustainable development of the nation's water resources.

2.2 Hydrological Classification Systems for River Ecosystems

It has been argued that every river has a characteristic "signature" representing its streamflow regime and biotic community (King and Schael, 2001). Consequently, the concept of classifying the heterogeneous characteristics of different river systems into some order of homogenous groups is a tenuous exercise for many researchers and stakeholders. Nonetheless, it is useful to clarify similarities in streamflow patterns, at both temporal and spatial resolutions, so that the likely responses of the hydrological regime to a wide range of human activities across catchments can be predicted.

Grouping similar rivers based on *hydrological indicators* or attributes is an attractive river management concept since hydrological indicators are more easily measured than geomorphic, biotic or anthropogenic indicators (*c.f.* Chapter 2 of this thesis, Section 2.4.4). Hydrological classifications of river systems range from "top-down" approaches in which streamflow patterns are used to define the similarities of different systems, to "bottom-up" approaches in which the distribution records of biota are used to measure similarities among rivers (O'Keeffe and Uys, 2000). Top-down approaches such as those proposed by Richter *et al.* (1996) and Poff *et al.* (1997) have been criticised for their inherent assumptions that *different streamflow components sufficiently represent a variety of biophysical processes that are related to ecological response*. Bottom-up approaches such as the South African developed BBM (King and Tharme, 1994) described in Chapter 4, Section 3 which (comprehensively) assess the streamflows needed to sustain the habitat requirements of a variety of aquatic species have been criticised with regard to the *uncertainty of ecological response to hydrological functioning, across a diversity of different river systems*, largely as a result of incomplete datasets.

The ecological and societal relevance of Ecological Reserve determinations is described in Chapter 4 of this Thesis. However, it is pertinent to emphasise that in addition to a *quantitative assessment* of the Ecological Reserve to meet environmental flow requirements, current methods of Reserve determinations embrace a *qualitative classification* for both the present and the desired future state or Ecological Reserve Category of the river systems that exist in South Africa (*c.f.* Table 4.2 of Chapter 4). This approach is intended to allow stakeholders to assess the "cost" of utilising certain goods and services to the integrity of the ecosystem. However, it is still necessary to quantify any alteration that could be expected to the streamflow regime, not only for ecosystem health, but also for competing downstream users.

2.3 Hydrological Characteristics of Streamflow Regimes

Natural streamflow regimes represent a continuum of energy flows determined by water and sediment patterns as well as by chemical and biological processes, which result from catchment characteristics of climate, vegetation, geology, slope and other features of the landscape (*c.f.* Figure 5.2). Recognising the relationships among the hydro-geomorphic landscape, the streamflow regime, habitat dynamics and aquatic ecosystems has allowed researchers to develop river classifications based on physical features and dynamic processes (Naiman *et al.*, 2002). The underlying premise of such classifications is that the streamflow regime is a major determinant of the physical habitat in river and floodplain ecosystems, which in turn is a major determinant of biotic composition (Resh *et al.*, 1988). The influence of the streamflow regime in shaping and maintaining channel morphology and a diversity of riverine habitats is well documented (*e.g.* Hynes, 1970; Vannote *et al.*, 1980; Junk *et al.*, 1989; King and Tharme, 1994). There is, however, increasing recognition of the role of the natural streamflow regime in maintaining aquatic biodiversity (Poff and Ward, 1989; Poff *et al.*, 1997) and the focus of this Section is the identification of hydrological indices which can be used to assess, monitor and promote biodiversity within aquatic ecosystems.

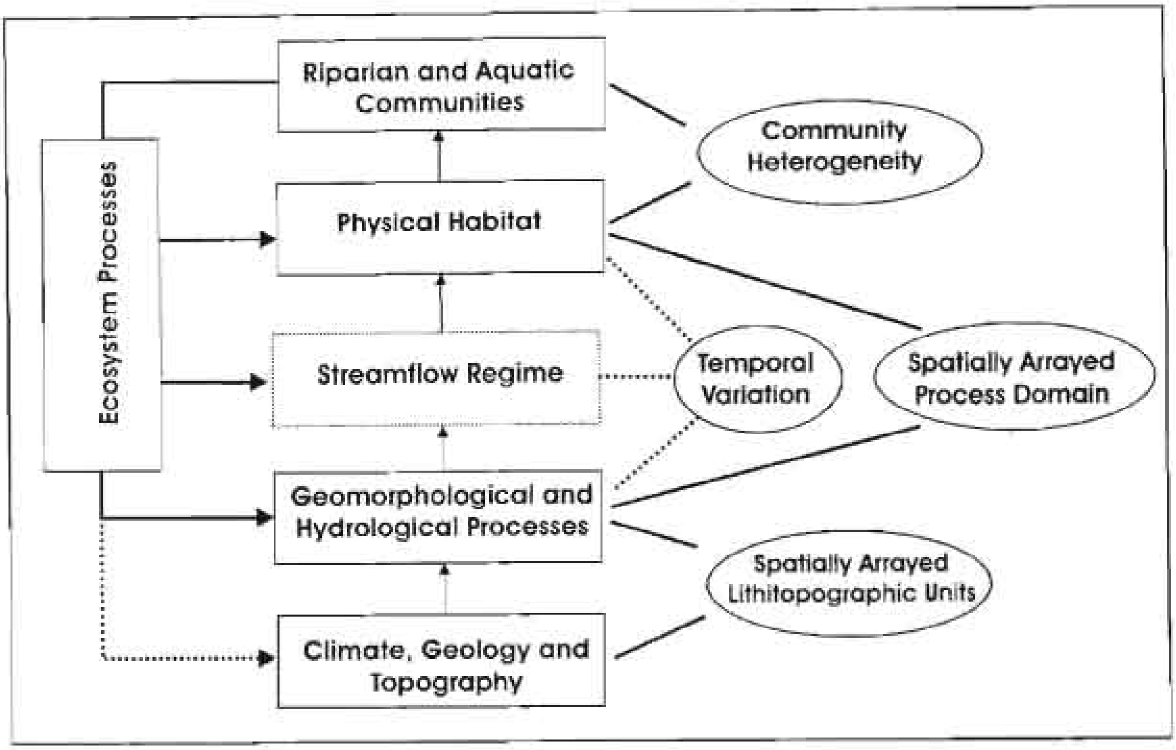

Figure 5.2 Representation of the temporal and spatial linkages among the hydro-geomorphic landscape, the streamflow regime, habitat dynamics and aquatic ecosystems; broken lines indicating modification of the schematic described by Naiman *et al.* (2002)

Figure 5.3 shows how the natural streamflow regime influences aquatic biodiversity and ecosystem functioning in several interconnected mechanisms (Bunn and Arthington, 2002). For example, intra-annual variability in the streamflow regime contributes to biodiversity by influencing the seasonality and predictability of the overall "flow pattern", triggering a variety of abiotic mechanisms including sediment deposition or mobilisation, as well as changes in temperature and chemical composition (Bunn and Arthington, 2002). In turn, abiotic mechanisms act as environmental triggers for the different life stages of aquatic biota. Inter-annual variability in the streamflow regime contributes to biodiversity by influencing spatial heterogeneity in the overall habitat and is needed to trigger dispersal mechanisms and out-of channel connectivity, including riparian seedling redistribution and nutrient exchange between floodplains and river channels (Figure 5.3). More pertinently, aquatic biota have evolved in response to the overall flow pattern of magnitude, duration, timing, frequency and rates of change of the streamflow regime (Bunn and Arthington, 2002).

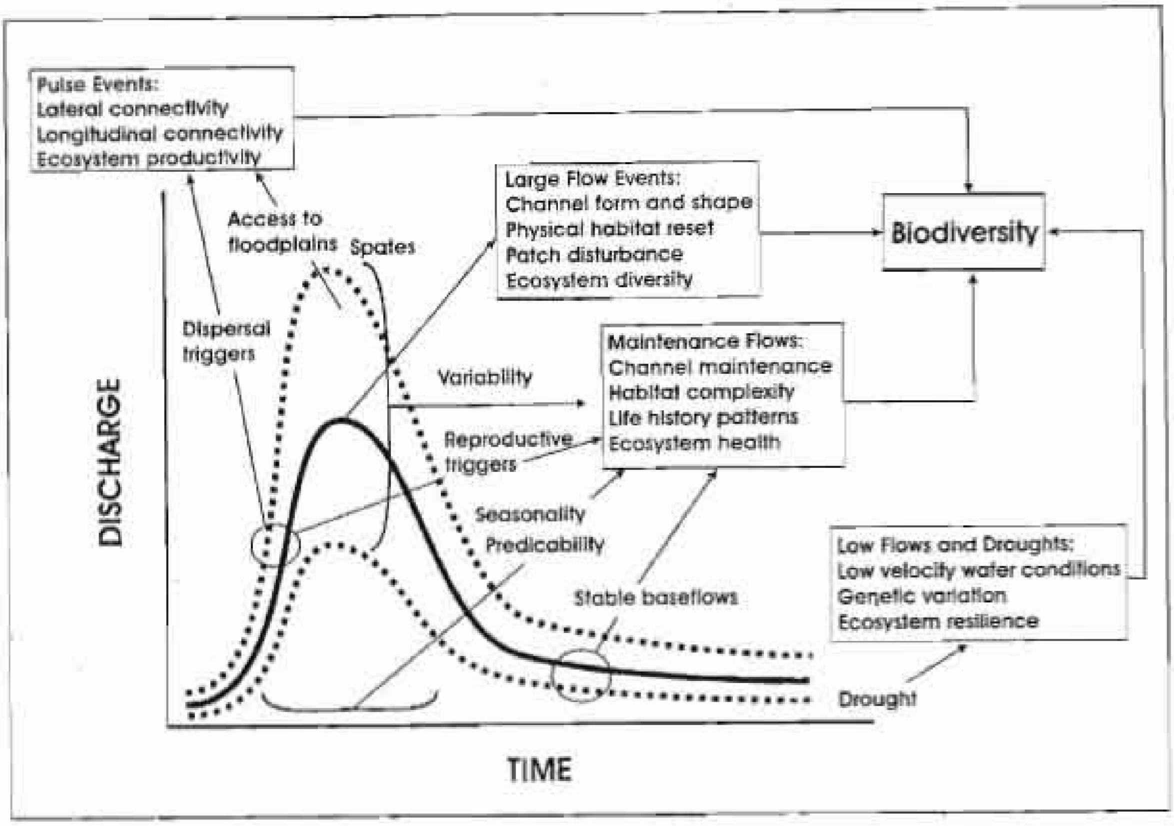

Figure 5.3 The influence of the natural streamflow regime on aquatic biodiversity (modified from Bunn and Arthington, 2002)

There are many hydrological indices which can be extracted from long-term streamflow records. Poff (1996) defines four general categories of hydrological indices that could be used to characterise streamflow regimes. The following sub-sections describe these categories and their relevance to aquatic ecosystem biodiversity.

2.3.1 General hydrological indices

General catchment descriptors such as *elevation*, *basin drainage area*, *daily mean discharge* and *mean annual runoff* are often included in datasets of environmental variables for aquatic studies. Poff (1996) describes these variables as *static basin descriptors* since they do not contain any information about hydrological variability, but rather describe the size of the river (Joubert and Hurly, 1994). Hydrological indices which describe streamflow variability, in terms of seasonality and the predictability of flow, the degree of cessation of flow and the recurrence intervals of flooding are of much greater interest to ecologists when assessing the response of aquatic biota to streamflow patterns (Joubert and Hurly, 1994).

2.3.2 General streamflow variability

Measures of *averages* of streamflow data, at daily, weekly, monthly, seasonal or annual time steps, do not provide sufficient information for the assessment of EFRs. Streamflow is naturally variable and information is also needed about the variation within data samples. Poff (1996) describes two general categories of hydrological variables, *viz. streamflow variability* and *streamflow predictability*, to assess the degree of variation in streamflow. Streamflow variability in magnitude, timing, duration, frequency and rates of change between rising and falling river levels, as well as the predictability of both high and low flow events, provide the template for ecological processes. Thus, streamflow variability and predictability are the foundation of any theory of aquatic ecosystem functioning (Naiman *et al.*, 2002) and have been the basis for fundamental ecological principles to guide understanding of aquatic ecosystem response to physical processes, at multiple scales (Poff and Ward, 1989; Richter *et al.*, 1996).

The *variability* within streamflow regimes can be described by the Coefficient of Variation, CV, at various temporal resolutions. The CV of any streamflow parameter is defined as the ratio of the standard deviation (the most widely applied measure of variability, and calculated directly from all observations of a particular parameter) to the mean of the data, and is used routinely in the quantitative analyses of variability within and between samples (Fowler and Cohen, 1990). However, hydrological data representative of climatic conditions in Southern Africa are often skewed by extreme events. Non-parametric methods employ more resistant statistics around the central tendency and dispersion (*i.e.* the median and Coefficient of Dispersion, CD) rather than the mean and variance of data and are therefore more appropriate to the analysis of streamflows of "highly variable" river systems where skewness of the data can present problems. Moreover, samples of data which comprise units of counts can never, in theory, possess the mathematical properties of the normal curve (Fowler and Cohen, 1990). Thus, non-parametric methods, where ranks rather than actual observations are used, are frequently more suitable than parametric methods for processing data which are counts or derived data, *e.g.* proportions or indices (Fowler and Cohen, 1990).

A *predictability* index was developed by Colwell (1974) which can be used to express the degree to which ecological data is predictably distributed across specified time intervals

(Poff, 1996). "Colwell's Predictability" is essentially a measure of the variation (or invariance) among successive periods; "when variation is low, predictability is high" (Colwell, 1974). Colwell (1974) defined predictability as comprising two independent components: *constancy*, which varies inversely with the amplitude of variations and is, therefore, a measure of temporal invariance, and *contingency*, as an inverse measure of "persistence". Consequently, constancy is the component of predictability resulting from streamflows which remain similar, or constant, throughout the year, whereas contingency is the component of predictability resulting from a predictable seasonal regime (Joubert and Hurly, 1994). Poff (1996) provides the example that "a stream with a discharge that never varies would be perfectly predictable", with all the predictability being due to constancy. On the other hand, "a stream with fluctuating discharge could also be completely predictable due to perfect contingency", since the streamflow changes with certainty on a daily basis (Poff, 1996). Measurement of predictability, constancy and contingency involves separating the streamflow data into different categories of flow states, based on data value. Gan *et al.* (1991) found that Colwell's Index of Predictability tends to stabilise with record lengths in excess of 20 years. However, Poff (1996, p 88) found that "the index is also sensitive to temporal resolution in the hydrological time series and that relative differences among streams may change depending on the time scale of analysis" and that "relative differences among streams for Colwell's Index can vary depending on the way one defines the flow states (*e.g.* the number of flow categories)". Thus, Poff (1996, p 88) concluded that "the Coefficient of Variation is a more straightforward, less arbitrary measure of streamflow variability that shows more stability with respect to length of the time series than does Colwell's predictability index".

2.3.3 Extreme events or disturbance in streamflow regimes

Hydrological extremes are recognised for the constraints, or disturbance, they present to aquatic ecosystem communities. The disturbance regime, comprising both high flow and low flow disturbance is an integral facilitator of the mechanisms that contribute to aquatic biodiversity. Yet, the temporal resolution in hydrological data required to describe these extremes is unclear (Poff, 1996).

High flow disturbance is best defined in terms of those streamflows of magnitude which exceed a specified flood frequency. However, the annual peak flow series, which provides the maximum instantaneous flow values, rather than records of daily average flow (Poff, 1996), are required to determine these flood values and probability of occurrence. Records of such "partial flow series" may be unavailable or inadequate for many regions. However, many useful indicators of high flow conditions can be extracted from records of daily average flow rates, including:

(a) Flow *magnitude* exceeding a specified percentile of daily flow values across all years of record and the *frequency* of such flow magnitudes. These indices relate to the pulsing nature of the flow regime. For example, at the highest section of the Flow Duration Curve these indices are a measure of the occurrence of floods which not only scour the channel, but also inundate the floodplain. Inundation of the floodplain is the mechanism whereby nutrients are recycled and energy flows between different components of the landscape (*c.f.* Figure 1.1). For many rivers there is a strong positive correlation between species richness and connectivity with the channel bank and floodplain suggesting that recurrent flooding is vital to aquatic biodiversity (Galat *et al.*, 1998).

(b) The magnitudes of annual maxima flow at 1-day, multi-day, or seasonal resolutions are an indication of the "energy" and dynamic nature of the river system. For example, the median over the period of record, of the annual maxima of daily flow values corresponds fairly well with the 1:2 year recurrence interval for floods (Poff, 1996) and could provide a coarse indicator of bankfull discharge, since a [peak] flow with a 1:2 year return period is globally recognised as "bankfull" for many regions and climates. However, the *recurrence interval* of bankfull discharge is acknowledged as being between two and five years in South Africa, with "2.3 years being an acceptable average" (Joubert and Hurly, 1994).

(c) The *timing* of the annual maxima flow influences biotic response. For example, riparian plants are sensitive to both increased *frequency* and changes in the *timing* of water-level fluctuations (Naiman *et al.*, 2002).

(d) The *seasonal predictability of flooding*, defined as the maximum proportion of all floods over the period of record that occur within a given 60 day period or "seasonal" window (Poff and Ward, 1989), and

(e) The *seasonal predictability of non-flooding* defined as the length of the flood-free season (Poff and Ward, 1989). Many aquatic biota have evolved life history patterns which are triggered by either of these seasonal events and patterns.

Low flow disturbance is a natural seasonal phenomenon, the hydrological drought being distinct from those low flows exacerbated by anthropogenic interference. Smakhtin and Watkins (1997) conducted a comprehensive review of low flow estimation in South Africa, wherein they describe low flow analysis from streamflow records as including flow duration analyses, analysis of frequency, magnitude and duration of continuous low flow events, low-flow frequency analysis and analysis of baseflow recessions.. In addition, Smakhtin (2001) provides a comprehensive review on low flow characteristics, their derivation from streamflow records and the application of low flow indices in ecological assessments. Indices of low flow disturbance include the following:

(a) Low flow indices related to extended periods of zero flow (the *degree of cessation of flow*) characterise seasonality and distinguish intermittent and ephemeral streamflow regimes from streams with perennial flow regimes (Smakhtin, 2001).

(b) At the low flow section of the Flow Duration Curve (typically the flow discharge equaled or exceeded 75% of the time Q75 – Q99) these indices can be a measure of drought or stress conditions for aquatic ecosystems, a threshold below which flows should not be maintained habitually for extended periods. For example, Q75 may be correlated with drought indices (Smakhtin, 2001). In some instances, the flows at certain percentages of exceedence have been correlated with inflection points (or threshold values) in channel morphology, and have been linked to the physical state at which ecological processes are initiated or accelerate (*c.f.* Section 2.4.3 of this Chapter).

(c) The magnitudes of annual minima at *multi-day or seasonal* resolutions represent indicators of the extent of hydrological drought. In a dry season, these indices characterise the release of water from storage via hydrological processes that link the river channel with subsurface flows and groundwater sources. Thus, these low flow indices describe the "persistence" of the streamflow regime.

(d) In addition, the *timing* of the lowest annual minima of daily flows also contains information relating to the seasonality of the streamflow regime.

(e) Indices of the *recurrence* (or *frequency*) of low flow magnitudes describe the severity of low flow disturbance in the regime. In natural streamflow regimes these indices may be regarded as measures of biotic intra- and inter-specific competition for habitat and resources and are essential indicators of aquatic biodiversity.

2.4 Ecologically Relevant Hydrological Indices

Ecologically relevant hydrological indices are useful for describing the consequences of altered streamflow regimes for biotic diversity and the *ecological integrity, health* or *vitality* of the ecosystem (Richter *et al*., 1996; 1997). These different ecosystem states are described more fully in Chapter 2 of this thesis and are not repeated here, save to emphasise the links between the natural environment and human culture. "*Ecologically relevant*" may be a spurious term in that ultimately the "*relevance*" has a human endpoint in the delivery of ecosystem goods and services. In a major review paper, Olden and Poff (2003) cross-referenced the ecological basis for 171 hydrological indices of streamflow regimes based on literature on aquatic ecology, citing 13 major eco-hydrological publications. The ecological basis of the Olden and Poff study was the selection of indices "which were derived to represent biologically relevant streamflow attributes" (Olden and Poff, 2003). Critics of the use of hydrological indicators, and in particular statistical attributes thereof, or "hydrological indices", to set ecosystem-based management targets as measures of ecological functioning often cite that the indices have not been rigorously tested. While such arguments are plausible, statistical measurements of "reference conditions" such as the frequency by which flows of a particular magnitude are exceeded, or expected to occur, are easily understood by a range of diverse stakeholders.

2.4.1 Readily computed hydrological indices

Understanding the ecological functioning of aquatic ecosystems requires high-resolution information on streamflow characteristics throughout and between years, as well as knowledge of the manner in which these characteristics affect conditions for aquatic biota (Joubert and Hurly, 1994). Hydrological indices derived from daily mean flows have been used increasingly in eco-hydrological studies, since records of daily mean streamflows are more likely than monthly mean streamflows to reveal details of low and high flow events, both of which have important roles in structuring aquatic ecosystems (Poff and Ward, 1989). The use of hydrological indices in EFAs has evolved from the early "historical" methods of focusing on average flow conditions (Tennant, 1976 and *c.f.* Chapter 3, Section 2) to the examination of suites of hydrological indices concurrently (Poff and Ward; 1989; Joubert and Hurly, 1994, Richter *et al*., 1996; 1997; Clausen and Biggs, 2000; Hughes and Hannart, 2003).

Olden and Poff (2003) examined a large data set of hydrological indices (171) for redundancy, so that a reduced number of indices could be selected for characterising streamflow regimes in the USA while still adequately describing the available information provided by the dataset. Minimum subsets of ecologically relevant indices for different river types are attractive options for water resource managers and stakeholders alike, especially if they can be incorporated into desktop methods of EFAs. Computing processing power has enhanced the generation of hydrological indices, or suites of indices, from daily streamflow records, for assessing the different characteristics of the streamflow regime (Richter *et al.*, 2003). Some of the more relevant and readily available indices, in terms of the current study, are described in the following subsections.

2.4.2 The Indicators of Hydrologic Alteration

The United States Conservancy developed the Indicators of Hydrologic Alteration (IHA) as a method of characterising the natural range of streamflow variability, or the extent to which streamflows have been altered by human activity, based on the statistical analysis of daily streamflow data (Richter *et al.*, 1996; 1997; 1998). The IHA method was mentioned in Chapters 3 and 4. However, it is appropriate to include a fuller description of the method in this Section. The statistical analysis in the method referred to is performed using the IHA software (Smythe Scientific Software, Boulder, Colorado, USA) designed to compute statistical information relating to the hydrological indices derived from files containing daily streamflow records.

The IHA method consists of a statistical analysis of 33 ecologically relevant hydrological parameters which characterise intra-annual variability in the streamflow regime. The method computes inter-annual statistics and produces 33 indices of central tendency and 33 indices of dispersion to characterise variability in the streamflow regime. The IHA indices represent five groups of streamflow characteristics (*c.f.* Table 4.5 in Chapter 4 of this thesis) which can be attributed to playing major roles in determining the nature of aquatic and riparian ecosystems (Poff and Ward, 1989). Poff and Ward (1989) as well as Richter *et al.* (1996) define the five streamflow characteristics as the *magnitude* of water condition; *timing, frequency* and *duration* of occurrence as well as *the rate of change* as a measure of how quickly streamflows rise and fall between consecutive days. Groups 1 and 5 (*c.f.* Table 4.5 in Chapter 4) are measures of the magnitude or rate of change of *average* flow conditions, whereas Groups 2, 3 and 4 (*c.f.* Table 4.5 in Chapter 4) focus on the magnitude, duration, timing and frequency of *extreme events*. The following sections summarise the descriptions of the five groups, as given by Richter *et al.* (1996; 1997).

(a) The *magnitude of the monthly means of daily flows* (Group 1) represents average daily flow conditions for the month and indicates the general amount of flow required for habitat availability and suitability for each month. The degree to which monthly means vary from month to month indicates the intra-annual variation of streamflow conditions, whereas the extent to which flows vary within any given month, but from year to year (the CV, or CD, of monthly means of flow), indicates the inter-annual variation of streamflow conditions.

(b) The *magnitude and duration of extreme annual conditions* (Group 2) are a measure of the different environmental disturbances, or stresses, which can occur throughout the year. The durations were selected by the developers of the IHA to represent natural or human induced cycles and comprise the 1-day, 3-day, 7-day (weekly), 30-day (monthly) and 90-day (seasonal) extremes. The 1-day events are the maximum and minimum daily streamflow values that occur in any given year and the multi-day events are the highest and lowest multi-day means of flow occurring in any given year. The number of zero-flow days, characteristic of non-perennial rivers, is also included in this group. The developers of the IHA define "baseflow" conditions as the 7-day annual minimum flow divided by the annual mean of daily flows. The inter-annual variation in the magnitude of these extreme conditions influences the extent to which environmental variation occurs within ecosystems.

(c) The Julian date of the 1-day maximum and minimum flow events represents the *timing of the annual extreme conditions* (Group 3) within annual cycles and provides a measure of the seasonal nature of environmental stresses or disturbances. The timing of these flows can influence the life cycles of aquatic organisms. The inter-annual variation in the timing of these extremes also influences the extent to which environmental variation occurs within ecosystems, but may be of greater significance in more temperate regimes where snow-melts consistently influence the daily maximum flow and also significantly reduce water temperatures and increase dissolved oxygen concentration. Alternatively, the timing of cyclones in tropical regions may be environmentally significant.

(d) The *frequency of conditions* during which the magnitude of streamflows exceeds an upper threshold or falls below a lower threshold within an annual cycle, and the *average duration of such occurrences* (Group 4) together reflect the pulsing behaviour of the streamflow regime within a given year. Richter *et al.* (1996; 1997) generally define high pulses as those periods within a year when the daily streamflow rises above the 75th percentile of all daily values across the record, and low pulses as those periods within a year when the daily streamflow falls below the 25th percentile of all daily values across the record. However, these thresholds can be altered by the user of the IHA software.

(e) The *rate and frequency of change in conditions* (Group 5) measure the number and average rate of both positive and negative changes in streamflow between consecutive days. These changes in the hydrograph indicate the intra-annual fluctuation of the streamflow regime.

2.4.3 Desktop indices for South African rivers

A significant amount of work has already been conducted to identify similarities within hydrological regimes in South African river systems in order to permit extrapolations of streamflow patterns from assessed to unassessed rivers within the same geographical region.

In 1994 Joubert and Hurly (1994) sought a comprehensive, ecologically relevant classification of naturally flowing rivers in South Africa, based on temporal and spatial hydrological variation (the "1994-Study"). The 1994-Study applied Department of Water Affairs and Forestry (DWAF) records of daily streamflow data from catchments assumed to be relatively unimpacted by human activities and identified groups of streamflow type and seasonality based on streamflow characteristics. The 1994-Study is described in more detail in Sections 3 and 4 of this Chapter. However, the overall aim of the 1994-Study was a classification that would be useful to water resources managers in the assessment of environmental flows. As such, the 1994-Study comprised a useful contribution to the key thrust of South African water resources management during the 1990s, namely, the assessment of environmental flows for river systems in the region.

EFA in South African water resources management has evolved from the assessment of the specific instream flow requirements (IFRs) of different biotic and abiotic components to embrace holistic and comprehensive evaluations of the streamflow regimes required to deliver the ecosystem goods and services that catchment stakeholders need, for the present as well as future use. The aims, merits and challenges of such assessments for the sustainable utilisation of the nation's water resources, and the classification thereof, are described in other Chapters of this thesis. It is, however, pertinent to note that ecological investigations of the comprehensively determined evaluations of the ER (*c.f., inter alia*, Chapter 4 of this thesis) by Hughes *et al.* (1998) as well as Hughes and Hannart (2003) have identified at least four ecologically relevant hydrological indices that are readily quantifiable from available streamflow time series. The first index, Q75 (*c.f.* Section 2.3.3 of this Chapter), is derived from analysis of daily streamflow records, whereas the remaining three indices, BFI, CV and CVB (see below for descriptions), are derived from the full natural monthly time series.

EFAs conducted using the comprehensive BBM (*c.f.* Chapter 4, Section 3) have indicated some correlation between Q75 on flow duration curves of daily streamflows across records and assessments of "drought low flow requirements" for many South African river systems (Hughes *et al.*, 1998). The regression relationships with Q75 and the proportion of the time that a river experiences zero flow were used to calculate a "baseflow index" in the original version of the Desktop Reserve model (Hughes and Hannart, 2003) following guidelines presented by Smakhtin and Watkins (1997).

Hughes and Hannart (2003) describe two measures of hydrological variability that are readily derived from long-term records of streamflows. The first is a "monthly-based coefficient of variation" index representing the *long-term variability* associated with the cycle of wet and dry seasons and is calculated from the CV for all monthly flows across the record for each calendar month. The average CVs for the three main months of both the wet and dry seasons are combined to produce a *composite* CV of the two seasonal averages (Hughes and Hannart, 2003).

The second is a "baseflow" index (BFI) developed to represent *short-term variability* in the hydrograph. Hughes *et al.* (2003) describe baseflow separation on the full natural monthly time series using the following technique:

$$q_m = \alpha q_{m-1} + \beta(1+\alpha)(Q_m - Q_{m-1}) \qquad \text{(Equation 1)}$$

$$QB_m = Q_m - q_m \qquad \text{(Equation 2)}$$

where Q = total flow

q_m = high flow component

QB_m = baseflow component

m = month and

α, β = separation factors

Previously, Smakhtin and Watkins (1997) and Smakhtin (2001) had applied the separation model to time series of daily and monthly streamflow data respectively and identified fixed values for the parameters α and β for a large number of observed flow time series in South Africa. In a different study, Hughes *et al.* (2003) used the model to compare the results of continuous baseflow separation from time series of daily streamflow data with time series of monthly streamflow data using data from 70 observed daily streamflow records. Comparisons between the daily and monthly separations, highlighted the shortcomings inherent in the model for time series of one day or longer, but produced improved regionalised values for α and β, for the whole of South Africa, when using the more readily available monthly data (*i.e.* the WR 90 simulated data of Midgley *et al.*, 1994).

It is important to mention that the baseflow separation model applied in the Desktop Reserve model focuses on the low amplitude, high frequency response in the monthly hydrograph (Hughes and Hannart, 2003) rather than associating the baseflow component of the streamflow regime with any catchment runoff generation process or origins of water sources (*c.f.* Chapter 4, Section 3.4). The BFI is the ratio between the mean annual baseflow and the mean annual total flow (*i.e.* the average discharge under the separated baseflow hydrograph and the average discharge of the total hydrograph respectively, Smakhtin, 2001). The approach has been applied to produce a BFI with values for South Africa's 1946 Quaternary Catchments, for use in the South African Reserve model, SARES, the national water balance model that is applied in strategic water resources planning and in Desktop Reserve model.

Hughes and Hannart (2003) combined the two complementary indices in a composite expression of CV and BFI to form CV/BFI as a representation of overall variability (CVB).

In addition to those studies emanating directly from research for BBM indices, South Africa's long-term daily streamflow records, and in particular the database of naturally flowing rivers identified by the 1994-Study, have been applied in a variety of projects to derive ecologically relevant hydrological indices for classifying the requirements of variable river systems. Two additional studies are described here, since they emphasise the usefulness of the establishment of a working database of reliable long-term hydrological records for ecohydrological studies.

(a) Hughes (1997) sought to identify a rule-based model for baseflow recession in South Africa, based on the streamflow response to catchment and climatic characteristics. The research established master baseflow recession curves, using daily flow records, for the streamflow regimes at 134 DWAF gauging sites representing "relatively natural streamflows". The aim of the research was to group catchments with similar baseflow recession responses. However, Hughes (1997) concluded that further research was needed for the establishment of a rule-based model for baseflow recession analysis in South Africa.

(b) Hollands (1998) selected 11 daily flow records of relatively undisturbed streams in the south-western Cape for identification of an inflection point at the lower end of the wetted perimeter-discharge curve. The aim of the study was to link the discharge at which the loss of the wetted perimeter accelerates as the wetted channel shrinks from the banks (Wetted Bed Flow, WBF), with hydrological indices which could be derived from daily streamflow records. The results of the study indicated that a WBF could be assessed from general, or relatively coarse, hydrological indices such as mean annual runoff, percentile of the daily flow duration curve and the 1:2 year maximum average daily flow, to provide information on critically low discharges at unmonitored sites.

The Downstream Response to Imposed Transformations (DRIFT) method developed by Brown and King (2000) for EFA in southern Africa also uses hydrological statistics to describe a variety of different ecologically relevant river levels. The method utilises the link between the streamflow regime and exposure, or inundation, of ecologically relevant river channel features by focusing on streamflow magnitude (ranges of volume), frequency (recurrence of specific streamflow magnitudes), timing (in terms of seasonality) and duration (through the construction of flow duration curves). DRIFT uses an ecological definition of low flows, *viz.*, "the flow residing in the river outside of the high flow events", (Brown and King, 2000) and, consequently, is somewhat similar to the hydrological definition of baseflow described above. The method separates the hydrological time series into its "low flow" and "high flow" elements before deriving the hydrological statistics and setting:

(a) *ranges* of flows for each low flow season

(b) *the percentage of time that any particular low flow is equaled or exceeded* (from flow duration curves) and

(c) the *duration*, temporal distribution, average number per year and average volume of different-size classes of floods, derived from the high flow time series.

Reductions in the range of flows or of flood events are equated with reductions in *streamflow variability*, which, in turn, are linked with the socio-economic impacts of the biophysical response to reductions in specific streamflow regime components. The DRIFT method classifies streamflow variability with hydrological statistics derived from daily flow records. However, the separation of the time series into *ecologically defined* low flow and high flow components before the hydrological statistics are calculated, and the need for specialist consultation to divide the low flow time series into a dry and a wet low flow season, clouds the identification of any particular DRIFT "desktop indicators" of variability that are distinct from those described in previous literature.

2.5 Linking Hydrological Indices with Ecosystem Response

The role of streamflow on aquatic and riparian ecosystem processes has highlighted the potential of the natural hydrograph in defining EFRs. One of the key quests is to identify the most influential aspects of the streamflow regime for structuring aquatic communities (Clausen and Biggs, 1998). Hydrological variability has been associated with increased habitat and food web complexity (Thoms and Sheldon, 1997) and indices that address hydrological variability have, for a variety of reasons, gained a good deal of attention in water resources management in recent years. Indeed, the most promising currently available flow assessment methods construct environmental flow regimes that mimic at least some of the natural intra- and inter-annual variability of hydrological regimes. The methodology selected for prescribing environmental flows (*c.f.* Chapter 3, Section 2) depends on the availability of ecological, hydrological and hydraulic data; expertise in the assessment of biophysical conditions; skill in integrating local knowledge, capacity and culture as well as time to integrate the data and information. However, few studies have such a wealth of resources to hand and emerging trends in EFA indicate a focus towards links between streamflow characteristics and ecological response, *e.g.* the South African BBM (King and Tharme 1994) and FS-R (O'Keeffe *et al.*, 2002) methods (*c.f.* Chapter 3, Section 2 and Chapter 4; Section 3). Moreover, there has been a resurgence of interest in the application of hydrological indices to predict the likely ecological response to streamflow characteristics (Clausen and Biggs, 1997; 1998), with a strong desire among environmental hydrologists to utilise streamflow records, as an ecological resource, for managing streams (Clausen and Biggs, 2000; Olden and Poff, 2003).

2.6 Summary

The natural streamflow regime is a major driver of aquatic biodiversity. Classification of different river systems, based on hydrological indices of the natural variability of the streamflow regime, is valuable to water resources management. Ecologically relevant hydrological indices of the full range of natural streamflow variability, including average and extreme flow conditions, are useful for measuring the impacts of human activities on ecosystem health and on downstream users. Readily computed hydrological indices which describe streamflow variability are attractive options for water resource management, especially if they can be incorporated into desktop methods of EFAs.

* * * *

While the streamflow regime can describe the temporal environmental variability in streams, it does not provide information on the contribution to the habitat template of equally important spatial heterogeneity (Poff and Ward, 1990; Poff, 1996). Interest in comparing hydrological similarity among river systems has grown internationally in the past decade (Joubert and Hurly, 1994; Poff, 1996; Clausen and Biggs 1998; Clausen and Biggs, 2000; Olden and Poff, 2003). This type of study is especially relevant where the sensitivity of hydrological indices under different flow regime types is unknown and stakeholders are required to "build" modified streamflow regimes to generate required ecosystem goods and services. The following Sections describe an investigation of these uncertainties for different river systems located in South Africa. First, Section 3 revisits the research conducted in 1994 by Joubert and Hurly and describes the methods applied in this Study to update and augment the 1994-Study.

3 DEVELOPMENT OF THE DATABASE OF DWAF GAUGING STATIONS RECORDING "RESEASONABLY NATURAL" STREAMFLOWS

3.1 Introduction

Information relating to the natural intra- and inter-annual variability, as well as the predictability, of flows which characterise a river is valuable to river ecologists and water resource managers (Joubert and Hurly, 1994). However, long-term, detailed studies of the likely response of aquatic biota to different flow conditions are rare for most regions in South Africa. Yet, long-term hydrological records represent a historical source of ecological information which is useful for the classification of different flow regimes in South African rivers.

This Section first summarises previous research undertaken to classify South African rivers using daily streamflow records and, thereafter, describes the approach and methods used in the current Study to extend those earlier studies.

3.2 Previous Research Using Daily Streamflow Data to Classify South African Rivers

The present Study builds on previous research (the 1994-Study) undertaken in the early 1990s. During the 1994-Study, Joubert and Hurly (1994) identified a database of Department of Water Affairs and Forestry (DWAF) gauging stations recording streamflows which they considered to be "most representative of natural flow conditions". Joubert and Hurly (1994) selected the DWAF gauging stations on the basis that:

(a) the gauging sites were upstream of all known major impoundments or abstractions,

(b) there was a minimum span of 20 years of recorded flows from which daily mean flows could be obtained and

(c) there were no observable trends in the data.

Joubert and Hurly (1994) addressed the last criterion through visual assessment of double mass plots of cumulative monthly rainfall representative of catchment conditions with monthly streamflows for each of the gauging stations.

The focus of the 1994-Study was an investigation of the daily flow records held by DWAF, in order to determine whether they could be used to group or classify, either by geographical region or by flow type, the diversity of rivers found in South Africa. A further aim of the 1994-Study was to identify the streamflow characteristics that distinguished each group and to assess the appropriateness of hydrological variables representing these characteristics for use by river ecologists and water resource managers as "ecologically relevant hydrological indicators" of water conditions.

3.2.1 Grouping river flow patterns

The methods and results of the 1994-Study are described in detail in King and Tharme (1994). In essence, Joubert and Hurly (1994) categorised the streamflow characteristics of South African rivers by flow type (*i.e.* based on hydrological similarity) using four general variables, being;

(a) the *intra-annual coefficient of variation*, a non-temporal measure of overall streamflow variability, derived by taking the mean over the years of record of the intra-annual coefficients of variation. This variable measures within-year variability (*i.e.* the change from wet to dry seasons) and not variability between years.

(b) the number of *days of zero flow* in a year, and

(c) the predictability of flow in terms of *constancy* and *contingency* (Colwell, 1974).

In addition to these four general variables they also derived four flood variables, being the *predictability, frequency* and *duration* of floods, and the *interval* between floods.

Their choice of these eight hydrological variables for grouping rivers by flow type was influenced by research undertaken in 1989 by Poff and Ward (1989) which had highlighted the ecological importance of these variables for different "streamflow types" in the USA. Joubert and Hurly (1994) followed two different approaches and methods of analysis, both conducted on a database of 204 DWAF gauging stations with daily streamflow records with a minimum of 10 years of data. The average record length among the 204 DWAF stations was 20 years. The aim of the first method (Method One) was to group and characterise rivers using variables and techniques that had been applied elsewhere, *e.g.* Poff and Ward (1989), and comprised non-hierarchical cluster analyses and stepwise discriminant analysis to derive the groups. The aim of the second method (Method Two) was to investigate new techniques which allowed boundaries to be formed by the decisions of the researcher rather than the software used. Method Two comprised covariance biplot, or correspondence analysis techniques, where the stations were represented on a series of two-dimensional plots to summarise the information contained in all the hydrological variables in as few dimensions as possible. Thus, the aims of the two methods were different and, consequently, the groupings reached by the methods were very different. Method One produced eight groups which were considered homogenous to the dominant variables for each group. Method Two produced ten groups (contained within three super-groups), which were considered homogenous to the frequency and duration of flood events. While Method One produced results that were reproducible, Method Two was based on the researcher's intuition and the results were unlikely to be reproducible (Joubert and Hurly, 1994).

In their conclusions, Joubert and Hurly identified several shortcomings of their study, including the use of different years of daily recorded flow data for different sites (which subsequently influenced the "groupings" of flow type) and the definition of flood variables. Joubert and Hurly also recommended the use of medians rather than means to define some of the variables describing flow characteristics.

3.2.2 Extrapolating river flow types from common hydrological indices

The main benefit to water resource development of a grouping "natural" streamflow patterns according to ecologically relevant hydrological indices is the opportunity of extrapolating information from gauged sites to ungauged sites. This is particularly pertinent where stakeholder decisions are needed for rivers which have not been monitored in terms of biota, channel morphology or hydraulics. If the differences among streamflow types can be described by a few, high information (Olden and Poff, 2003) and readily understood hydrological indices, stakeholders will be in a better position to attribute the costs and benefits of their water use patterns. High resolution (daily) hydrological data and information on "natural", or reference, conditions are essential for assessing the response of biota to changes in sediment and water flow patterns. Thus, extrapolating the characteristics of a streamflow type, based on common hydrological indices that are readily computable from long-term records of daily flows to unknown, yet hydro-geographically similar, situations could provide general recommendations on modified flow patterns for any one streamflow type (Joubert and Hurly, 1994).

3.3 Extending the Database

In 1995 the School of Bioresources Engineering and Environmental Hydrology (BEEH) at the University of Natal (now KwaZulu-Natal) in Pietermaritzburg obtained a database of the "best200" gauging weirs (*c.f.* stations listed in Appendix 5A, Table 5.A1) from A. R. Joubert, based on the results of the 1994-Study. Although this database has been used in a number of research studies within the School, it had not been updated since 1996. Consequently, it was necessary to review the data from the "best200" gauging weirs to ensure that their extended records of daily streamflow data could still be considered of "good or better" quality (Joubert and Hurly, 1994) and that their representation of "natural flow conditions" was reasonable.

One shortcoming of the 1994-Study was that the records assessed varied in length (from 11 to 54 years). Not only does this have implications for the stability of the statistical moments associated with the hydrological variable being analysed, but time series spanning different years are subject to different general climate patterns.

In addition, in the 1994-Study, the DWAF records were screened for non-homogeneity of daily streamflow data by using pairs of rainfall and streamflow gauging records (*c.f.* Section 3.2 of this Chapter). The streamflows recorded at each streamflow gauging station were matched with an appropriate catchment "driver" rainfall gauging station (Dent *et al.*, 1989) to create double mass plots of cumulative monthly streamflow against monthly rainfall representative of each station. The plots were checked by visual assessment to identify points of deflections from general trends. Only where divergences in the plots occurred, was the gauging station, or the data after the break, eliminated from the study.

The test of homogeneity of time series of streamflows applied in the 1994-Study was reviewed for the present Study, since the practice of a *visual assessment* of any break in the slope of cumulated plots of monthly streamflows against rainfall by double-mass analysis was considered to be tenuous for a number of reasons, in accordance with Dahmen and Hall, 1990) *viz*:

(a) Double-mass analysis assumes a linear relationship between the time series of data, yet the relationship between streamflow and rainfall is not linear. Antecedent soil moisture conditions and seepage from various hydrological partitions influence catchment runoff processes and streamflow response times to rainfall events.

(b) Time series of *monthly totals* of streamflow from rivers with considerable "persistence" of groundwater response from one month to the next, are unlikely to be independent, *i.e.* there is correlation between successive observations.

(c) In double mass analysis it is necessary to evaluate the *significance of a change in the slope* of a double mass line to ensure that the probability of an abrupt change does not occur by chance.

(d) Double mass analysis verifies *relative* consistency and homogeneity and is not suitable for testing the stationarity of a time series.

It was considered that while the visual assessment of relative homogeneity of rainfall to streamflow plots in the 1994-Study provided an indication of general trends, a more robust assessment of the stationarity and consistency of the records represented by the stations in the "best200" database was required for the present Study.

3.3.1 Methods of selection of the DWAF streamflow records used in the present study

The records of the stations in the "best200" database were screened using the procedure shown in Figure 5.4. The screening procedure outlined in this Study was adapted from a procedure devised by Dahmen and Hall (1990) to test annual or seasonal series for absence of trend and the stability of the variance and mean. The Spearman Rank-Correlation method applied by Dahmen and Hall (1990) to test for absence of trend is useful to this Study since it is based on ranking of the data, thereby making it appropriate for application with data which are not normally distributed (*i.e.* skewed). However, it was considered that analyses of the data for variability within each of the parameters and associated with

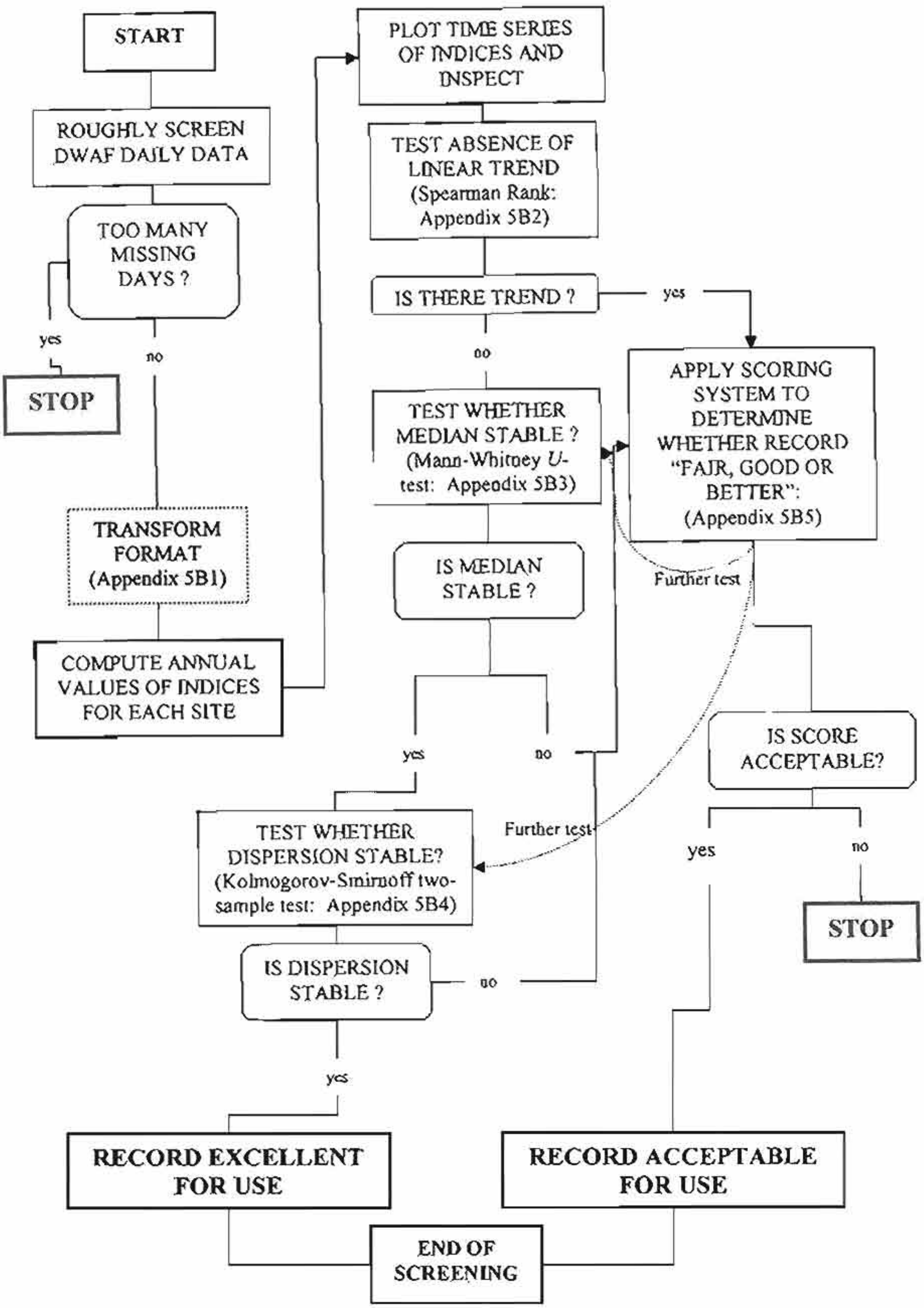

Figure 5.4 Screening procedure for the selection of DWAF streamflow records used in the multivariable approach to identify high-information hydrological indices of the streamflow regimes found in South Africa (modified from Dahmen and Hall, 1990)

linear regression (*i.e.* the mean and variance), as applied by Dahmen and Hall (1990), would be less useful in any test of stationarity in time series of streamflows from "highly variable" river systems. Therefore, the tests for stability of the mean and variance applied by Dahmen and Hall (1990) were replaced with non-parametric statistical tests, employing the median and dispersion, which are statistically more robust. Details of the tests shown in Figure 5.4 are discussed in Appendix 5B, which is referred to, where appropriate, in the following text.

Data relating to the DWAF stream gauging network is available via the World Wide Web. The DWAF site (http://www.dwaf.gov.za/hydrology/cgi-bin/his/cgihis.exe/station) was accessed between 1 September 2003 and 1 December 2003 to obtain information on the availability of daily average flow rates ($m^3.s^{-1}$) for each of the recording stations comprising the "best200" database. First, the full time span of data available for each station (*c.f.* Appendix 5A, Table 5.A1) was assessed to ascertain the longest common time period among the maximum number of recording stations. In addition to the "best200" stations, information relating to gauging station U1H005 was retrieved (*c.f.* Appendix 5A, Table 5.A1), since a separate study had shown this site to be recording "reasonably natural" flow conditions (Taylor, 2001). A common time period was sought in order that any analysis undertaken in the present Study would be made on streamflow regimes that had resulted from similar general climatic patterns. The DWAF introduced automation to the recording of daily streamflows at its gauging stations in 1960. Thus, most effort focused on identifying the longest common time period from the 1959 hydrological year onwards. Daily data for records spanning the hydrological years 1959 to 2001 (the most up-to-date available at the commencement of this Study) were extracted from the DWAF web site for initial screening of completeness and quality of records. The DWAF continuously updates and reviews the coding of streamflow records. The quality codes describe whether the data are *inter alia*, continuous, edited, above the rating limit, unknown, either a minimum or maximum value, estimated, permanently or temporarily missing or unreliable.

Six of the DWAF stations in the "best200" database (*i.e.* B5H002, C3H003, D1H003, D1H009, D2H001 and D3H003) have upstream catchment areas in excess of 10 000 km^2. These stations record flows from catchments that are much larger than the others (*c.f.* Appendix 5A, Table 5.A1) and the streamflow characteristics at these stations are likely to be influenced by dissimilar catchment processes and hydrological mechanisms. It was, therefore, decided to eliminate these records from the research undertaken in this Study.

At the time of this Study, the longest common time period available for the maximum number of potentially useful records was the 36 year period from 1 October 1965 to 30 September 2001. Ninety-eight streamflow records from stations in the "best200" database (denoted with an asterisk in Appendix 5A, Table 5.A1), were assessed to identify those with predominantly good continuous data over this time period. Streamflow records with missing data were not rejected outright since it was considered that a partial record could be analysed, or that the "missing data" would not detract from the Study if it could be reasonably infilled (*c.f.* Figure 5.4 and Appendix 5B1). Nonetheless, streamflow records found to have several years of missing data (*e.g.* at B7H010, B9H001, C5H012, E2H002, J2H005, J2H006, J2H007, L6H001, Q9H002 and U1H006) or many years of missing data (*e.g.* R2H005, T1H004 and T5H002) were rejected (*c.f.* Figure 5.4 and Appendix 5A, Table 5.A1). As a result, 85 DWAF stations (Table 5.1) remained for assessment with the next steps of the screening procedure, as outlined in Figure 5.4. The following Sections summarise the screening procedure. Details of the approach, hypothesis, statistical tests and power to reject a false hypothesis at each stage of the investigation, as outlined in Figure 5.4, are described in Appendix 5B, which is referred to where appropriate.

Table 5.1 DWAF gauging records with predominantly good, continuous data for the period 1 October 1965 to 30 September 2000

A2H029	C8H003	K3H004	T2H002	V1H001	X2H010
A2H032	D5H003	K4H002	T3H004	V1H009	X2H012
A4H002	G1H008	K4H003	T3H008	V1H010	X2H013
A4H005	G1H009	K5H002	T3H009	V3H002	X2H014
A4H008	G1H010	K6H001	T4H001	V3H007	X2H015
A5H004	G1H011	K7H001	T5H003	V3H009	X2H022
A9H003	G1H012	K8H001	T5H004	V6H003	X2H024
A9H004	G4H006	K8H002	U1H005	V6H004	X3H001
B1H002	G5H008	L8H001	U2H006	V7H012	X3H002
B1H004	H1H007	R1H014	U2H007	W1H004	X3H003
B4H005	H1H013	R2H001	U2H011	W4H004	
B6H001	H3H005	R2H006	U2H012	W5H006	
B6H003	H7H004	R2H008	U2H013	W5H008	
C5H007	J4H003	S3H006	U4H002	X2H005	
C7H003	K3H002	S6H003	U7H007	X2H008	

First, the records of daily average streamflow for the 85 gauging sites (Table 5.1) were extracted from the DWAF retrieval website for the 36 year period from 1 October 1965 to 30 September 2001 and the DWAF file format was transformed for further investigation using the method described in Appendix 5B1.

3.3.2 Computing the hydrological indices

A total of 74 hydrological indices, derived to represent ecologically relevant streamflow characteristics (*c.f.* Section 2.4 of this Chapter), was examined to identify high information, non-redundant indices (Olden and Poff, 2003) of the river systems found in South Africa, *viz*:

(a) The 66 Indicators of Hydrological Alteration, consisting of:
- *33 hydrological indices* of central tendency (in this Study, medians) as well as
- *33 hydrological indices of dispersion*, (in this Study, CDs) described by Richter *et al.*, (1996; 1997);

(b) Eight hydrological indices of the streamflow regime that apply to the time period as a whole, comprising the following:
- *Predictability* of flow and the
- *Proportion* of predictability attributed to constancy (both derived using Colwell's predictability index, predictability = constancy + contingency, Colwell (1974)
- Percentage of floods that occur during a given 60 day period in all years, as a measure of the *seasonality of flooding* and
- Length of the flood-free season, as a measure of the *seasonality of non-flooding* (both described by Poff and Ward (1989)
- An "alternative" *Baseflow Index*, Alt-BFI as a measure of short-term variability, and an alternative to the baseflow index included in the IHA suite
- CDB, a combination of the Alt-BFI and a composite Coefficient of Dispersion, CD, associated with the average CDs for the three main months of both the wet and dry seasons as a measure of *overall variability* (both modified from, Hughes and Hannart, 2003)
- HFI, a *high flow index*, also a measure of short-term variability and

- Q75, the simple *low flow index* often applied in initial water resource assessments.

Descriptions of the indices, their derivation from daily mean flow records and reference in the literature are provided in Table 5.2. The 74 indices were grouped in the categories discussed by Olden and Poff (2003), following Richter *et al*. (1996) and Poff *et al.* (1997). These comprised the magnitude (n = 30), duration (n = 27), timing (n = 7), frequency (n = 4) and rate of change (n = 6). In addition, magnitudes were further separated into average (n = 25), high (n = 1) and low (n = 4) categories; duration into high (n = 13) and low (n = 14) categories; timing into average (n = 2), high (n = 3) and low (n = 2) categories and frequency into high (n = 2) and low (n = 2) categories to produce a total of 11 subcategories to describe the different characteristics of the streamflow regime, following Olden and Poff (2003). The symbol notation for the indices used in this thesis also follows that of Olden and Poff (2003). Indices of both the central tendency and dispersion were grouped within the subcategories, following Olden and Poff (2003), except that in this Study medians and coefficients of dispersion were used to describe the statistical moments of the distribution of hydrological indices across the time series. For example, in this Study, both the index for the low pulse count, and the coefficient of dispersion thereof, were included in the subcategory for "low frequency" and annotated F_L1 and F_L2 respectively. The 74 indices were calculated using a combination of the IHA software (Smythe Scientific Software, Boulder, Colorado, USA), the IHA output files, Microsoft Excel and Visual Basic programming language (Microsoft Officesuite2000, Microsoft Corp., USA). The IHA software directly computed all but three of the indices, *viz.* the Alt-BFI, the CDB and the HFI. Nonetheless, some of the files generated by the IHA software were also used to compute these indices.

Table 5.2 Hydrological indices used in the Study, their derivation and source of reference, with symbol notation for indices as in Olden and Poff (2003)

Symbol	Unit	Definition	Reference
Magnitude of flow events			
Average flow conditions			
M_A1	$m^3.s^{-1}$	Mean monthly flow for October	
M_A2	$m^3.s^{-1}$	Mean monthly flow for November	
M_A3	$m^3.s^{-1}$	Mean monthly flow for December	
M_A4	$m^3.s^{-1}$	Mean monthly flow for January	
M_A5	$m^3.s^{-1}$	Mean monthly flow for February	

M_A6	$m^3.s^{-1}$	Mean monthly flow for March	Richter *et al.* (1996; 1997)
M_A7	$m^3.s^{-1}$	Mean monthly flow for April	
M_A8	$m^3.s^{-1}$	Mean monthly flow for May	
M_A9	$m^3.s^{-1}$	Mean monthly flow for June	
M_A10	$m^3.s^{-1}$	Mean monthly flow for July	
M_A11	$m^3.s^{-1}$	Mean monthly flow for August	
M_A12	$m^3.s^{-1}$	Mean monthly flow for September	
M_A13	-	Coefficient of Dispersion of M_A1, (*i.e.* difference between 75th percentile of values and 25th percentile of values divided by the median of values across all years of record)	
M_A14	-	Coefficient of Dispersion of M_A2	
M_A15	-	Coefficient of Dispersion of M_A3	
M_A16	-	Coefficient of Dispersion of M_A4	
M_A17	-	Coefficient of Dispersion of M_A5	
M_A18	-	Coefficient of Dispersion of M_A6	
M_A19	-	Coefficient of Dispersion of M_A7	
M_A20	-	Coefficient of Dispersion of M_A8	
M_A21	-	Coefficient of Dispersion of M_A9	
M_A22	-	Coefficient of Dispersion of M_A10	
M_A23	-	Coefficient of Dispersion of M_A11	
M_A24	-	Coefficient of Dispersion of M_A12	
M_A25	-	Ratio of seasonal variability to baseflow (CDB)	Hughes and Hannart (2003)
Low flow conditions			
M_L1	-	7-day annual minimum flow divided by mean daily flow for year	Richter *et al.*, (1998)
M_L2	-	Coefficient of Dispersion in M_L1	
M_L3	-	Ratio of baseflow volume to total volume (Alt-BFI)	Hughes and Hannart (2003)
M_L4	$m^3.s^{-1}$	Q75, the 25th percentile of flow values across the record	
High flow conditions			
M_H1	-	Median of annual maximum flows (HFI)	Olden and Poff (2003)
Frequency of flow events			
Low flow conditions			
F_L1	$year^{1-}$	Low pulse count (*i.e.* number of annual occurrences during which the magnitude of flows is below a lower threshold. Low flow pulses are those periods within a year when flow is less than the 25th percentile of all daily values across the record)	Richter *et al.* (1996; 1997)
F_L2	-	Coefficient of Dispersion in F_L1	
High flow conditions			
F_H1	$year^{1-}$	High pulse count (*i.e.* number of annual occurrences during which the magnitude of flows is above a higher threshold. High flow pulses are those periods within a year when flow is greater than the 75th percentile of all daily values across the record)	Richter *et al.* (1996; 1997)

F_H2	-	Coefficient of Dispersion in F_H1	
Duration of flow events			
Low flow conditions			
D_L1	$m^3.s^{-1}$	Annual minimum 1-day average flow	
D_L2	$m^3.s^{-1}$	Annual minimum 3-day average flow	
D_L3	$m^3.s^{-1}$	Annual minimum 7-day average flow	
D_L4	$m^3.s^{-1}$	Annual minimum 30-day average flow	
D_L5	$m^3.s^{-1}$	Annual minimum 90-day average flow	
D_L6	$Year^{-1}$	Average annual number of days having zero daily flow	
D_L7	days	Average duration of F_L1	
D_L8	-	Coefficient of Dispersion in D_L1	Richter *et al.* (1996; 1997)
D_L9	-	Coefficient of Dispersion in D_L2	
D_L10	-	Coefficient of Dispersion in D_L3	
D_L11	-	Coefficient of Dispersion in D_L4	
D_L12	-	Coefficient of Dispersion in D_L5	
D_L13	-	Coefficient of Dispersion in D_L6	
D_L14	-	Coefficient of Dispersion in D_L7	
High flow conditions			
D_H1	$m^3.s^{-1}$	Annual maximum 1-day average flow	
D_H2	$m^3.s^{-1}$	Annual maximum 3-day average flow	
D_H3	$m^3.s^{-1}$	Annual maximum 7-day average flow	
D_H4	$m^3.s^{-1}$	Annual maximum 30-day average flow	
D_H5	$m^3.s^{-1}$	Annual maximum 90-day average flow	
D_H6	days	Average duration of F_H1	
D_H7	-	Coefficient of Dispersion in D_H1	Richter *et al.* (1996; 1997)
D_H8	-	Coefficient of Dispersion in D_H2	
D_H9	-	Coefficient of Dispersion in D_H3	
D_H10	-	Coefficient of Dispersion in D_H4	
D_H11	-	Coefficient of Dispersion in D_H5	
D_H12	-	Coefficient of Dispersion in D_H6	
D_H13	days	Average annual maximum number of days in a water year during which no floods occur across the period of record	Poff and Ward, 1989; Poff (1996)
Timing of flow events			
Average flow conditions			
T_A1	-	Predictability of flow, comprising two independent components, *viz.* constancy (i.e. a measure of temporal invariance) and contingency (i.e. a measure of periodicity)	Richter *et al.* (1996; 1997)
T_A2	-	Proportion of Predictability due to Constancy	
Low flow conditions			
T_L1	-	Average Julian date of the 1-day minimum flow over period of record	Richter *et al.* (1996; 1997)
T_L2	-	Coefficient of Dispersion in T_L1	
High flow conditions			

T_H1	-	Average Julian date of the 1-day maximum flow over period of record	Richter *et al.* (1996; 1997)
T_H2	-	Coefficient of Dispersion in T_H1	
T_H3	-	Maximum proportion of all floods over the period of record that fall within any 60-day period	Poff and Ward, 1989; Poff (1996)
Rate of change in flow events			
Average flow conditions			
R_A1	$m^3.s^{-1}.d^{-1}$	Average rate of positive changes in flow from one day to the next	Richter *et al.* (1996; 1997)
R_A2	$m^3.s^{-1}.d^{-1}$	Average rate of negative changes in flow from one day to the next	
R_A3	-	Average number of negative and positive changes in water conditions from one day to the next	
R_A4	-	Coefficient of Dispersion in R_A1	
R_A5	-	Coefficient of Dispersion in R_A2	
R_A6	-	Coefficient of Dispersion in R_A3	

The Alt-BFI was derived following an approach similar to the calculation of the BFI applied in the Desktop Reserve model (*c.f.* Section 2.4.3 of this Chapter). Annual values of the baseflow component of the time series for each of the 85 recording stations were derived using the calendar month parameters (Table 5.2) provided in the IHA annual summary output files. A monthly time series of streamflows over the 36 year period was generated in Microsoft Excel and a baseflow separation carried out for each of the 85 recording stations using the model described by Hughes *et al.* (2003) and Hughes *et al.* (2003) and described in Section 2.4.3 of this Chapter. Regionalised values for the separation parameters α and β for the whole of South Africa were provided by Professor Denis Hughes at the Institute for Water Research (IWR), at Rhodes University in Grahamstown (Hughes, 2004; pers. com.). The values of the separation parameters used in the computation of the Alt-BFI for each site used in this Study are shown in Table 5.3. The Alt-BFI applied in this Study was the median of annual values of the proportion of baseflow to total flow. This calculation differs slightly from the traditional BFI calculation of the mean annual baseflow to the mean annual total flow. The reason for this deviation from the norm was to maintain the non-parametric approach to the analysis of the inter-annual statistics adopted in this Study. Further, while the BFIs applied in the Desktop Reserve model are available for each of the South Africa's 1946 Quaternary Catchments, their values are based on a study of baseflow parameters by Hughes *et al.* (2003) of observed streamflow records with different time steps and for different record lengths to the 36 year time series (1965-2000) applied in this Study. Nonetheless, the values of the

Alt-BFI derived for this Study were compared with the BFI values applied in the Desktop Reserve model and provided by Professor Denis Hughes from the Institute of Water Research at Rhodes University in Grahamstown to ensure that there were no marked discrepancies (Figure 5.5). For most stations the difference was less than 10% (*c.f.* Figure 5.5a). Even where the difference was greater, this was not appreciably so (as shown in 5.5b). Nonetheless, the *derivation* of the Alt-BFI was considered to be acceptable. With the exception of the gauging stations located in primary catchment A, the extent of the difference between the values of the Alt-BFI and the values of the Desktop Reserve model BFI is random. This confirms that the main differences are attributable to (a) the different time steps applied to (b) different lengths of hydrological record and to (c) the different calculation of the index as a result of the different statistical measures applied.

Table 5.3 Values of the baseflow parameters, α and β (provided by Professor Denis Hughes from the Institute for Water Research, Rhodes University, Grahamstown), applied in the calculation of the Alt-BFI used in this Study

Region	**DWAF Station**	**Parameters**	
		α	β
Western Cape (wet)	G1H008; G1H009; G1H010; G1H011; H1H007; H1H007	0.950	0.42
Western Cape (dry)	G2H012; G4H006	0.970	0.42
Western Karoo	H3H005	0.995	0.42
Eastern Karoo	D5H003; H7H004	0.990	0.44
Southern Cape (dry)	G5H008; J4H003	0.970	0.42
Southern Karoo	L8H001	0.990	0.44
Southern Cape (wet)	K3H002; K3H004; K4H002; K4H003; K5H002; K6H001; K7H001; K8H001; K8H002	0.955	0.43
Eastern Cape (arid)	C5H007	0.990	0.44
Eastern Cape	S3H006	0.970	0.44
Amatole	R1H014; R2H001; R2H006;R2H008; S6H003	0.960	0.44
Transkei Region	T3H004; T3H008; T3H009; T5H003; T5H004	0.960	0.43
Transkei Region (Coast)	T2H002; T4H001; U7H007	0.950	0.43
Drakensberg	U1H005	0.950	0.43
Southern Natal	U2H006; U2H007; U2H011; U2H012; U2H013; U4H002; V1H001; V1H009; V1H010; V7H012	0.955	0.43
Northern Natal	V3H002; V3H007; V3H009; V6H003; V6H004; W4H004; W5H006	0.955	0.44
Zululand	W1H004	0.960	0.44
Eastern Escarpment	A9H003; A9H004; B6H001; B6H003; W5H008; X2H005; X2H008; X2H010; X2H012; X2H013; X2H014; X2H015; X2H024; X3H001; X3H002; X3H003	0.960	0.44
Lowveld	A2H032; A4H002; A4H005; A4H008; A5H004; X2H022	0.990	0.45
Vaal	C7H003; C8H003	0.960	0.44
Oliphants	A2H029; B1H002; B1H004; B4H005	0.950	0.44

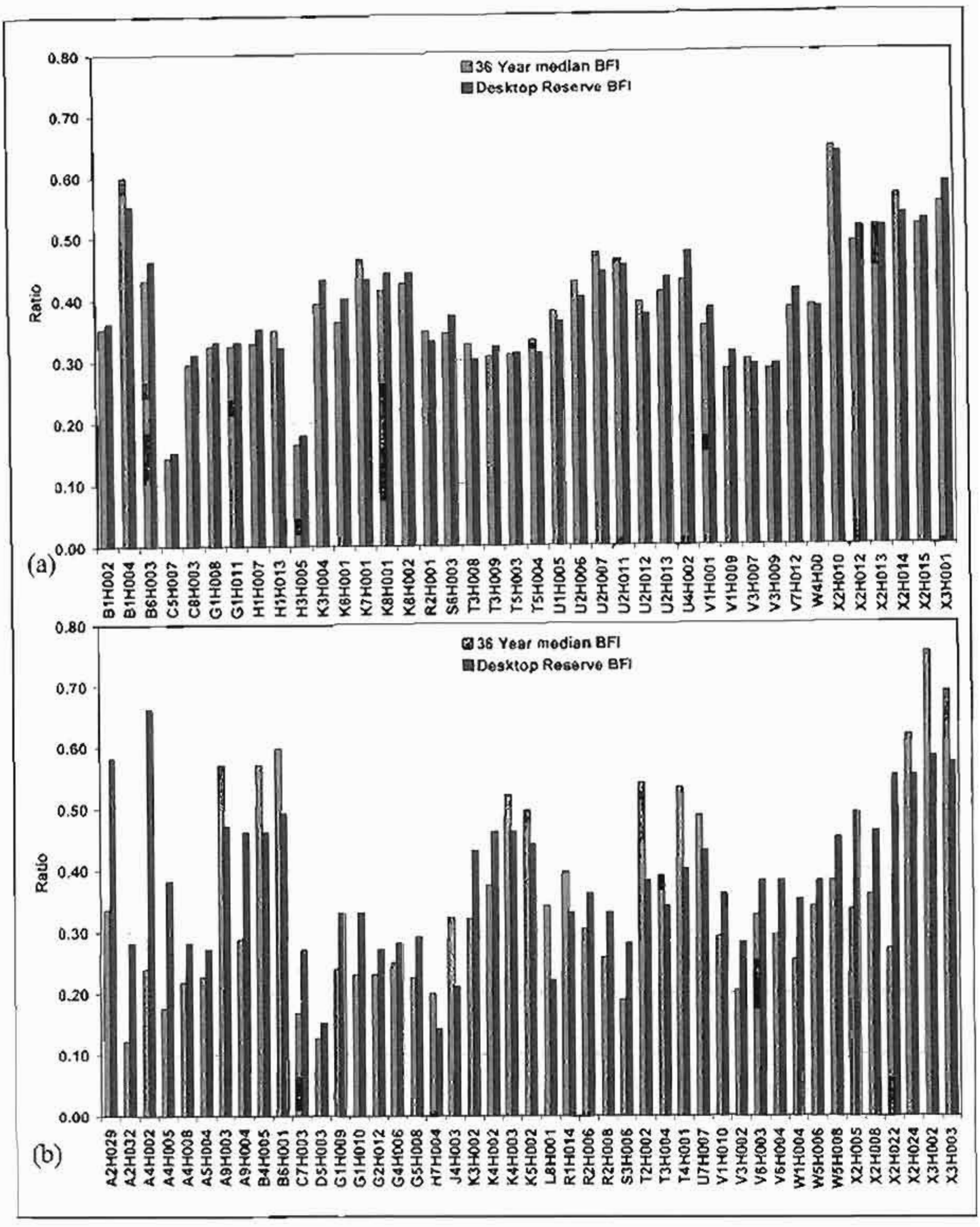

Figure 5.5 Comparison of values (ratios) of the "36 year median" BFI (Alt-BFI) calculated for application in this study with the BFI used in the Desktop Reserve model. Both indices are based on the baseflow separation model described by Hughes *et al.* (2003) and Hughes and Hannart (2003). The main differences between the calculations are the time step, span of time series and the statistics used (parametric *vs* non-parametric). Figure (a) shows stations with ratios which differ by less than 10% and (b) shows the stations with ratios which differ by more than 10%.

The CDB index used in this Study is based on the CVB index (CV/BFI) described by Hughes and Hannart (2003), which they depict as a logical combination of two indices of variability, BFI (representing short-term variability) and a "seasonal" CV (representing the long-term variability of both the wet and dry season flows) to produce an *overall index* of variability. Following the approach outlined in Hughes and Hannart (2003), a "seasonal" CD (Coefficient of Dispersion) was derived, based on the CD of all monthly streamflows for each calendar month, and comprised the sum of the average CDs for the three main months of both the wet and dry seasons. The CDB index was then calculated from the Alt-BFI and the aggregated CD index described above (*i.e.* CDB = aggregated CD/Alt-BFI). Thus, the CDB index in this Study was calculated using non-parametric statistics rather than parametric statistics and for a different time series to that applied by Hughes and Hannart (2003). The CDB for each recording station was derived from the station's Alt-BFI value for the 36 year period and the monthly-based Coefficient of Dispersion for wet and dry months over the time period, and provided in the IHA score output files. The values of the CDB index derived for this Study were compared with the CVB values applied in the Desktop Reserve Determinations and provided by Professor Denis Hughes from the IWR at Rhodes University in Grahamstown (Figure 5.6). For 41 of the stations analysed, the differences were not large (*c.f.* Figure 5.6a). For 44 stations there were some large discrepancies (*i.e.* over 50% difference as shown in Figure 5.6b). However, similarly to the derivation of the Alt-BFI described above relating to the differences in the time step, record lengths and statistical measures applied, together with the combination of the Alt-BFI and of seasonal variability in *derivation* of the CDB index, it was considered that these differences were acceptable.

Annual values of HFI were derived from the 1-day maximum flow value in the IHA annual summary files and from the median daily flow value for each of the 36 years of record, using Microsoft Excel spreadsheets, before ascertaining the median across the record.

The various indices of the intra-annual variability, inter-annual variability, predictability, seasonal predictability and overall variability of the major facets of the streamflow regime described in this Section are summarised in Figure 5.7.

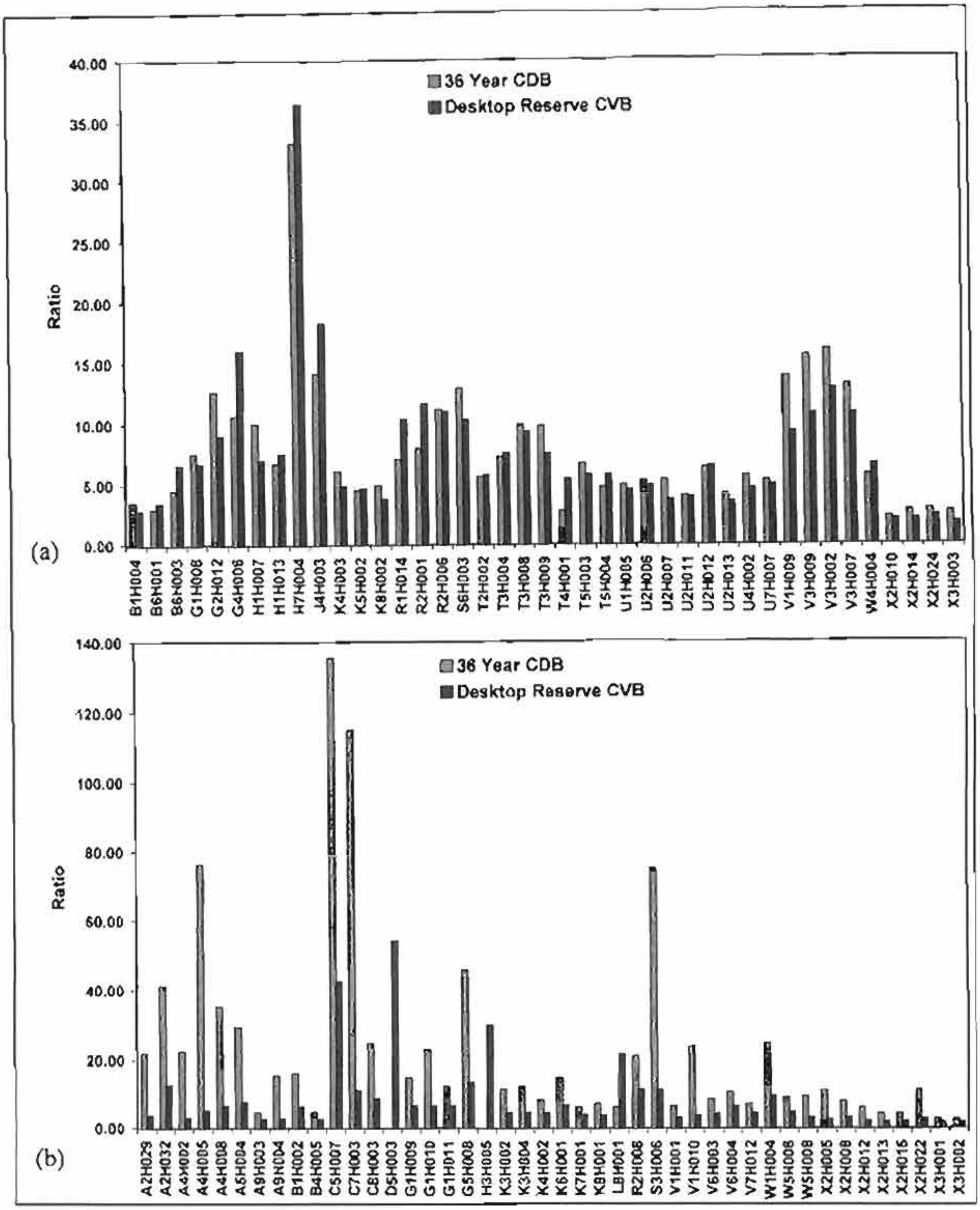

Figure 5.6 Comparison of values of the "36 year median" CDB calculated for this study with the CVB used in the Desktop Reserve model. Both indices are based on a method described by Hughes and Hannart (2003). The main differences are the time step, span of time series and the statistics used (parametric *vs* non-parametric). Figure (a) shows those stations with values which differ by less than 50% and (b) shows the station with values which differ by more than 50%.

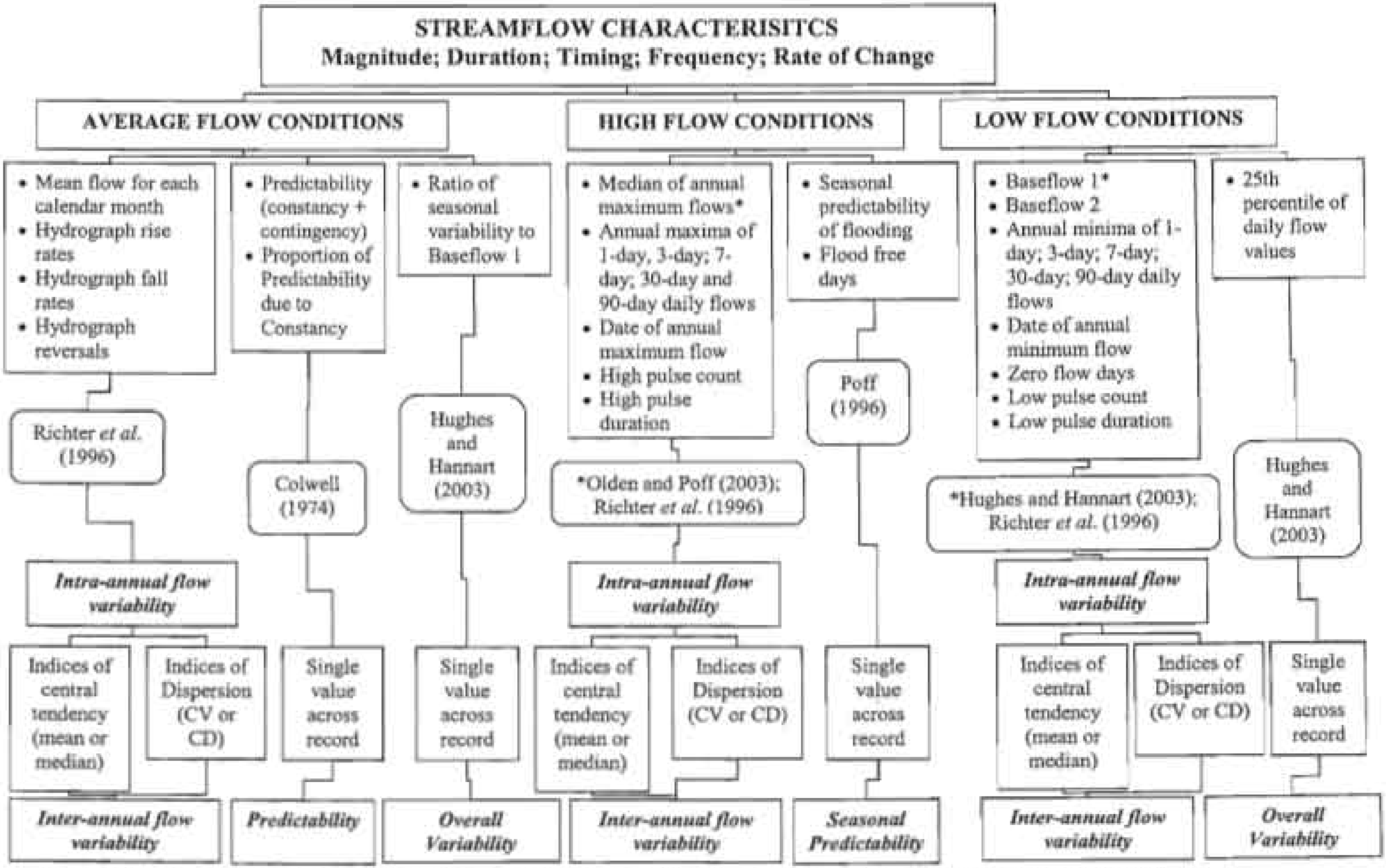

Figure 5.7 Summary of the major characteristics of the streamflow regime and the readily computable indices used in this Study

3.4 Identifying Trends, Stationarity and Homogeneity in the Records

Annual time series of the 33 IHA hydrological parameters of intra-annual variability, the Alt-BFI and the HFI (*i.e.* a total of 35 intra-annual indices) over the 36 year period of record for each of the 85 DWAF sites were first plotted for a visual detection of any long term trends or discontinuities. This initial test was augmented by three more rigorous tests, *i.e.* the Spearman Rank-Correlation method to test for the absence of linear trend in each of the 35 intra-annual indices and two further tests, the Mann-Whitney *U*-test and the Kolmogorov-Smirnoff two-sample test to verify the stationarity, consistency and homogeneity of the indices. The three tests are described in Appendices 5B2, 5B3 and 5B4 for linear trend, stability of the median and of the dispersion respectively. The tests were adapted from the basic screening procedure described by Dahmen and Hall (1990), but were performed using non-parametric rather than parametric statistical tests, because the former are more appropriate to the analysis of streamflows of "highly variable" river systems (*c.f.* Section 3.3.1 of this Chapter). The screening approach is summarised in Figure 5.4. In essence, the three tests were applied in this Study to verify the statistical properties of annual time series of each of the 35 intra-annual hydrological indices over the 36 year record. It is important to note that an underlying assumption was that the rainfall over the sub-continent was also free of trend, was stationary and homogenous for the same time period. This could be considered to be a shortcoming of the screening approach and it is recognised that the tests applied in this Study may not hold any more credibility than the tests adopted by Joubert and Hurly (1994; *c.f.* 3.2.2 of this Chapter). However, since 36 years is a relatively short time span in terms of any climatic change, this assumption was felt not to be unreasonable, given the aim of selecting stations recording "reasonably natural streamflows". Discussions on the efficiency and power (*i.e.* ability to reject a null hypothesis) of the Spearman-Rank Correlation method statistical tests for detecting monotonic trends in time series data may be found in Yue *et al.* (2002). Siegel and Castellan (1988) discuss the power of the Mann-Whitney *U* and Kolmogorov-Smirnoff tests. While their power depends on the pre-assigned significance level, magnitude of trend, sample size and amount of dispersion within a time series, the efficiency of the tests increases with increasing sample size. In this part of the study, the sample size, $n = 36$, was in excess of the "no fewer than 10 observations" recommended by Dahmen and Hall (1990). Dahmen and Hall (1990) also argue that "when the differences in test values are small, it may be of minor practical importance if a test fails to reject the test hypothesis".

An obstacle was encountered when organising the values of the timing of the annual maximum and minimum streamflows for screening, since the Julian dates of extreme flow for many South African rivers are widely spread over the hydrological year (*c.f.* Table 5B1 in Appendix 5B). The developers of the IHA deal with the problem of averaging the date of occurrence of maximum or minimum water conditions by trying to guarantee that the mean (or median) is computed to fall within the quarter of the year with the most occurrences (IHA, 2001). In the IHA software, the dates of maximum and minimum streamflow values are put into quarterly bins and the average is computed from logical arrangements of the quarters. A similar approach was applied in this Study to give more meaningful values to the 36 annual values of these dates before any testing for the absence of linear trend or the variance of the median or dispersion of the data for each station.

The problems surrounding missing days of data, inadequate recording of extreme events (both high and low flow conditions) and inconsistencies in records as a result of faulty equipment, or shortcomings in data retrieval, are well documented throughout the literature. Any such problems result in shortcomings in the reliability, and consequently the acceptability, of streamflow records. The screening procedure adopted in this Study revealed that, depending on the region, some components of the streamflow regime (*e.g.* magnitudes of monthly averages of flows) were more robust over the time series than others (*e.g.* durations of flows) and performed better in the testing procedure. These features are described in greater detail in Section 4 of this Chapter. Nonetheless, a network of streamflow records is extremely important and useful to water resources assessment. Consequently, the long-term records assessed at the 85 DWAF sites were not discounted solely on the failure of the 35 intra-annual indices to meet *some* of the tests described in Appendices 5B2, 5B3 and 5B4. A scoring system was applied to the results of the tests. The details of the scoring system, and the classification of the scores, are described in Appendix 5B5. In essence, scores for the tests of absence of trend, stationarity, consistency and homogeneity of the time series of the each of the 35 intra-annual indices across the record were totaled for *each of the 85 DWAF stations*. The total scores for the stations were then ranked and assigned quality classes (*c.f.* Appendix 5B5) to maximise the number of stations that could be used in the multivariable approach to identify useful hydrological indices of the river systems found in South Africa.

3.5 The Working Database

The screening procedure shown in Figure 5.4 revealed that only two of the 85 DWAF gauging stations (A4H008 and G5H008) have streamflow records which are "excellent" (*i.e.* passing all three tests) for the statistical analysis of *all 35* intra-annual indices in the 36 year time series from 1965 to 2000 (*c.f.* Table 5.4). Twenty-two stations have "very good" scores (*i.e.* passed with between 93% and 99% success). The bulk of the stations (43) have "good" scores (*i.e.* between 72% and 93% success). Sixteen stations have "fair" scores (between 50% and 70% success), whereas two stations (T2H002 and W5H008) had "poor"scores of less than 50% of the target (*i.e.* scores of 58 and 21 respectively out of a possible score of 140) and were not used further in this Study. Thus, the daily streamflow records from 83 DWAF gauging stations were regarded as suitable for statistical analysis of the 74 hydrological indices described in Section 3.3.2 of this Chapter. The names and details of these gauging stations are provided in Table 5.4 and their distribution over South Africa is shown in Figure 5.8. These sites are a subset of the stations identified by the 1994-Study and represent a wide range of streamflow conditions found in South Africa. The 74 hydrological indices of the streamflow regimes over the 36 year time span from 1965 to 2001 at the 83 sites comprise the "working database" of this Study.

3.6 Summary

The DWAF's network of gauging weirs with relatively long records of "reasonably natural" streamflow" represents an important historical and ecological resource for use in eco-hydrological studies of South African rivers. High resolution data and information such as the DWAF daily streamflow records can be used to derive ecologically relevant hydrological indices of streamflow variability and predictability. Classifying the streamflow patterns of South Africa's rivers using ecologically relevant hydrological indices derived from daily streamflow records has been performed in the past. However, previous studies were performed with relatively few indices (*i.e.* no more than eight indices) and across disparate record lengths. This section described the development of a new and relatively large working database comprising 74 ecologically relevant hydrological indices of variability, each derived from a common 36-year period among 83 DWAF daily streamflow records representing different sites of "reasonably natural" streamflow across South Africa.

Table 5.4 Gauging stations comprising the "best83" stations used in this study

Gauging Station	Upstream Area (km^2)	Record Span Start Year*	End Year*	Longitude (degrees, decimal)	Latitude (degrees, decimal)	Weir Site	Acceptability of time series of hydrological parameters (score)
		*hydrological yr					
A2H029	129	1962	2001	28.383	-25.650	Edenvalespruit	Good (127)
A2H032	522	1963	2001	27.017	-25.633	Selonsrivier	Good (120)
A4H002	1777	1948	2000	28.083	-24.267	Mokolorivier	Fair (70)
A4H005	3786	1962	2000	27.767	-24.067	Mokolorivier	Good (108)
A4H008	504	1964	2000	27.967	-24.200	Sterkstroom	Excellent (140)
A5H004	629	1956	2000	28.400	-23.967	Palalarivier	Fair (90)
A9H003	62	1931	2000	30.517	-22.883	Tshinanerivier	Very Good (130)
A9H004	320	1932	2000	30.367	-22.767	Mutalerivier	Good (114)
B1H002	252	1956	2000	29.333	-25.817	Spookspruit	Very Good (136)
B1H004	376	1959	2000	29.167	-25.667	Klipspruit	Good (101)
B4H005	188	1960	2000	30.217	-25.033	Watervalrivier	Very Good (135)
B6H001	518	1910	2000	30.800	-24.667	Blyderivier	Very Good (132)
B6H003	92	1959	2000	30.800	-24.683	Treurrivier	Very Good (136)
C5H007	348	1923	2000	26.317	-29.133	Renosterspruit	Good (123)
C7H003	914	1947	2000	27.283	-27.350	Heuningspruit	Good (128)
C8H003	806	1954	2000	28.933	-27.833	Corneliusrivier	Very Good (138)
D5H003	1509	1927	2000	20.350	-31.800	Visrivier	Good (118)
G1H008	395	1954	2000	19.067	-33.300	Klein-Bergrivier	Good (116)
G1H009	5.7	1964	2000	19.167	-33.383	Brakkloofspruit	Good (129)
G1H010	10	1964	2000	19.150	-33.383	Knolvleispruit	Good (101)
G1H011	27	1964	2000	19.100	-33.367	Watervalsrivier	Good (128)
G2H012	244	1965	2000	18.733	-33.450	Dieprivier	Very Good (131)
G4H006	600	1963	2000	19.600	-34.400	Kleinrivier	Fair (92)
G5H008	382	1964	2000	20.017	-34.283	Soutrivier	Excellent (140)
H1H007	84	1935	2000	19.133	-33.567	Witrivier	Good (129)
H1H013	53	1965	2000	19.283	-33.350	Koekedourivier	Good (112)
H3H005	76	1965	2000	20.050	-33.700	Keisierivier	Very Good (137)
H7H004	28	1951	2000	20.700	-33.912	Huisrivier	Fair (74)
J4H003	95	1965	2000	21.583	-34.017	Weyersrivier	Fair (93)
K3H002	1.04	1961	2000	22.450	-33.933	Rooirivier	Good (124)
K3H004	34	1961	2000	22.417	-33.950	Malgasrivier	Very Good (130)
K4H002	22	1961	2000	22.833	-33.867	Karatararivier	Fair (96)
K4H003	72	1961	2000	22.700	-33.900	Dieprivier	Very Good (132)
K5H002	133	1961	2000	23.017	-33.883	Knysnarivier	Good (106)
K6H001	165	1961	2000	23.133	-33.800	Keurboomsrivier	Good (107)
K7H001	57	1961	2000	23.633	-33.950	Bloukransrivier	Good (104)
K8H001	35	1961	2000	24.017	-33.967	Kruisrivier	Good (125)
K8H002	35	1961	2000	24.050	-33.967	Elandsrivier	Good (129)
L8H001	21	1965	2000	23.833	-33.850	Waboomsrivier	Good (118)
R1H014	70	1953	2001	26.933	-32.633	Tyumerivier	Good (104)
R2H001	29	1946	2001	27.283	-32.717	Buffelsrivier	Very Good (133)
R2H006	119	1948	2001	27.367	-32.850	Mgqakweberivier	Good (122)
R2H008	61	1947	2001	27.367	-32.767	Quencwerivier	Very Good (137)
S3H006	2170	1964	2000	26.783	-31.917	Klaas-Smitrivier	Good (117)
S6H003	215	1964	2001	27.517	-32.500	Toiserivier	Good (129)
T3H004	1029	1947	2001	29.417	-30.567	Mzintlavarivier	Good (107)
T3H008	2471	1962	2000	29.150	-30.567	Mzimvuburivier	Good (105)
T3H009	307	1964	2000	28.350	-31.067	Mooirivier	Good (118)

T4H001	715	1951	2000	29.817	-30.733	Mtamvunarivier	Good (108)
T5H003	140	1949	2000	29.533	-29.733	Polelarivier	Fair (94)
T5H004	545	1949	2000	29.467	-29.767	Mzimkulurivier	Good (120)
U1H005*	1741	1960	2000	29.906	-29.744	Mkomazirivier	Very Good (134)
U2H006	339	1954	2000	30.267	-29.367	Karkloofriver	Very Good (136)
U2H007	358	1954	2000	30.150	-29.433	Lionsrivier	Fair (99)
U2H011	176	1958	2000	30.250	-29.633	Msunduzerivier	Good (127)
U2H012	438	1960	2000	30.483	-29.433	Sterkrivier	Good (107)
U2H013	299	1960	2000	30.117	-29.500	Mgenirivier	Good (126)
U4H002	316	1949	2000	30.617	-29.150	Mvotirivier	Fair (93)
U7H007	114	1964	2000	30.233	-29.850	Lovurivier	Good (101)
V1H001	4176	1925	2000	29.817	-28.733	Tugelarivier	Fair (93)
V1H009	196	1955	2000	29.767	-28.883	Bloukransrivier	Very Good (136)
V1H010	782	1965	2000	29.533	-28.817	Klein Tugelarivier	Fair (98)
V3H002	1518	1929	2000	29.933	-27.600	Buffelsrivier	Fair (99)
V3H007	129	1948	2000	29.833	-27.833	Ncandurivier	Very Good (133)
V3H009	148	1958	2000	29.950	-27.883	Hornrivier	Very Good (133)
V6H003	312	1954	2000	30.133	-28.300	Wasbankrivier	Fair (92)
V6H004	658	1954	2000	30.000	-28.400	Sondagsrivier	Good (111)
V7H012	196	1963	2000	29.867	-29.000	Kleinboesmanrivie	Good (128)
W1H004	20	1948	2000	31.450	-28.867	Mlalaziriver	Very Good (130)
W4H004	948	1950	2000	30.850	27.517	Bivanerivier	Good (111)
W5H006	180	1950	2000	30.833	-27.100	Swartwaterrivier	Good (107)
X2H005	642	1950	2000	30.967	-25.417	Nelsrivier	Very Good (135)
X2H008	180	1948	2000	30.917	-25.783	Queensrivier	Very Good (139)
X2H010	126	1948	2000	30.867	-25.600	Noordkaaprivier	Fair (79)
X2H012	91	1956	2000	30.250	-25.650	Dawsoni'sspruit	Good (123)
X2H013	1518	1959	2000	30.700	-25.433	Krokodilrivier	Fair (76)
X2H014	250	1959	2000	30.700	-25.367	Houtbosloop	Good (114)
X2H015	1554	1959	2000	30.683	-25.483	Elandssrivier	Good (102)
X2H022	1639	1960	2000	31.317	-25.533	Kaaprivier	Good (101)
X2H024	80	1964	2000	30.817	-25.700	Suidkaaprivier	Fair (79)
X3H001	174	1948	2000	30.767	-25.088	Sabierivier	Good (128)
X3H002	55	1964	2000	30.667	-25.083	Klein-Sabierivier	Very Good (137)
X3H003	52	1948	2000	30.800	-24.983	Mac-Macrivier	Very Good (130)

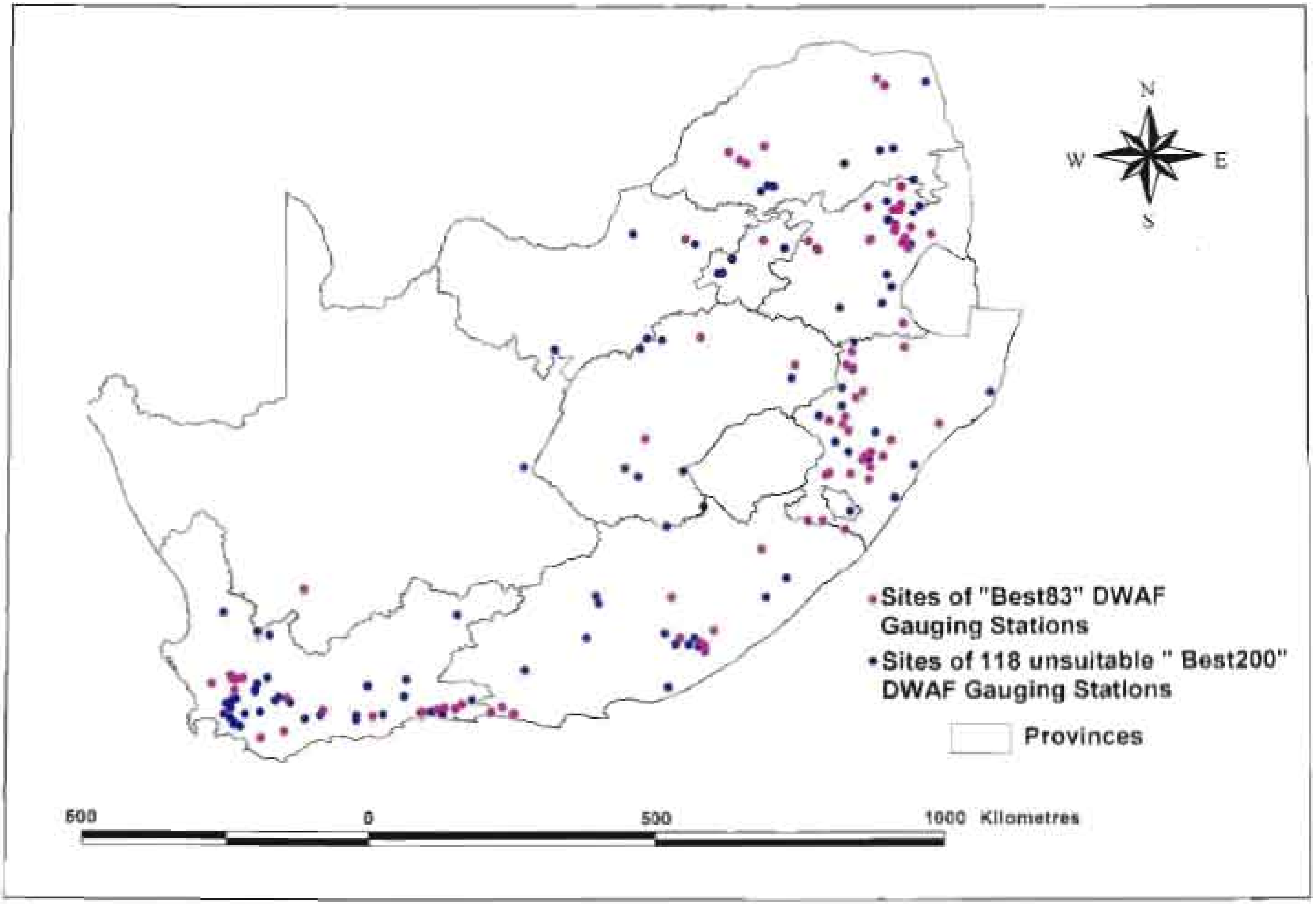

Figure 5.8 Distribution of the 83 stream sites across South Africa

4 HIGH INFORMATION, NON-REDUNDANT, HYDROLOGICAL INDICES FOR ASSESSING ECOLOGICAL FLOW REQUIREMENTS

4.1 Introduction

The identification of high information, non-redundant, hydrological indicators for characterising streamflow regimes is useful to water resources management, particularly where thresholds of streamflow regime characteristics have ecological relevance. The use of multivariate analysis to discard redundant variables in ecological datasets, particularly where there are a large number of variables available for measurement, can permit the selection of a subset of variables that contain the most information (King and Jackson, 1999). Reducing variable redundancy in large datasets prior to statistical analysis is important in ecological studies for strengthening the reliability of the results (King and Jackson, 1999; Olden and Poff, 2003). In this Section, a relatively large hydrological dataset (74 hydrological indices for 83 stream sites) is investigated using Principal Component Analyses, PCA, to display the correlations, or redundancies, among the hydrological indices and to identify a reduced subset (or subsets) of hydrological indices which explain a dominant proportion of statistical variation in the entire set of indices and which adequately represent the different facets of the streamflow regimes found in South Africa, using the methods of Olden and Poff (2003). Knowledge of these high information, non-redundant hydrological indices could be useful for streamflow-related scenarios, particularly when used in conjunction with ecological information (Olden and Poff, 2003). In addition, the record length required for the stability of these indices, among and for each of the different streamflow groups, is examined so that reliable assessments can be made of the streamflow patterns required to sustain ecological functioning.

4.2 Multivariate Analyses for Selecting High Information Variables

The approach of Method Two in the 1994-Study (*c.f.* Section 3.2.1 of this Chapter) to group streamflow types by associating stations grouped naturally using *covariance* analyses of eight streamflow variables was considered innovative at that time (Joubert and Hurly, 1994). In recent years, and internationally, several studies have adopted multivariate ordination approaches to identify the subsets of hydrological variables for characterising different streamflow regimes. Clausen and Biggs (2000) used PCA to identify four groups of hydrological variables for ecological studies, based on covariance among the sites of 62 New Zealand rivers. In a different approach, Olden and Poff (2003) employed PCA using a correlation matrix to examine the major patterns of inter-correlation among 171 hydrological indices at 420 stream sites across the USA and to provide statistically and ecologically based recommendations for the selection of a reduced set of indices for use in hydro-ecological studies.

The main purpose of the PCA applied in this Study is to examine the relationships among 74 hydrological indices at 83 stream sites in South Africa with records of streamflows representing relatively natural environmental conditions (*c.f.* Section 3 of this Chapter). Thus the approach adopted in this Study followed that described by Olden and Poff (2003) and a PCA was conducted on the correlation matrix rather than the covariance matrix, since the main objective of this Study is to investigate relationships among the indices rather than among the streamflow sites. Nevertheless, PCA does provide information on both the descriptors (in this case, hydrological indices) and the objects (in this instance the streamflow sites) and an additional benefit of using the correlation matrix is that the descriptors are standardised in the analysis. Thus, since the 74 hydrological indices (*c.f.* Section 3.3.2 of this Chapter) are not dimensionally homogenous, using the correlation matrix permits all the indices to contribute equally to the PCA and to the clustering of the objects (stream sites) in reduced space (Legendre and Legendre, 1998).

The range of values of the 74 hydrological indices, and the medians thereof, for the 83 stream sites for the period from 1 October 1965 to 30 September 2001 are provided in Table 5.5. PCA was originally defined for data with multinormal distributions; while deviations from normality do not necessarily bias the analysis, it is important to ascertain that the data are not unreasonably skewed (Legendre and Legendre, 1998). Analysis conducted on data with strongly skewed distributions of the hydrological indices would lead to the first few principal components separating only a few of the stream sites with extreme values rather than displaying the main axes of variation of all the stream sites in the Study (Legendre and Legendre, 1998). Table 5.6 shows that the distributions many of the hydrological indices in the dataset are strongly skewed. The information relating to whether the distributions of the hydrological indices were skew was sourced from the output files generated by the Genstat Version 6 computer software for the PCA described

Table 5.5 Hydrological indices and their ranges for 83 sites on South African rivers for the 36 year period from 1 October 1965 to 30 September 2001. All indices are based on daily mean flow data obtained from DWAF.

Hydrological Index		Unit	Range (median)	Hydrological Index		Unit	Range (median)
Magnitude of Average Flow Conditions	M_A1	$m^3.s^{-1}$	0 – 6.061 (0.315)	**Duration of Low Flow Conditions**	D_L9	-	0 – 15.10 (0.79)
	M_A2	$m^3.s^{-1}$	0 – 13.328 (0.446)		D_L10	-	0 – 9.33 (0.80)
	M_A3	$m^3.s^{-1}$	0 – 25.706 (0.625)		D_L11	-	0 – 19.71 (0.86)
	M_A4	$m^3.s^{-1}$	0 – 36.100 (0.762)		D_L12	-	0 – 38.82 (0.90)
	M_A5	$m^3.s^{-1}$	0 – 62.848 (0.877)		D_L13	-	0 – 22.5 (0)
	M_A6	$m^3.s^{-1}$	0 – 38.347 (0.781)		D_L14	-	0 – 3.99 (1.14)
	M_A7	$m^3.s^{-1}$	0 – 17.319 (0.520)	**Duration of High Flow Conditions**	D_H1	$m^3.s^{-1}$	0.045 – 255.96 (14.592)
	M_A8	$m^3.s^{-1}$	0 – 6.508 (0.339)		D_H2	$m^3.s^{-1}$	0.039 – 199.512 (8.389)
	M_A9	$m^3.s^{-1}$	0 – 7.447 (0.316)		D_H3	$m^3.s^{-1}$	0.025 – 152.146 (5.713)
	M_A10	$m^3.s^{-1}$	0 – 9.003 (0.255)		D_H4	$m^3.s^{-1}$	0.011 – 106.977 (2.764)
	M_A11	$m^3.s^{-1}$	0 – 6.709 (0.235)		D_H5	$m^3.s^{-1}$	0.010 – 59.889 (1.731)
	M_A12	$m^3.s^{-1}$	0 – 4.596 (0.245)		D_H6	days	2 – 30.5 (8.69)
	M_A13	-	0 – 188.15 (1.62)		D_H7	-	0.31 – 39.61 (1.23)
	M_A14	-	0 – 50.47 (1.38)		D_H8	-	0.40 – 24.97 (1.28)
	M_A15	-	0 – 11.7 (1.43)		D_H9	-	0.52 – 20.64 (1.21)
	M_A16	-	0 – 12.33 (1.46)		D_H10	-	0.59 –11.18 (1.10)
	M_A17	-	0 – 46.22 (1.58)		D_H11	-	0.4 – 16.01 (1.01)
	M_A18	-	0 – 23.41 (1.36)		D_H12	-	0.34 – 13.90 (1.15)
	M_A19	-	0 – 40.76 (1.82)		D_H13	days	0 – 125 (9)
	M_A20	-	0 – 293.5 (1.45)				
	M_A21	-	0 – 10.52 (1.20)	**Timing of Average Flow Conditions**	T_A1	-	0.23 – 0.86 (0.33)
	M_A22	-	0 – 26.28 (1.33)		T_A2	-	0.31 – 0.94 (0.64)
	M_A23	-	0 – 7.41 (1.16)				
	M_A24	-	0 – 17.54 (1.32)				
	M_A25		0 – 133.45 (8.00)	**Timing of Extreme Flow Conditions**	T_L1	-	200.5 – 527.5 (275)
Magnitude of Low or High Flow Conditions	M_L1	-	0 – 0.51 (0.05)		T_L2	-	0 – 0.49 (0.11)
	M_L2	-	0 – 6.87 (0.72)		T_H1	-	183 – 499.5 (398.5)
	M_L3	-	0.12 – 0.75 (0.34)		T_H2	-	0.09 – 0.5 (0.18)
	M_L4	$m^3.s^{-1}$	0 – 4.194 (0.12)		T_H3	-	0.21 – 1 (0.36)
	M_H1	-	4.30 – 15700 (47.59)	**Frequency of Extreme Flow Conditions**	F_L1	year^{1-}	0 – 12 (4.5)
Duration of Low Flow Conditions	D_L1	$m^3.s^{-1}$	0 – 1.490 (0.028)		F_L2		0 – 3.83 (1)
	D_L2	$m^3.s^{-1}$	0 – 1.592 (0.033)		F_H1	year^{1-}	1 – 25.5 (7.5)
	D_L3	$m^3.s^{-1}$	0 – 1.668 (0.039)		F_H2	-	0.21 – 2.00 (0.63)
	D_L4	$m^3.s^{-1}$	0 – 2.355 (0.059)	**Rate of Change of Average Flow Conditions**	R_A1	$m^3.s^{-1}.d^{-1}$	0.008 – 8.808 (0.542)
	D_L5	$m^3.s^{-1}$	0 – 3.420 (0.117)		R_A2	$m^3.s^{-1}.d^{-1}$	0.011 – 4.742 (0.262)
	D_L6	year^{1-}	0 – 360 (0)		R_A3	-	2 – 127 (79)
	D_L7	days	0 – 25.4 (9.1)		R_A4	-	0.44 – 10.35 (1.08)
	D_L8	-	0 – 12.95 (0.76)		R_A5	-	0.45 – 4.74 (0.94)
					R_A6	-	0.11 – 7.00 (0.31)

Table 5.6 Distribution of values of 74 hydrological indices at 83 DWAF streamflow gauging stations for the period 1965 to 2001

HYDROLOGICAL INDEX		DISTRIBUTION (√ = skewed; * = not skewed)			
		No transformation		Log transformation	
		Skewness (x)	Value of x	Skewness (x)	Value of x
Magnitude of Average Flow Conditions	M_A1	√	3.30	√	1.66
	M_A2	√	4.00	√	1.81
	M_A3	√	4.18	√	1.37
	M_A4	√	4.37	√	1.28
	M_A5	√	4.65	√	1.19
	M_A6	√	4.25	√	1.17
	M_A7	√	3.61	√	1.33
	M_A7	√	2.69	√	1.41
	M_A9	√	3.36	√	1.72
	M_A10	√	4.22	√	2.02
	M_A11	√	3.57	√	2.01
	M_A12	√	2.90	√	1.85
	M_A13	√	8.58	√	3.00
	M_A14	√	7.00	√	2.30
	M_A15	√	3.72	*	0.51
	M_A16	√	3.73	*	0.49
	M_A17	√	6.84	*	2.08
	M_A18	√	6.22	*	1.28
	M_A19	√	6.08	*	1.59
	M_A20	√	9.05	√	3.48
	M_A21	√	3.22	*	0.76
	M_A22	√	6.94	*	1.44
	M_A23	√	2.20	*	0.56
	M_A24	√	4.17	*	1.04
	M_L25	√	3.73	*	0.45
Magnitude of Extreme Flow Conditions	M_L1	√	1.65	√	1.42
	M_L2	√	2.43	*	0.87
	M_L3	*	0.58	*	0.35
	M_L4	√	1.56	√	2.33
	M_H1	√	4.48	*	1.51
Duration of Low Flow Conditions	D_L1	√	2.89	√	2.42
	D_L2	√	2.92	√	2.42
	D_L3	√	2.89	√	2.40
	D_L4	√	3.11	√	2.40
	D_L5	√	3.22	√	2.26
	D_L6	√	2.53	√	1.47
	D_L7	*	0.12	*	-1.41
	D_L8	√	4.34	√	1.11

HYDROLOGICAL INDEX		DISTRIBUTION (√ = skewed; * = not skewed			
		No transformation		Log transformation	
		Skewness (x)	Value of x	Skewness (x)	Value of x
Duration of Low Flow Conditions	D_L9	√	4.49	√	1.48
	D_L10	√	3.52	√	0.85
	D_L11	√	4.79	√	1.45
	D_L12	√	7.32	√	2.19
	D_L13	√	5.63	√	2.34
	D_L14	*	0.58	*	-0.51
Duration of High Flow Conditions	D_H1	√	3.95	*	0.06
	D_H2	√	4.22	*	0.26
	D_H3	√	4.33	*	0.45
	D_H4	√	4.86	*	0.82
	D_H5	√	4.51	*	0.99
	D_H6	*	1.60	*	-0.06
	D_H7	√	8.40	√	3.60
	D_H8	√	7.71	√	3.17
	D_H9	√	7.60	√	3.39
	D_H10	√	5.83	√	2.66
	D_H11	√	7.44	√	3.25
	D_H12	√	6.07	√	1.94
	D_H13	√	2.62	*	-0.15
Timing of Average Flow Conditions	T_A1	√	1.40	*	1.17
	T_A2	*	-0.17	*	-0.34
Timing of Extreme Flow Conditions	T_L1	*	2.13	*	1.56
	T_L2	*	1.70	*	1.44
	T_H1	*	-1.01	*	-1.32
	T_H2	*	1.21	*	1.09
	T_H3	√	2.90	*	2.55
Frequency of Extreme Flow Conditions	F_L1	*	0.44	*	-1.08
	F_L2	*	0.78	*	-0.41
	F_H1	*	1.41	*	-0.14
	F_H2	*	1.33	*	0.66
Rate of Change of Average Flow Conditions	R_A1	√	3.34	*	1.57
	R_A2	√	4.03	√	2.34
	R_A3	√	-0.98	√	-3.07
	R_A4	√	5.13	√	2.53
	R_A5	√	2.96	*	1.87
	R_A6	√	6.81	√	3.79

in Section 4.4.1 of this Chapter. The values of skewness were determined using the data analysis tools in Microsoft Excel. In addition, many of the hydrological indices have values of zero, which according to Legendre and Legendre (1998) can "lead to ordinations that produce inadequate estimates of the distances among sampling sites". Nevertheless, the values of zero in this Study represent real values of either the absence of streamflow, or of counts, or of proportions of streamflows. The anomalies of the skewed distribution of the dataset and the many zero values were addressed by logarithmic transformation of the data. Since the log of zero is meaningless, the value (x) for each of the indices was replaced by $\log(x+1)$ as in Fowler and Cohen (1990). Table 5.6 indicates that this transformation of the data is effective in reducing the skewness of the distribution of several of the hydrological indices among the 83 stream sites.

4.3 Revising the Streamflow Classification

The main purpose of this Study is to highlight patterns of redundancy among a large set of hydrological indices and to identify a minimum subset (or subsets) of indices that explain a dominant proportion of statistical variation in the entire set of indices while adequately representing the different facets of the flow regimes found in South Africa (using the methods of Olden and Poff, 2003), rather than to reassess the streamflow groups that had already been classified in the 1994-Study by Joubert and Hurly (1994). Nonetheless, updating the daily streamflow database for the 83 DWAF stations (Section 3 of this Chapter) provided an opportunity to confirm the suitability of the 1994-Study classification of flow types for the reduced selection of stations used in this Study.

4.3.1 The 1994-Study classification

The 1994-Study highlighted differences between ecological and hydrological definitions of different characteristics of the streamflow regime. For example, hydrological descriptions of flow conditions such as perennial, seasonal and episodic flow have no ecologically relevant descriptions, mainly because ecosystems function on a *continuum of flow conditions*. Consequently, the 1994-Study redefined these descriptions and created a new term *quasi-perennial-seasonal* to bridge the boundaries between the hydrological descriptions. In addition, Joubert and Hurly (1994) provide generally accepted definitions for different kinds of streamflow *viz.*:

(a) *Episodic,* to describe flow that occurs only after rainfall events and which does not necessarily occur each year;

(b) *Extreme Seasonal,* for flow that usually occurs for less than half the year, every year and during the same season(s);

(c) *Seasonal,* which is flow that usually occurs for more than half the year, every year and during the same season(s);

(d) *Quasi-Perennial-Seasonal*, which describes flow that in some years continues all year, but in other years ceases for anything from a few days to most of the year;

(e) *Perennial*, *i.e.* flow usually continues all year, every year; and

(f) *Flashy* flow which is characterised by frequent floods of short duration.

The 1994-Study applied two different statistical techniques to the long-term daily streamflow records to group South African rivers by streamflow type (*c.f.* Section 3.2.1 of this Chapter). While Method Two of the 1994-Study used ordination techniques (biplot analysis), the aim of Method One in the 1994-Study (*i.e.* describing the flow types of different river systems by the "different combinations of variables dominating the formation of each group") is more in keeping with the aims of this Study, than the aim of Method Two, which was to form geographical boundaries based on perenniality, each with further subdivisions made in a user-based decision process. Moreover, the two methods produced very different results, with the classification of the 10 streamflow groups formed by Method Two being more subjective, and difficult to reproduce. On the other hand, Method One formed eight streamflow groups (Groups A to H) which were determined by the dominant streamflow variable (of eight variables, *c.f.* Section 3.2.1 of this Chapter) applied in the 1994-Study. The Method One streamflow groups are briefly summarised in Table 5.7.

PCA provides information about the relationships among both objects and descriptors (Legendre and Legendre, 1998), since both the descriptor-axes and the objects can be plotted in reduced space. A scatter plot of PCA scores for the first two principal components from the 74 x 74 correlation matrix (*c.f.* Figure 5.9) was examined to highlight the clustering of the 83 DWAF sites, with respect to first and second principal component axes.

Table 5.7 The classification of streamflow groups, using eight hydrological variables (see Section 3.2.1 for details), identified by Method One of the 1994-Study by Joubert and Hurly (1994)

Streamflow Conditions	Group	Streamflow Type
Extreme Seasonal	A	Predictable low / zero flow conditions
Extreme Seasonal to Perennial	B	Long flood durations
	G	Unpredictable
	D	Strongly seasonal
	H	Unpredictable flow and floods
Perennial	C	Predictable infrequent floods
	E	Frequent floods
	F	Long intervals between floods

The positions of the sites shown in Figure 5.9 were reviewed with the groups formed by Method One of the 1994-Study. Thus, the letters shown in Figure 5.9 refer to the streamflow group used in the classification of Method One (eight groups, described as A to H). Numbers annexed to the letters are used to distinguish the stations within each of the eight groups of Method One. The relevant group letters and numbers used in Method One of the 1994-Study, as well as the roman numerals used to annotate the groups formed in this Study, are shown in Table 5.8.

Fewer stations were analysed in this Study than in the 1994-Study classification. Therefore, it was considered acceptable to reduce the total number of streamflow groups to six groups (*c.f.* Figure 5.9). Grouping objects based on their PCA scores does involve a degree of subjectivity (Ndlovu, 2004). However, this was considered to be acceptable within the realms of this study. Stations at the extremes of the plot were placed in three separate groups (Groups I, II, and V respectively), whereas three additional groups were formed from the remaining stations based on the similarity of their PCA scores along the first two principal component axes. In addition, some stations were grouped in a logical manner despite small differences in their scores. Some examples follow:

(a) Sites B7 (X3H002), C1 (A9H003), E13 (X2H024) in Figure 5.9 have negative scores on the first principal component axis, yet were placed in Group VI since they have similar scores with the Group VI stations on the second principal component axis. In addition, it was more reasonable to group these stations together given the similarities of their high values of M_L3, the Alt-BFI, (0.75 for

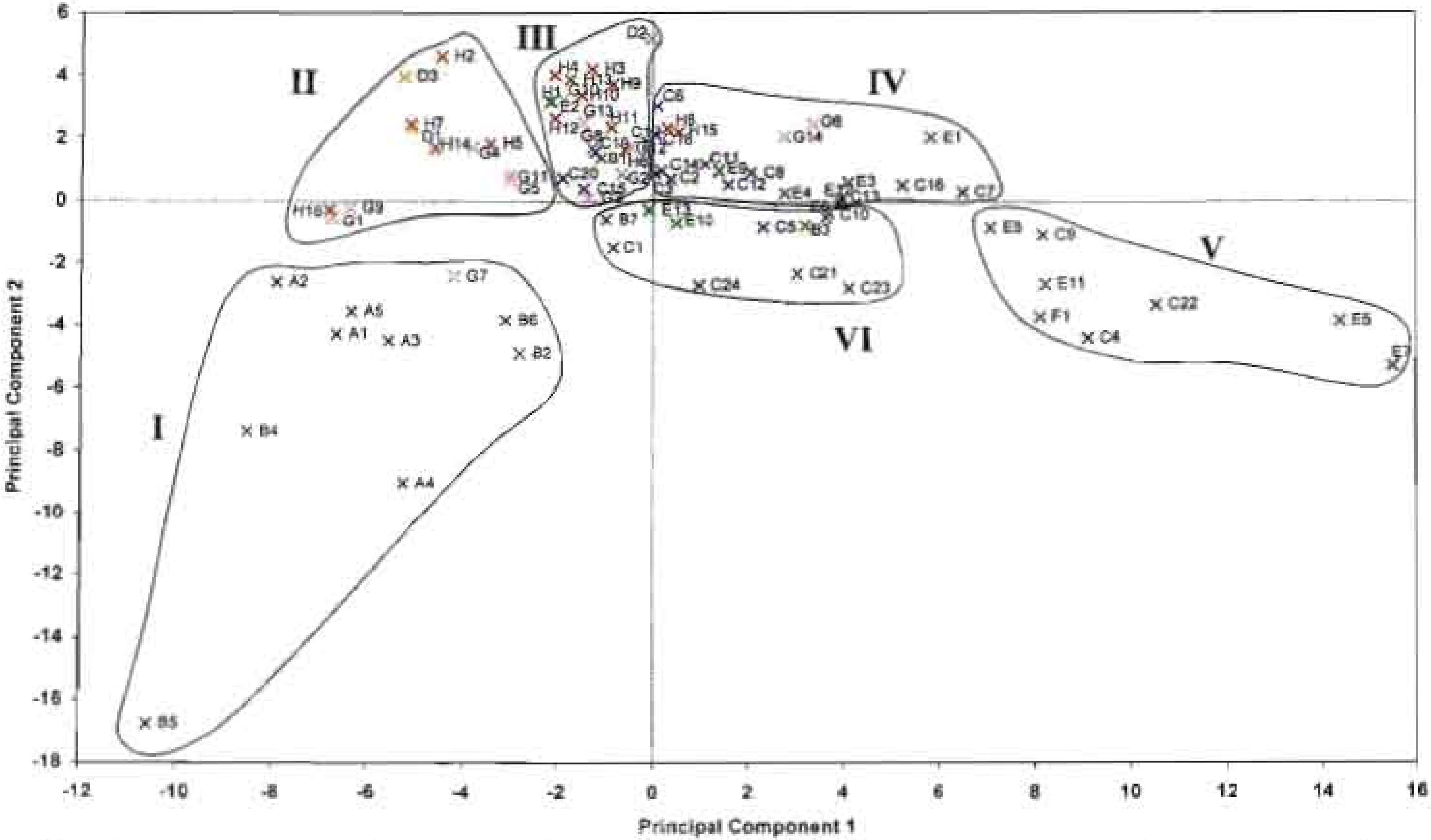

Figure 5.9 Bivariate plot of scores for the first two principal components from a PCA of the 83 stream sites based on the correlation matrix of 74 hydrological indices. Letters A to H refer to the classification of Method One of the 1994-Study by Joubert and Hurley (1994). Numbers annexed to letters distinguish the sites within the 1994-Study grouping. Roman numerals I to VI indicate the revised grouping used in the present Study.

Table 5.8 Revised Groups of the 83 DWAF gauging stations

Gauging Station	1994-Study Group	Revised Group
A2H029	G1	II
A2H032	A1	I
A4H002	B1	III
A4H005	B2	I
A4H008	G2	III
A5H004	G3	III
A9H003	C1	VI
A9H004	B3	VI
B1H002	G4	II
B1H004	C2	IV
B4H005	C3	IV
B6H001	C4	V
B6H003	C5	VI
C5H007	A2	I
C7H003	A3	I
C8H003	G5	II
D5H003	A4	I
G1H008	G6	IV
G1H009	D1	II
G1H010	A5	I
G1H011	D2	III
G2H012	G7	I
G4H006	G8	III
G5H008	B4	I
H1H007	E1	IV
H1H013	C6	IV
H3H005	B5	I
H7H004	D3	II
J4H003	H1	III
K3H002	H2	II
K3H004	H3	III
K4H002	H4	III
K4H003	H5	II
K5H002	H6	III
K6H001	H7	II
K7H001	H8	IV
K8H001	H9	III
K8H002	H10	III
L8H001	E2	III
R1H014	H11	III
R2H001	H12	III
R2H006	H13	III
R2H008	H14	II
S3H006	G9	II
S6H003	G10	III
T3H004	E3	IV
T3H008	C7	IV
T3H009	E4	IV
T4H001	F1	V
T5H003	C8	IV
T5H004	C9	V
U1H005	E5	V
U2H006	E6	VI
U2H007	C10	VI
U2H011	C11	IV
U2H012	C12	IV
U2H013	C13	IV
U4H002	C14	IV
U7H007	C15	III
V1H001	E7	V
V1H009	G11	II
V1H010	C16	IV
V3H002	B6	I
V3H007	G12	III
V3H009	G13	III
V6H003	H15	IV
V6H004	G14	IV
V7H012	C17	III
W1H004	H16	II
W4H004	E8	V
W5H006	C18	IV
X2H005	E9	IV
X2H008	C19	III
X2H010	E10	VI
X2H012	C20	III
X2H013	E11	V
X2H014	C21	VI
X2H015	C22	V
X2H022	E12	IV
X2H024	E13	VI
X3H001	C23	VI
X3H002	B7	VI
X3H003	C24	VI

B7; 0.57 for C1 and 0.62 for E13) compared with the much lower M_L3 values for sites in Groups I, II and III (group medians of 0.17, 0.27 and 0.34 respectively).

(b) Sites G1 (A2H029), G9 (S3H006) and H16 (W1H004) have negative scores on the first principal component axis, yet were grouped with Group II, rather than Group I because of the similarities of their low values for T_A1, the predictability index, (0.25 for G1; 0.42 for G9 and 0.29 for H16) compared with the much higher values for sites in Group I (group median of 0.63).

Figure 5.9 shows a general environmental gradient from "extreme seasonal", or "harsh arid", conditions represented by ephemeral river systems (Group I), through mixed quasi-perennial-seasonal and perennial regimes (Groups II, III and IV) to the more moderate regimes found in perennial systems (Group V) and (Group VI). These Groups are discussed further in Section 4.3.2 of this Chapter.

While there are some discrepancies between the grouping of Method One of the 1994-Study and the present Study, the main distinctions among the streamflow groups are maintained. Examples follow:

(a) Those sites comprising the Extreme Seasonal Group of Method One of the 1994-Study (sites annotated A in Figure 5.9) are all still shown to be included in a group distinct from all other groups.

(b) In general, those sites comprising the Unpredictable Flow and Floods group of Method One of the 1994-Study (annotated H in Figure 5.9) are still shown to be together within a group (*i.e.* mainly in Group III in Figure 5.9). However, the present Study shows, that there is some drift from the pattern in 1994-Study and two sets of sites (H2, H5, H7 and H14) and (H8 and H15) are now included in two different Groups (*i.e.* Groups II and IV in Figure 5.9).

(c) The sites classified as Unpredictable in Method One of the 1994-Study (annotated G in Figure 5.9) show most differences between Method One and the present Study, and the sites can now be shown among four different groups (*i.e.* Groups I, II III and IV in Figure 5.9). However, none of the sites are shown to be at the more moderate, perennial end of the environmental range in the present Study.

(d) With only four exceptions (C15, C17, C19 and C20), the sites classified as perennial streamflow regimes with Predictable Infrequent Floods in Method One of the 1994-Study (annotated C), are still classed as perennial streamflow regimes, but are allocated to three different groups in the present Study (*i.e.* Groups IV, V and VI in Figure 5.9).

(e) Those sites distinguished in Method One of the 1994-Study by their Frequent Floods (annotated E) and by their Long Intervals between Floods (annotated F) are also still classed as having perennial flow regimes but are allocated to three different Groups in the present Study (*i.e.* Groups IV, V and VI in Figure 5.9).

(f) The sites classified as having Long Flood Durations and comprising Group B of Method One of the 1994-Study were the most affected group in the revised classification of streamflow types. Although 50% of these sites (*i.e.* four sites) have been included with Group I in at the harsh arid end of the environmental range (*c.f.* Figure 5.9), 25% of the sites (*i.e.* two sites) have been included with Group VI at the perennial end of the range of streamflow types (*c.f.* Figure 5.9).

4.3.2 Revised groupings

The six revised groups were categorised within three main groups of streamflow type using a similar set of indices to those of Method One's classification system in the 1994-Study. In the interest in preserving commonality of terminology with the 1994-Study, the same general flow types, ranging from "extreme-seasonal" to "perennial" (as defined in Section 4.3.1 of this Chapter) were used to distinguish the three main-groups of flow types. However, because different indices of variability and predictability were used in this Study, particularly those relating to the Desktop Reserve model indices of variability, Alt-BFI and CDB, as well as the majority of the IHA indicators, there were some divergences in the indices used to form distinctions *within* the three main groups. Not all of the eight indices used in the 1994-Study were comparable to the indices used in the present Study (*e.g.* the 1994-Study index GRCV, the intra-annual coefficient of variation, was not used in this Study). In addition, there were differences between this Study and the 1994-Study regarding the derivation of even comparable indices. The indices used in this Study to form distinctions within the three main groups of flow patterns (and thus to reclassify the streamflow types found in South Africa), their derivation and comparable index applied in the 1994-Study, are shown in Table 5.9. Table 5.9 indicates that for the purposes of this Study, HICOUNT can be equated with the 1994-Study variable FLOFRQ; HIGHDUR with FLODUR; %FLOODS with FLDPRED and FLOODFREE with FLOINT.

Table 5.9 The indices derived from daily flow data used to reclassify the streamflow types found in South Africa

Present Study Indicator	**Description of Present Study Indicator**	**1994-Study Equivalent**	**Description of 1994-Study Indicator** (from Joubert and Hurly, 1994)
ZERODAY	Average over the period of the record of the number of days in a year with zero flow	**ZERODAY**	Same as Present Study
LOWDUR	Average duration over the period of the record of the number of occurrences of low pulses within a year (flow less than the 25th percentile of all daily flows for the time period), Richter *et al.* (1996)	No equivalent	None
HIGHCOUNT	Average over the period of the record of the number of occurrences of high pulses within a year (flow greater than the 75th percentile of all daily flows for the time period), Richter *et al.* (1996)	**FLOFRQ**	Number of floods per year
HIGHDUR	Average duration over the period of the record of the number of occurrences of high pulses within a year (flow greater than the 75th percentile of all daily flows for the time period), Richter *et al.* (1996)	**FLODUR**	Mean duration of floods
REVERSALS	Average over the period of the record of the number of positive and negative changes between consecutive daily values in a year, Richter *et al.* (1996)	No equivalent	None
Alt-BFI	The "baseflow" index (*c.f.* Sect 3.3.2, Ch 5) based on the South African Desktop Reserve Determination baseflow index, BFI (Hughes and Hannart, 2003)	No equivalent	None
CDB	The "overall variability" index (*c.f.* Sect 3.3.2, Ch 5), based on the South African Desktop Reserve Determination index, CVB (Hughes and Hannart, 2003)	No equivalent	None
HFI	A high flow index, based on the median of annual maximum flows, being the average over the period of record of the highest annual daily flow divided by the median annual daily flow (Olden and Poff, 2003)	No equivalent	None
PRED	Predictability of flow using Colwell's Index (*c.f.* Sect 2.2.2, Ch 5); Predictability, (PRED) = Constancy (CONST) + Contingency (CONT), following Colwell (1974)	**PRED**	Same as Present Study
PROP	Proportion of PRED due to CONST (Colwell, 1974)	**PROP**	Same as Present Study
%FLOODS	Percentage of floods that occur during a given 60 day period in all years (Poff and Ward, 1989)	**FLOPRED**	The maximum proportion of floods occurring in a 60 day period (Poff and Ward, 1989)
FLOODFREE	Length of flood-free season (Poff and Ward, 1989)	**FLOINT**	Median number of days between floods (Poff and Ward, 1989)
No equivalent	None	**GRCV**	Average over the period of record of intra-annual coefficients of variation (Poff and Ward, 1989)

The revised classification of the streamflow types (or groups) used in this Study, numbers of sites within each group, group medians (large numerals) and CDs (smaller numerals) of selected hydrological indices (*c.f.* Table 5.9) for the six groups of DWAF gauging stations determined using the object (stream site) scores from the PCA (*c.f.* Figure 5.9), are shown in Table 5.10. Shading around the medians indicates a distinguishing characteristic of the Group. Figure 5.10 shows the distribution of the six revised groups of DWAF gauging stations. Stations recording mainly extreme seasonal or intermittent flow were placed in an exclusive group (Group I). Some stations recording perennial flow were grouped into two subgroups based on distinct streamflow characteristics (Groups V and VI). Three further subgroups (Groups II, III and IV) were formed from the remaining stations, some of which were recording quasi-perennial-seasonal and some perennial flow, which were further distinguished by streamflow characteristics relating to patterns of predictability of flow and flood events. The revised groupings are discussed below, with reference to group medians (*c.f.* Table 5.10) as well as the regional trends (Figure 5.10). For the sake of consistency among the different groupings, the style and terminology used mirror that applied in Method One of the 1994-Study of Joubert and Hurly (1994).

Extreme Seasonal Main Group

Nine stations in **Group I** recorded extreme seasonal flow during the 36 year period analysed (1965 – 2000), while one station (A4H005) recorded quasi-seasonal-perennial flow. At A4H005 flow occurs for less than half the year in seven of the 36 years analysed. In common with the other stations in this group, the streamflow regime at A4H005 has a very low baseflow component (Alt-BFI). Five of the stations record flows that are perfectly seasonal (%FLOODS, each having a value of 1). The high degree of constancy (PROP) for the streamflow regimes at most of the Group I stations (*c.f.* Figure 5.10) is as a result of flow being zero for most of the year. The sites of these stations are shown on Figure 5.10 as being mostly west of the Drakensberg escarpment, extending through the interior of the sub-continent to the Southern Karoo and the Western Cape.

Mixed Quasi-Perennial-Seasonal and Perennial Main Group

- ***Short, Unseasonal Floods***

Group II comprises a mixed group with 11 stations recording quasi-perennial-seasonal flow and one station recording perennial flow. The stations in this group all record unpredictable flow (group median PRED of 0.28), with an average of only 18% of PRED

Table 5.10 Revised classification of streamflow types found in South Africa. Group medians (large numerals) and the Coefficients of Dispersion (smaller numerals) of selected hydrological indices of each of the six groups determined from the PCA. Shading around the medians indicates a distinguishing index of the Group.

	EXTREME SEASONAL	MIXED			PERENNIAL	
	Group: I Number: 10	II 12	III 22	IV 20	V 8	VI 11
INDICATOR	(ES)	Short, Unseasonal Floods (M-SUF)	Unpredictable Flow and Floods (M-UFF)	Seasonally Predictable Flow and Floods (M-SPFF)	Runoff (P-R)	Sustained Baseflow (P-SB)
ZERODAY	278.00 0.19	26.00 1.52	0.00 0.00	0.00 0.00	0.00 0.00	0.00 0.00
LOWDUR	0.00 0.00	7.42 1.22	8.98 0.38	11.11 0.62	13.95 0.32	10.50 0.47
HICOUNT	4.00 0.5	9.00 0.67	10.00 1.00	7.75 0.52	7.25 0.28	6.00 0.42
HIGHDUR	12.25 0.72	6.55 0.50	7.66 0.86	9.52 0.56	8.89 0.07	9.46 0.59
REVERSALS	20 1.85	68.50 0.23	82.00 0.17	91.50 0.22	99.00 0.19	77.00 0.28
BFI	0.17 0.49	0.27 0.42	0.34 0.33	0.34 0.30	0.45 0.30	0.56 0.37
CDB	32.05 1.99	18.63 0.59	9.61 0.72	7.16 0.58	4.46 0.18	2.84 0.91
HFI	3739.67 3.09	112.64 2.73	62.99 1.09	28.68 2.60	22.38 0.78	10.44 1.28
PRED	0.63 0.30	0.28 0.07	0.29 0.28	0.31 0.36	0.40 0.10	0.48 0.29
PROP	0.85 0.12	0.68 0.25	0.63 0.37	0.50 0.32	0.65 0.29	0.77 0.23
%FLOODS	0.72 0.88	0.32 0.32	0.32 0.38	0.36 0.14	0.40 0.15	0.41 0.22
FLOODFREE	7.00 1.71	0.00 0.00	7.00 4.00	18.00 1.28	19.50 0.62	13.00 1.69

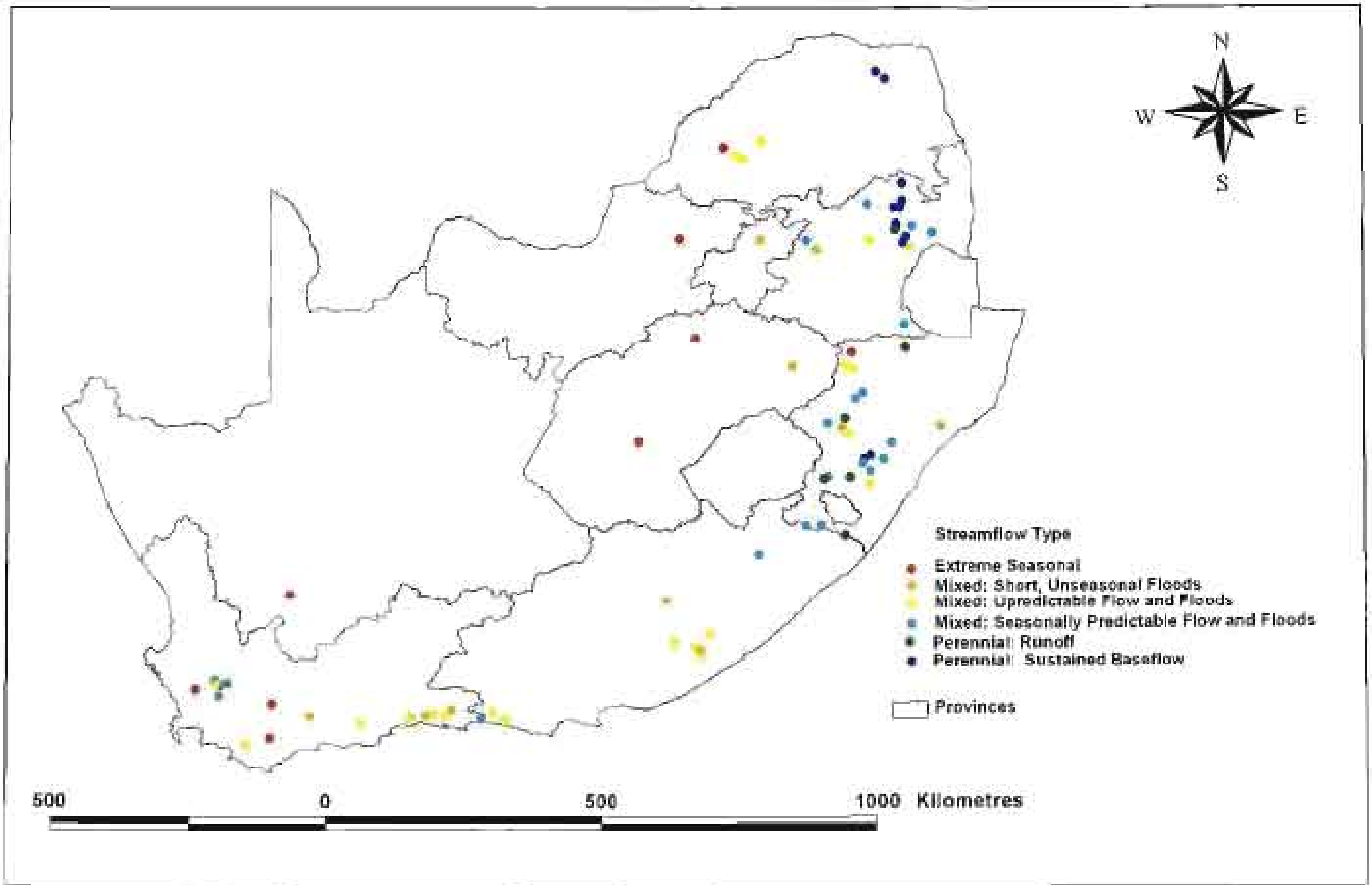

Figure 5.10 The distribution of the six revised groups of DWAF gauging stations

being attributable to constancy. The stations in this group also record very low numbers of days during a year for which no floods have ever occurred over the 36 year record (group median FLOODFREE of 0.00). Flood regimes are therefore highly unseasonal and unpredictable (group median %FLOODS of 0.32). The stations in this group also record, on average, the shortest flood durations (group median HIGHDUR of 6.55) and are relatively "flashy". A relatively large proportion of the flow recorded for the stations in this Group constitutes the "high flow component" (HFI). Figure 5.10 shows that most of these stations are also located in the interior of the sub-continent, but generally east of the stations in Group I, with four stations located along the southern and eastern Cape coastal belt.

- ***Unpredictable Flow and Floods***

Group III is the largest of all the groups with 10 stations recording quasi-perennial-seasonal flow and 12 stations recording perennial flow. The stations in this Group record similar flow patterns to those in Group II. Overall predictability of flow (PRED) and of floods (%FLOODS) for stations in Group III both have, on average, the same low values as Group II. Group III stations have, on average, a slightly higher number of floods each year (group median HICOUNT of 10 compared with 9), which on average are longer than those of the Group II stations (7.66 days compared with 6.55 days). However, stations in Group III generally have a higher incidence of days on which no floods occurred (higher FLOODFREE, group median of 7.00 compared with 0.00) and a slightly lower PROP (group median of 0.63 compared with 0.68) than those in Group II. Thus, Group III flood regimes are generally more seasonal than those of Group II stations. In addition, the stations in Group III generally have a higher baseflow component (group median Alt-BFI of 0.34 compared with 0.29) which, together with a much lower CDB (group median of 9.61 compared with median of 18.63), indicates that, generally, these streamflow regimes are also less variable than those recorded by the stations in Group II. Group III stations are located throughout the sub-continent, on either side of the escarpment and with many along the southern and eastern Cape coastal belt reaching into the former Transkei and KwaZulu-Natal.

- ***Seasonally Predictable Flow and Floods***

Group IV comprises nine stations recording quasi-perennial-seasonal flow and 11 stations recording perennial flow. In common with stations in Groups II and III, stations in this Group also record flow with low overall predictability (group median of 0.31). However, on average, constancy was generally lowest for Group IV stations, as is evidenced by the low group median value for PROP (0.50). Consequently, these stations record flow with a high degree of seasonal predictability. Seasonal predictability of non-flooding is also generally high (group median FLOODFREE of 18.00). Figure 5.10 shows that these stations are mostly in the wetter, summer rainfall areas of the subcontinent, but with a few located closely together in the western Cape.

Perennial Main Group

In common with the 1994-Study, some stations in the perennial main group do record days of zero flow. Very occasionally, some stations recorded days of no flow for more than ten days, the value that Joubert and Hurly (1994) used as a threshold for determining "perennial stations". However, it was considered that the 1994-Study definition of perennial flow as "usually continues all year, every year" was not compromised in this Study, since all stations in the perennial main group record predominantly "perennial flow".

- ***Runoff***

Group V comprises eight stations, which on average, record flows with the highest seasonal predictability of non-flooding (group median FLOODFREE of 19.50) of all the Groups. Moreover, the stations comprising Group V generally have long spells of low flow (group median LOWDUR of 13.95 days, where flows, in a year, are less than the 25th percentile of all flows across the record) and relatively short flood durations (group median HIGHDUR of 8.89). The high average (group median of 99.00) of hydrograph REVERSALS (change in rising and falling river levels) indicates that the flow regimes in this group generally respond rapidly to rainfall events and could equally be described as "flashy". Group V stations are all located in the high rainfall areas in the western region, extending from the Transkei to Limpopo Province.

- ***Sustained Baseflow***

In general, the eleven stations in **Group VI** record flows with a large "baseflow regime" (high Alt-BFI, with a group median of 0.56) and, not surprisingly, a low "high flow component" (HFI, group median of 10.44). Generally, stations in this group are more predictable and less variable than those in Group V and have shorter intervals between flood events (FLOODFREE group median of 13.00 compared with 19.50). The change in rising and falling river levels (REVERSALS) is lower than that for Group V stations (group median of 77.00 compared with 99.00) as a result of the sustained baseflow contribution to the streamflows recorded for stations in Group VI. With the exception of two stations (U2H006 and U2H007 located in KwaZulu-Natal) all stations in Group VI are located the closely together in the northernmost regions of the subcontinent in Mpumalanga and Limpopo provinces.

4.4 Investigating the Hydrological Indices: Methods Applied to the Working Database

PCA was extracted from the 74 by 74 correlation matrix of the 74 hydrological indices described in Section 3.3 of this Chapter using the Genstat Version 6 computer software. Following the methods described by Olden and Poff (2003), PCA was conducted to highlight general patterns of redundancy among the hydrological indices of different streamflow regimes found in South Africa and to ascertain subsets of indices describing the major sources of variation (in the indices). First, a PCA was performed using the streamflow data from the working database for the combined set of streams (*c.f.* Section 3.5 of this Chapter), then six separate PCAs were conducted for the six different streamflow types described in Section 4.3.2. The following Subsections 4.4.1 to 4.4.4 describe the methods applied. The results of the investigation are described in Section 4.5 of this Chapter, where subsections 4.5.1 to 4.5.4 link with subsections 4.4.1 to 4.4.4 respectively.

4.4.1 Statistical analysis of the principal components

Following Olden and Poff (2003), statistical significance of the principal component axes was evaluated using the Broken-stick model (Frontier, 1976). According to King and Jackson (1999) "Jackson (1993) identified the Broken-stick model to be a consistent approach for determining a suitable number of components for interpretation". The basis of the model is that the total variance is shared among the components and that the associated proportion of the expected eigenvalues follows a *broken stick distribution* (King and Jackson, 1999). This distribution is generated by considering "the variance shared among the principal axes to be a resource embedded in a stick of unit length" (Legendre and Legendre, 1998). For example, if the PCA had proportioned the variance among the principal axes at random, the "fractions of total variance would be about the same as the relative lengths of the pieces obtained by breaking the unit stick at random into as many pieces as there are axes" (Legendre and Legendre, 1998). In this application of the broken stick distribution, the expected values of the relative lengths of successively smaller pieces are given by,

$$b_k = \frac{1}{p}\sum_{i=k}^{p}\frac{1}{i}$$

where p is the number of variables and b_k is the extent of the eigenvalue for the kth component under the Broken-stick model. The observed eigenvalues computed by each PCA were compared with a decreasing list of the expected eigenvalues generated by the Broken-stick model. In accordance with King and Jackson (1999), the observed eigenvalues were considered to be meaningful when they were larger than the values generated by the model. In the PCA, the Genstat Version 6 software computed the eigenvectors (and loadings) of the hydrological indices on each of the principal components. Following Olden and Poff (2003), loadings of the hydrological indices on each significant component identified by the Broken-stick model were used to identify indices that explained the major sources of variation while minimising redundancy.

4.4.2 General patterns of inter-correlation, or redundancy, among the indices

Correlation among the hydrological indices was ascertained by examining the relationships among the hydrological indices in reduced space. Computing the correlations among the indices comprised scaling each eigenvector k to a length equal to its standard deviation, $\sqrt{\lambda_k}$ (Legendre and Legendre, 1998). Using this scaling for the eigenvectors, "the length of the descriptor-axes are their standard deviations in multidimensional space" and the combination of two descriptor axes (*i.e.* the cosine of the angle between their index-axes), "corresponds to their angle in the multidimensional space" and represents their correlation, r_{jl} (Legendre and Legendre, 1998).

4.4.3 Selecting high information, non-redundant indices

The analysis described in Sections 4.4.1 and 4.4.2 of this Chapter identified groups (or clusters) of indices that exhibited the largest absolute loadings on each significant principal component axis for (a) the combined set of streams and (b) the six distinct stream types. These indices contain the most information associated with the variation provided by all the indices in the dataset. However, as the principal components are linear combinations of the indices themselves and are, therefore, correlated, the significance of the index loadings cannot be tested using a routine statistical test for correlation coefficients (Legendre and Legendre, 1998; Olden and Poff, 2003). For their study, Olden and Poff (2003), addressed this anomaly by selecting at a least 25 indices (from a total of 171) with the highest absolute loadings on the significant principal component axes for each of the different stream types to produce a list of high information indices. In the present Study, fewer indices (74) were examined for redundancy and consequently fewer indices (12) were selected to produce a list of high information indices from which researchers can draw on for use in environmental flow studies of South African rivers. Following the procedure outlined by Olden and Poff (2003), this was achieved by setting the number of indices selected from each principal component "equal to the proportion of variation explained by the component compared to all significant axes" (Olden and Poff, 2003).

In addition to this selection procedure, high information, non-redundant indices were also chosen to represent each of the main facets of the streamflow regime. Following the procedure outlined by Olden and Poff (2003), this was achieved by selecting the index with the highest absolute loading on each of the significant principal component axes for each of the main streamflow characteristics. Given that the magnitude, duration, frequency, timing and rate of change in streamflow conditions are ecologically important (Richter *et al.*, 1996; 1997), researchers may wish to select indices representing one or more of these characteristics. Thus, knowledge of which indices of these streamflow characteristics contain high information (relating to variation in the complete set of indices), across the different streamflow types, may be useful for ecohydrological studies. For example, where the timing of flow events is known to be ecologically important, researchers could select the index of timing which contains the highest variation (in a statistical sense) among the indices representing a particular streamflow type. However, given that some categories of the main characteristics of streamflow regime, are poorly represented in the complete dataset (*i.e.* the magnitude of high flow conditions (only one index used in this Study) and the frequency of either high or low flow conditions (only two indices used in this Study for both categories), it was decided to reduce the number of characteristics representing different facets of the streamflow regime to seven (from 11, *c.f.* Section 3.3.2 of this Chapter) for the selection of high information, non-redundant, hydrological indices for individual streamflow types, *viz.*:

(a) The magnitude of average flow conditions ($n = 25$).

(b) The magnitude of either low or high flow conditions (n = 5). Technically, these flow conditions should comprise two separate groups. However, only one index of the magnitude of high flow conditions was investigated in this Study and it was considered that it was not unreasonable to combine these different characteristics since they are both distinct from the magnitude of average flow conditions.

(c) The duration of low flow conditions (n = 14).

(d) The duration of high flow conditions (n = 13).

(e) The timing of flow events (n = 7). Following Olden and Poff (2003), this sample size was too small to split into low and high flow categories.

(f) The frequency of flow events (n = 4). Again, this sample size was too small to split into low and high flow categories.

(g) The rate of change of flow events (n = 6).

This second procedure had the potential to provide a greater selection of high information indices (*i.e.* for each stream type, seven times the number of significant principal components rather than just 12 spread across the number of significant principal components), which are relatively non-redundant, and from which researchers could draw on for ecohydrological studies. Again the procedure followed that identified by Olden and Poff (2003).

4.4.4 Sensitivity of the indices to record length

The inter-annual variability of the 35 intra-annual indices (*c.f.* Figure 5.7) provides an indication of the length of record required to produce reliable site averages, or indices of central tendency (*i.e.* M_A1 to M_A12; M_L1 and M_L3; M_H1; D_L1 to D_L7; D_H1 to D_H6; T_L1 and T_H1; F_L1 and F_H1; R_A1 to R_A3) for use in water assessment and ecohydrological studies. This measure of *temporal variability* is assessed by the CV or, in this Study, the CD of the distribution of the annual observations of the intra-annual indices over the period of record. High values of CD indicate a greater likelihood of observing an annual value different from the long-term average (Clausen and Biggs, 2000).

On the other hand, the dispersion (using the CD) of the indices of the *inter-annual streamflow variability, predictability, seasonal predictability* or *overall variability* (*c.f.* Figure 5.7) across the different streamflow sites, and for which there is only one value across the record period for each site, represents a measure of the *hydrogeographical* (or *spatial) variability* among the different streamflow sites or regimes. High values of the CDs of these indices indicate a greater disparity among the sites than lower values. Nonetheless, comparison of these CD values, among different record lengths for each index, can also give an indication of the length of record required for the stability of these indices.

One of the aims of this Study is whether there are distinctions *among different flow streamflow types* regarding the length of record required to obtain reliable hydrological indices. In this Study, the length of record required to ensure stable estimates of the hydrological indices is investigated for the diversity of streamflow types located in South Africa.

Clausen and Biggs (2000) describe a method to indicate the degree of inter-annual variability of 34 streamflow variables, derived from daily mean flows from a common 7-year period, for 62 New Zealand perennial streams. As they were only interested in the overall behaviour of the streamflow variables, they standardised all seven annual values of the variables for each of the 62 sites by their at-site mean for the seven years. They then calculated the dispersion (in their case, the CV) of *all* the standardised annual values in order to "confound among-site and among year variation". Please note that the "standardisation" of the values of the hydrological indices referred to here and in any application in the study of the record length required for stable indices ***is different*** to that computed by the statistical software for the correlation matrix applied in the PCA analysis. In the latter case the hydrological indices are standardised so that they all contributed equally to the PCA and that the contributions are scale-dependent (Legendre and Legendre, 1998).

The sensitivity of the hydrological indices to record length in this Study was initially carried out on the entire set of 74 indices. The reason for this was that previous statements about record length requirements among different streamflow types in the region tend to be general, specifying neither the time step nor temporal resolution applied. However, this Study provided the opportunity to examine a large set of hydrological indices, derived from daily streamflow records, and which also represented a large range of intra-annual resolutions. Thereafter, the Study focuses on the indices of high information described in Section 4.5.3 of this Chapter.

The procedure described by Clausen and Biggs (2000), and which "confounds among-site and among year variation" as narrated above, was adopted and modified for the purposes of this Study. First, a "baseline" of the overall behaviour of the *hydrological indices of intra-annual variability* across the 36-year period from 1965 to 2000, was investigated for the entire set of 83 streams. For each site, all annual values for each of the 35 intra-annual indices (*c.f.* Section 3.4 of this Chapter) were standardised by their at-site median for the 36 years, and the Coefficient of Dispersion (CD) was calculated using these standardised values. Secondly, the behaviour of these hydrological indices among the diversity of streamflow types was investigated, across the 36-year period from 1965 to 2000, for different hydrogeographic regions. The same procedure to standardise the 36 annual values of each intra-annual index was applied before calculating the associated CDs. Thus, in the first analysis, the CD for each index was calculated across the standardised values for the entire set of streams, whereas in the second analyses six separate CDs for each index were calculated, one for each streamflow type. The number of standardised values used in the calculation of the CDs in each instance is provided in Table 5.11.

Table 5.11 Number, *n*, of standardised annual values applied in the sensitivity analyses of the indices of central tendency to record length at streamflow gauging sites in South Africa

	Streamflow Type						
	Extreme Seasonal	**Mixed**			**Perennial**		**All-Streams**
	Extreme Seasonal	**Short, Unseasonal Floods**	**Unpredictable Flow and Floods**	**Seasonally Predictable Flow and Floods**	**Runoff**	**Sustained Baseflow**	**All-Streams**
Entire set of 83 DWAF stations							
Number of sites	10	12	22	20	8	11	83
36 years	360	432	792	720	288	396	2988
Reduced Set of 43 DWAF stations							
Number of sites	4	5	8	10	7	9	43
42 years	168	210	336	420	294	378	1806
36 years	144	180	288	360	252	324	1548
20 years	80	100	160	200	140	180	860

Thirdly, since the inter-annual variability of hydrological indices can give an indication of the number of years that are needed for the central tendency (in this Study, medians) to stabilise (Clausen and Biggs, 2000), the length of record which is necessary to obtain reliable hydrological indices, with minimal influence of climatic variation, was investigated. Table 5.4 comprising the "best83" stations of the working database was re-examined for a suitable subset of those 83 DWAF gauging sites for such an investigation, in this case with less emphasis on obtaining the largest number of gauging stations with a common record, but rather on selecting the longest record span among an adequate number of stations for analysis. This resulted in a reduced set of 43 DWAF stations (Table 5.12) being identified as having records from 1959 to 2000 (*c.f.* Table 5.4).

Three different time spans of daily flow data for a common set of 43 DWAF stations were compared, *viz.*

(a) The 20-year record from 1 October 1981 to 30 September 2001;

(b) The 36-year record from 1 October 1965 to 30 September 2001 and

(c) The 42-year record from 1 October 1959 to 30 September 2001.

Again, standardised annual values of each of the 35 intra-annual indices were calculated for each of these record lengths for (i) the reduced dataset of 43 sites and (ii) the subsequently reduced datasets of sites representing the different streamflow types used in this Study, before the CDs of the standardised annual values were calculated. The number of standardised annual values applied in (i) and (ii) of this reduced set of sites is provided in Table 5.11.

Indices of streamflow variability (using CV or CD), predictability and seasonality (*c.f.* Figure 5.7) have been highlighted as being important for distinguishing between different streamflow regimes types, including the river types found in South Africa (Joubert and Hurly, 1994; Hughes and Hannart, 2003). Indices of the dispersion of the each of the 33 IHA intra-annual indices (M_A13 to M_A24; M_L2, D_L8 to D_L14; D_H7 to D_H12; T_L2; T_H2; F_L2; F_H2; R_A4 to R_A6) as well as indices of *predictability* (T_A1, and T_A2), *seasonal predictability* (T_H3 and D_H13) *and overall variability* (M_A25 and M_L4) are examined in this Study for the choice of hydrological indices that characterise streamflow regimes. Thus, it is pertinent to have information regarding the number of years required for these indices to stabilise across different streamflow types.

Table 5.12 Gauging stations comprising the "best43" stations used in this study

Gauging Station	Upstream Area (km^2)	Record Span Start Year*	Record Span End Year*	Longitude (degrees, decimal)	Latitude (degrees, decimal)	Weir Site	1994-Study Group	Revised Group
		*hydrological yr						
A4H002	1777	1948	2000	28.083	-24.267	Mokolorivier	B1	III
A5H004	629	1956	2000	28.400	-23.967	Palalarivier	G3	III
A9H003	62	1931	2000	30.517	-22.883	Tshinanerivier	C1	VI
A9H004	320	1932	2000	30.367	-22.767	Mutalerivier	B3	VI
B1H002	252	1956	2000	29.333	-25.817	Spookspruit	G4	II
B1H004	376	1959	2000	29.167	-25.667	Klipspruit	C2	IV
B6H001	518	1910	2000	30.800	-24.667	Blyderivier	C4	V
B6H003	92	1959	2000	30.800	-24.683	Treurivier	C5	VI
C5H007	348	1923	2000	26.317	-29.133	Renosterspruit	A2	I
C7H003	914	1947	2000	27.283	-27.350	Heuningspruit	A3	I
C8H003	806	1954	2000	28.933	-27.833	Corneliusrivier	G5	II
D5H003	1509	1927	2000	20.350	-31.800	Visrivier	A4	I
G1H008	395	1954	2000	19.067	-33.300	Klein	G6	IV
H1H007	84	1935	2000	19.133	-33.567	Witrivier	E1	IV
R1H014	70	1953	2001	26.933	-32.633	Tyumerivier	H11	III
R2H001	29	1946	2001	27.283	-32.717	Buffelsrivier	H12	III
R2H006	119	1948	2001	27.367	-32.850	Mgqakweberivier	H13	III
R2H008	61	1947	2001	27.367	-32.767	Quencwerivier	H14	II
T3H004	1029	1947	2001	29.417	-30.567	Mzintlavarivier	E3	IV
T4H001	715	1951	2000	29.817	-30.733	Mtamvunarivier	F1	V
T5H004	545	1949	2000	29.467	-29.767	Mzimkulurivier	C9	V
U2H006	339	1954	2000	30.267	-29.367	Karkloofrivier	E6	VI
U2H007	358	1954	2000	30.150	-29.433	Lionsrivier	C10	VI
U2H011	176	1958	2000	30.250	-29.633	Msundusirivier	C11	IV
U4H002	316	1949	2000	30.617	-29.150	Mvotirivier	C14	IV
V1H001	4176	1925	2000	29.817	-28.733	Tugelarivier	E7	V
V1H009	196	1955	2000	29.767	-28.883	Bloukransrivier	G11	II
V3H002	1518	1929	2000	29.933	-27.600	Buffelsrivier	B6	I
V3H007	129	1948	2000	29.833	-27.833	Ncandurivier	G12	III
V6H003	312	1954	2000	30.133	-28.300	Wasbankrivier	H15	IV
V6H004	658	1954	2000	30.000	-28.400	Sondagsrivier	G14	IV
W1H004	20	1948	2000	31.450	-28.867	Mlalazirivier	H16	II
W4H004	948	1950	2000	30.850	27.517	Bivanerivier	E8	V
W5H006	180	1950	2000	30.833	-27.100	Swartwaterrivier	C18	IV
X2H005	642	1950	2000	30.967	-25.417	Nelsrivier	E9	IV
X2H008	180	1948	2000	30.917	-25.783	Queensrivier	C19	III
X2H010	126	1948	2000	30.867	-25.600	Noordkaaprivier	E10	VI
X2H012	91	1956	2000	30.250	-25.650	Dawson/'sspruit	C20	III
X2H013	1518	1959	2000	30.700	-25.433	Krokodilrivier	E11	V
X2H014	250	1959	2000	30.700	-25.367	Houtbosloop	C21	VI
X2H015	1554	1959	2000	30.683	-25.483	Elandsrivier	C22	V
X3H001	174	1948	2000	30.767	-25.088	Sabierivier	C23	VI
X3H003	52	1948	2000	30.800	-24.983	Mac-Macrivier	C24	VI

First, the variation of each of these hydrological indices (*i.e.* the indices of inter-annual variability, predictability, seasonal predictability and overall variability) was assessed to obtain a baseline of the **overall behaviour of the indices**, at a broad regional scale. This was achieved by calculating the dispersion, CD, of the at-site values, derived from the 36 year record (1965 to 2000) across the 83 sites. Secondly, this procedure was repeated for comparisons of the variation among the different streamflow types for a finer spatial analysis, by calculating the CD of the at-site values within each of the streamflow types. Thirdly, the dispersion of the at-site indices derived from the 42 year, 36 year and 20 year records was calculated for the reduced set of 43 DWAF gauging stations, before assessing the record length required to stabilise the indices for each of the different streamflow types. The number of CD values, n, applied in this component of the analysis are provided in Table 5.13.

Table 5.13 Number, n, of values applied in the sensitivity of the indices of streamflow variability and of streamflow predictability to record length at streamflow gauging sites in South Africa

	Streamflow Type						
	Extreme Seasonal	**Mixed**				**Perennial**	**All-Streams**
	Extreme Seasonal	**Short, Unseasonal Floods**	**Unpredictable Flow and Floods**	**Seasonally Predictable Flow and Floods**	**Runoff**	**Sustained Baseflow**	**All-Streams**
Entire set of 83 DWAF stations							
Number of sites	10	12	22	20	8	11	83
36 years	10	12	22	20	8	11	83
Reduced Set of 43 DWAF stations							
Number of sites	4	5	8	10	7	9	43
42, 36 and 20 years	4	5	8	10	7	9	43

4.5 Results of the Analyses of the Working Database

This Section comprises the results of the five parts of the Analyses described in Section 4.4. As most of the methods applied in the present study followed those of the Olden and Poff Study (Olden and Poff, 2003), the results of the present study and, to a certain extent the terminology used, are presented in a similar way to the Olden and Poff study. The benefits of this approach are discussed in Section 5 of this Chapter and in Chapter 7.

4.5.1 The meaningful principal components

The results from the PCA of the 83 stream sites, based on 74 hydrological indices, are shown in Table 5.14. The number of statistically significant principal component axes, using the proportions computed by the Broken-stick model, ranged from four (*e.g.* the Extreme Seasonal group) to six (*e.g.* the Runoff group). The first six principal components (PC1, PC2, PC3, PC4, PC5 and PC6) explained 31.62%, 15.10%, 10.70%, 7.78%, 5.69 and 4.85% respectively of the variation in the hydrological indices and together explained 75.74% of the variation for the combined set of stream types, "All-83 Streams". However, the first two principal components explained only 47% of the variation for the combined set of "All-83 Streams". This is not surprising, considering the diversity represented among the stream sites and the high number of indices used in the Study. In studies involving many descriptors (in this Study, hydrological indices), the first two principal components usually do not account for a large fraction of the variability (Legendre and Legendre, 1998).

Table 5.14 Results from the principal component analysis on the correlation matrix of 74 hydrological indices based on 83 stream sites grouped into six streamflow types

Stream flow type	Principal Component (% variation explained)						Total
	1	2	3	4	5	6	
Extreme Seasonal	34.67	23.66	17.54	7.99	-	-	83.86
Mixed: Short, Unseasonal Floods (M:SUF)	27.77	21.84	14.73	9.21	7.31	5.46	86.32
Mixed: Unpredictable Flow and Floods (M:UFF)	32.79	18.34	12.01	9.27	7.94	-	80.35
Mixed: Seasonally Predictable Flow and Floods (M:SPFF)	25.05	20.47	14.38	9.18	6.02	5.43	80.53
Perennial: Runoff (P: R)	36.90	25.84	15.93	7.82	5.63	5.30	97.42
Perennial: Sustained Baseflow (P:SB)	41.41	25.20	13.14	7.07	-	-	86.82
All-83 Streams	31.62	15.10	10.70	7.78	5.69	4.85	75.74

While these results may initially appear to be of limited use, the PCA can still be used to answer other questions of ecological interest. For example:

(a) The environmental "gradient" of the streamflow types found in South Africa, distinguished by different characteristics of their hydrological regimes, was discussed in Section 4.3.1 of this Chapter and shown in the scatter plot of Figure 5.9;

(b) Plotting the positions of the hydrological indices-axes in the plane of the first two principal axes reveals important inter-correlations, or redundancies, of the hydrological indices of the combined set of All-83 streams (Figure 5.11). In addition, Figure 5.11 shows which indices contribute the most to the formation of the reduced space in the PCA for All-83 streams; and

(c) The main objective of the present Study (*i.e.* to identify high information, non-redundant, hydrological indices that explain a dominant proportion of the statistical variation provided by the database of hydrological indices for the combined set of streamflow types, as well as each of the six distinct streamflow types) can be achieved by focusing on the hydrological indices with the highest absolute loadings on each statistically significant principal component as in Olden and Poff (2003).

4.5.2 General patterns of inter-correlation among the indices

Figure 5.11 shows the ordination from the PCA of the 83 stream sites ("All-83 Streams") based on 74 indices, plotted in the plane determined by the first two principal component axes. The co-ordinates of the indices are the apices of the eigenvectors, plotted as a function of $\sqrt{\lambda}$ (*i.e.* each eigenvector was rescaled to length $\sqrt{\lambda_k}$, the square root of the k-th eigenvalue). In Figure 5.11, the correlation among indices is given by the angle between the index-axes (*i.e.* between the vectors joining the origin and the position of the apex of the descriptor axis in reduced space) rather than the proximity between the apices of their axes (Legendre and Legendre, 1998; Olden and Poff, 2003). Consequently, as explained by Olden and Poff (2003), indices separated by small angles (*e.g.* indices M_A1 and M_A2 with high loadings on PC 1, as shown in Figure 5.11), are highly positively correlated; indices separated by angles close to 180^{O} (*e.g.* D_L6 and D_L7 with opposite loadings on PC1 and PC 2, as shown in Figure 5.11), are highly negatively correlated; and indices separated by a right angle (*e.g.* indices T_L1 and T_A2, as shown in Figure 5.11) are not correlated.

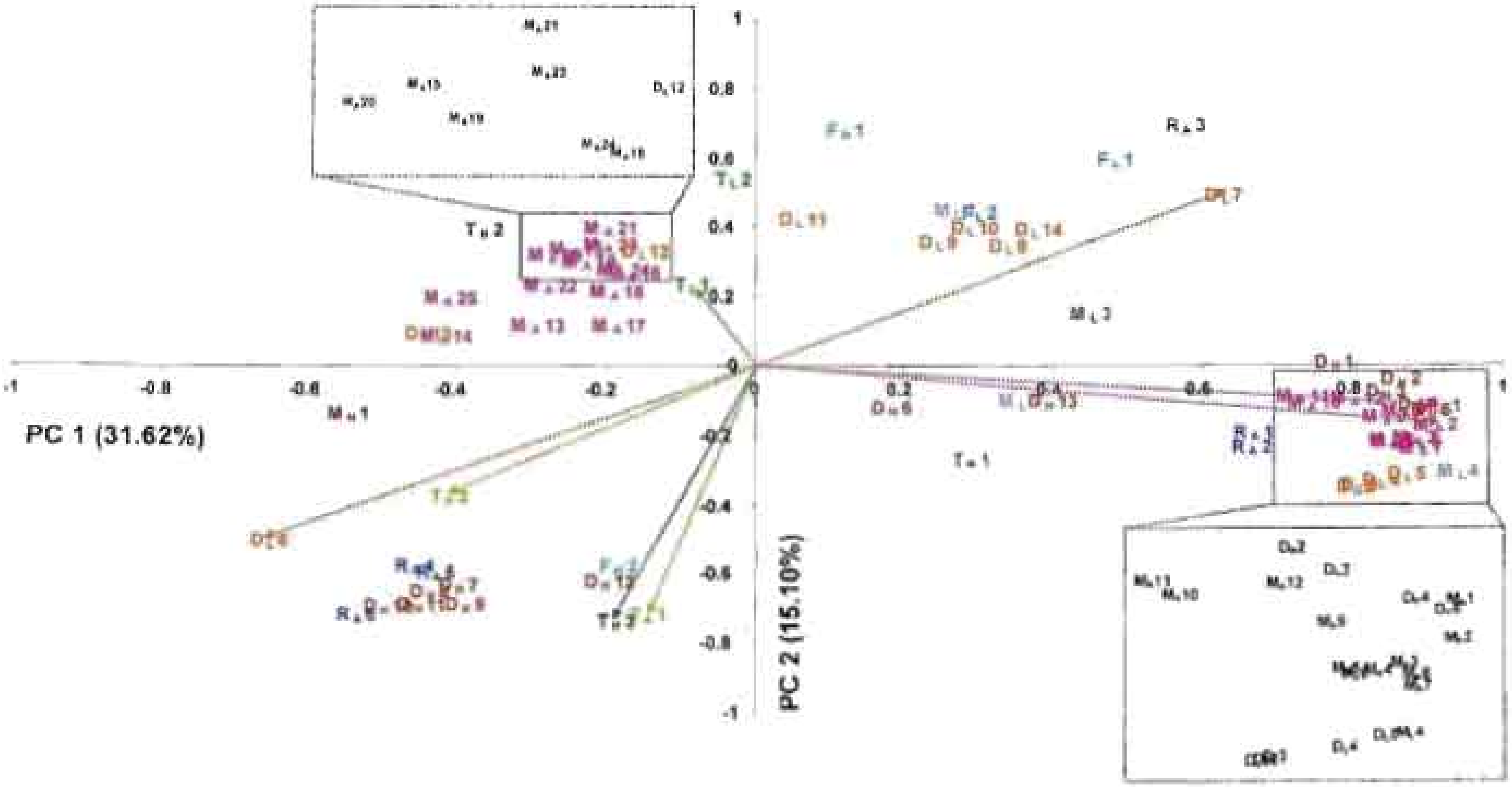

Figure 5.11 Ordinations of the 74 hydrological indices, from the PCA of 83 stream sites in South Africa, in the plane of the first two principal component axes. Correlations among the indices are interpreted as the cosine of the angle separating their index-axes. Each eigenvector was rescaled to the length $\sqrt{\lambda_k}$ to display the correlations among the indices. Selected index axes have been plotted and are described in the text. Some of the data have been shown in enlargement boxes for clarity.

As in Olden and Poff (2003), the positions of the index apices shown in Figure 5.11 indicate that many of hydrological indices are highly inter-correlated. Figure 5.11 shows that, for the combined set of streamflow regimes (All-83 Streams), the highest correlation is generally, although not exclusively, among indices of central tendency describing certain streamflow characteristics, and includes indices of the following streamflow conditions:

(a) The *magnitudes of average flow conditions* (*i.e.* the mean monthly streamflows);

(b) A range of *durations of high flow conditions* (*i.e.* the 1-day, multiple-day and seasonal high flow events);

(c) A range of *durations of low flow conditions* (*i.e.* the 1-day, multiple-day and seasonal low flow events);

(d) The *magnitude of low flow conditions* (*i.e.* a flow threshold that is equaled or exceeded 75 per cent of the time, Q75); and

(e) The *rates of change of average flow conditions* (*i.e.* the rise and fall rates of river levels).

These indices (M_A1-M_A12; D_H1-D_H5; D_L1-D_L5; M_L4 and R_A1-R_A2) are clustered (Cluster 1) in the lower right quadrant of Figure 5.11, with high loadings on the first principal component (PC1), and are highly positively correlated with each other.

There is also inter-correlation among a cluster of indices shown in the bottom left quadrant of Figure 5.11. The *variability in the duration of high flow conditions* (D_H7, D_H8, D_H9, D_H10 and D_H11) and the *variability in the rates of change* of rising and falling river levels (R_A4, R_A5 and R_A6) have fairly high loadings on the second principal component (PC2) and are, highly positively correlated with each other (Cluster 2). Cluster 2, which comprises dispersion-based indices, is in a different quadrant to Cluster 1, and, thus, the indices in the two different clusters are negatively correlated.

Figure 5.11 also indicates a cluster of correlated dispersion-based indices in the upper left quadrant (Cluster 3). The indices in Cluster 3 represent the *variability in magnitudes of average flow conditions* for several calendar months; in particular, M_A16, M_A21, M_A23 and M_A24 which represent the variability in flows for January, June, August and September respectively. These indices mostly have low to medium loadings on both PC1 and PC2 and are positively correlated with each other. In addition, index D_L12 (the variability in the 90-day minimum flow) is inter-correlated with these indices. This is not surprising since June, August and September are among the main low flow season months for many streamflow regimes in South Africa represented in the dataset (*i.e.* those in summer rainfall regions), whereas January is the main low flow month for those rivers in the dataset located in winter rainfall regions). The indices in Cluster 3 are uncorrelated to the indices in Cluster 2 and are negatively correlated to the indices in Cluster 1.

The upper right quadrant in Figure 5.11 shows another cluster of indices of dispersion for the magnitude, frequency and duration of low flow conditions (M_L2; F_L2 and D_L8-D_L10). The indices in this cluster (Cluster 4) have moderate loadings on both PC1 and PC 2 and are negatively correlated to the indices in Cluster 2 and uncorrelated to the indices in both Clusters 1 and 3.

In addition, there are a number of smaller groups of correlated indices. These include the following:

(a) The predictability of flows (T_A1) and the seasonal predictability of flooding (T_H3);

(b) The variability in both the number of high pulses (F_H2) and their durations (D_H12);

(c) The constancy of flows (T_A2), as an indicator of the predictability of the seasonality of the flow regime, and the duration of extreme low flow (no flow), as represented by the number of days in a year with zero flow (D_L6), which indicates the relevance of periods of no flow in the ecological functioning of many of South Africa's rivers;

(d) The frequency of high pulses (F_H1) and the variability in the 30-day minimum flow (D_L11):

(e) Rates of change in streamflow conditions represented by the number of hydrograph reversals (R_A3) and the frequency of low pulses (F_L1); and

(f) The IHA baseflow index (M_L1) and the seasonality of non-flooding (D_H13).

As in the Olden and Poff study (Olden and Poff, 2003), there are number of indices which are closer to the origin and are generally uncorrelated with the other indices that have higher loadings on the first two principal components. These include the high pulse durations (D_H6), the timing of Julian date of the minimum flow (T_L1) and the variability in the magnitude of average flows in October (M_A13) and February (M_A17).

4.5.3 High information, non-redundant indices

The first two principal components do not account for a large fraction of the variation within the dataset of hydrological indices for the combined set of All-83 Streams nor for the datasets for the six streamflow regime types (Table 5.14). Indeed, the proportion of variation explained by the first two principal components for the dataset representing Group IV, *viz.* those streams with Seasonally Predictable Flow and Floods, is lower than that of All-83 Streams. Nonetheless, PCA can still be used to identify high information indices through the process described in Section 4.4.3 of this Chapter.

Table 5.15 shows the 12 hydrological indices (discussed in Section 4.4.3 of this Chapter) with the largest absolute loadings on each of the statistically significant principal components. The number of indices shown for each principal component was equivalent to the proportion of variation explained by the component compared to all significant axes as in Olden and Poff (2003). Selecting indices on each of the statistically significant principal components ensures that the subsets of indices (for each principal component) are relatively independent of each other (Olden and Poff, 2003). Thus, as in Olden and Poff (2003), Table 5.15 indicates the groups of high information indices that represent the major gradients of variation described by the database for the different streamflow types.

All seven main characteristics of the streamflow regime are represented in Table 5.15. Some of the high information hydrological indices in Table 5.15 are common to more than one streamflow type. These are, most notably, D_H4 (30-day annual maximum flow) and D_H8 (variability in the 3-day annual maximum flow), M_A1 (average flows in October) and M_A25 (the CDB, based on the Desktop Reserve model index of overall variability *c.f.* Section 3.3.2 of this Chapter), each of which has four occurrences in Table 5.15. Indices D_H3 (7-day annual maximum), D_H5 (90-day annual maximum) and D_L6 (number of days with zero flow) each have three occurrences among the different streamflow types. In general, indices of the magnitude of average flow conditions have the greatest representation in Table 5.15. However, this is more than likely as a result of the higher incidence of these indices in the dataset compared with other indices. Dominant indices representing the duration of high flow conditions are also strongly represented across most of the different streamflow types, particularly for sites with more predictable flow and

Table 5.15 Hydrological indices with the largest absolute loadings on each statistically significant principal component. Streamflow types are based on a revised classification of the streamflow types found in South Africa by Joubert and Hurley (1994). Indices annotated with "–" have a negative loading of the index on the principal component, although it is the magnitude of the loading rather than its direction that is relevant to this analysis. Different colours represent the 7 main characteristics of the streamflow regime.

Principal Component	Streamflow Type						
	Extreme Seasonal	Mixed			Perennial		All Streams
	Extreme Seasonal	Mixed: Short, Unseasonal Floods	Mixed: Unpredictable Flow and Floods	Mixed: Seasonally Predictable Flow and Floods	Perennial: Runoff	Perennial: Sustained Baseflow	All-83 Streams
1	D_L6	D_L12-	F_H1	D_H4-	M_A1	M_L3	M_A1
	M_A1-	M_A24-	M_A25-	M_A25	D_H3	D_H2-	M_A2
	D_H4-	[illegible]	D_L12-	D_H5-	D_H4	D_H3-	D_H5
	M_A6-	D_L6-	D_H12-	D_H3-	[illegible]	D_H4-	M_A7
	D_H5-		M_A24-			M_A13-	M_A8
						D_H1-	
2	M_H1	M_A1-	M_A15	M_A12	D_L3	D_L4	T_H3-
	[illegible]	M_A9-	M_A11	M_A11	D_L1	F_L2-	T_H1
	D_H8-	M_A10-	M_A10	T_H1-	D_L2	D_L3	
3	F_H2	M_A17	M_A2-	D_L1	M_A21-	D_H8	M_A25
	T_L1-	D_H7	D_L5-	D_L2	M_A20	F_H3	M_L1
	T_L3						
4	M_A25	T_H2	D_L6-	D_L8	M_A15	D_L14	D_L10
5		T_H2	T_H1	D_H8-	D_H8-		M_A22
6		M_A13		D_L7	F_H1		T_H2-

flood regimes. This feature may also have been affected by the high incidence of streams in the dataset which record perennial flow ($n = 43$).

High information indices for All-83 Streams

Table 5.15 indicates that where there is little knowledge of streamflow regime type or pattern, researchers could choose from several high information indices which represent most of the main characteristics of the streamflow regime. M_A1, M_A2, M_A7 and M_A8 are all high information indices (average monthly flows for October, November, April and May respectively) for the combined set of All-83 Streams. Of course, it should be remembered that where the indices are derived from the same component axis, as indeed these are (*i.e.* all from PC1, *c.f.* Table 5.15), they are not independent from each other, and choices may need to be made. However, any of these indices could be paired with M_A25 (the CDB, based on the Desktop Reserve model index of overall variability) as statistically important measures of the magnitude of the streamflow conditions exhibited by the wide diversity streamflow regimes in the combined set of All-83 Streams. Colwell's predictability indices (T_A1, T_A2) are both dominant indices, whereas another index of the timing of flows, T_H3 (the seasonal predictability of flooding), also contains high information. In general, indices of central tendency are sufficient for "first-estimate", or low confidence, descriptors of the flow regimes found in South Africa. However, at this spatial resolution the variability associated with the flows in July (M_A22) and the 7-day minimum (D_L10) are also high information indices.

Indices for Extreme Seasonal streamflow regimes

Unsurprisingly, the number of days with zero flow (D_L6) and the timing of the minimum flow (T_L1; T_L2) are important indices for describing the dominant patterns of hydrological variability of Extreme Seasonal regimes (Table 5.15). The magnitude of average flows in October (M_A1) and March (M_A6) are good contenders for riverine studies in which the demarcation of the "start" or "end" of season flows are ecologically important. On the other hand, indices describing monthly (30 days) or seasonal (90 days) high flows (*i.e.* D_H4; D_H5), the high peak flows (M_H1), the variability in high flow pulses (F_H2) and the variability in the 3-day maximum flow (D_H8) are important measures of the high flow conditions of this streamflow type. In general, indices of *central tendency* are sufficient measures for describing the variation in the indices for this streamflow type. However, in addition to indices F_H2 and D_H8 already mentioned, the variability in the rise rate of streamflows (R_A4) and the CDB, as an index of overall variability, (M_A25) may be important for eco-hydrological studies of Extreme Seasonal streamflow regimes.

Indices for Short, Unseasonal Floods streamflow regimes

As expected, indices regarding the timing (T_H2, the variability of date of the annual maximum flow event) and duration (D_H7, variability of the 1-day maximum flow event) of the high flow events, and the (un)predictability of flow conditions (T_A2) are important measures of the variation in the indices for Short, Unseasonal Flood regimes (Table 5.15). The variability associated with changes between rising and falling river level (R_A6) is a good contender for describing the "flashy" nature of these streamflow regimes.

In general, hydrological indices describing the dispersion (*i.e. variability)* in different streamflow components are more appropriate than indices of the *central tendency* for explaining the variation in the indices for Short, Unseasonal Flood regimes. Nonetheless, either the number of days with zero flow (D_L6) or the variability of the 90-day minimum flow (D_L12) could be selected as an important measure of low flow conditions. The magnitude of flows in October (M_A1), June (M_A9) and July (M_A10) all contain high information, but are derived from the same component axis and, as such, are not independent of each other. On the other hand, the measures of the variability in the magnitude of streamflows in October (M_A13), February (M_A17) and September (M_A24) are all high information and relatively independent indices of average flow conditions. The sites in this streamflow group are located throughout the sub-continent, in both summer and winter rainfall regions, indicating the importance of maintaining the diversity of the streamflow regime at the height of the wet season and at the beginning of the dry season. There are no indices of frequency included in the "highest information" indices for this streamflow type (Table 5.15).

Indices for Unpredictable Flow and Floods streamflow regimes

Given their high information nature, the number of high pulses (F_H1), the variability associated with their durations (D_H12) and the predictability of flow events (T_A1) are important indices of the dominant patterns of hydrological variability for Unpredictable Flow and Flood regimes (Table 5.15). In addition, the overall index of variability (M_A25) is a good candidate to account for the hydrological variation represented by streamflow regimes in this streamflow type. The importance of the number of days with zero flow

(D_L6), and of the variability of the 90-day minimum flow (D_L12) is evident for Unpredictable Flow and Flood regimes. In addition, D_L5, the seasonal low flow conditions, also contains relatively independent, high information. Researchers could choose to select the monthly flow for November (M_A2) and either July (M_A10) or August (M_A11) as important indices of the magnitude of average flow conditions. Again, M_A24 and M_A19 (the variability associated with monthly flow conditions in September and also in April) are dominant indices.

In general, indices of dispersion are less appropriate than indices of the central tendency of flow conditions for explaining the dominant patterns of hydrological variability for the Unpredictable Flow and Flood streamflow regimes. There are no indices relating to the rate of change included in the "highest information" indices for these streams (Table 5.15).

Indices for Seasonally Predictable Flow and Floods streamflow regimes

As expected, indices describing weekly (7-day), monthly (30-day) and seasonal (90-day) high flows (*i.e.* D_H3, D_H 4 and D_H 5), and the timing of the maximum annual flow event (T_H1), are important for Seasonally Predictable Flow and Floods regimes (Table 5.15). The overall variability index (M_A25) accounts for much of the variation (in the indices tested) associated with streams in this streamflow type. The magnitude of average flows in August (M_A11) and September (M_A12) are also good contenders for eco-hydrological studies. D_L1 and D_L2, representing the shorter minimum flow events, are also dominant indices, as is the duration of low pulses (D_L7).

Indices of dispersion are less appropriate than indices of central tendency for explaining the dominant patterns of the hydrological variability for these streamflow regimes, with only the variability in the shorter low (D_L8) or high (D_H8) flows being dominant indices. There are no indices relating to the frequency or rate of change of streamflows included in the "highest information" indices for this streamflow type (Table 5.15).

Indices for Runoff streamflow regimes

These river systems are characterised by indices describing a "flashy" response to changes in climatic conditions. As expected, the frequency of high pulse counts (F_H1) contains high information regarding the patterns of hydrological variability associated with these perennial "runoff" streams (Table 5.15). The index for average October flows (M_A1) could be critical for describing the first of season flushing flows required by such river systems. Indices representing weekly (7-day) and monthly (30 days) extreme high flows (*i.e.* D_H3 and D_H4) as well as the fall rate of river levels (R_A2) also appropriate for describing the dominant patterns of hydrological variability for Runoff regimes. The shorter minimum flow events (D_L1, D_L2 and D_L3) are also high information indices.

Only four indices of dispersion are regarded as being of high information: the variability associated with the flows in the start-of-winter low flow months of May (M_A20) and June (M_A21) as well as the start-of-summer high flow month of December (M_A15) and the variability in the 3-day maximum flow (D_H8). There are no indices of timing included in the "highest information" indices for this streamflow type (Table 5.15).

Indices for Sustained Baseflow streamflow regimes

Unsurprisingly, the dominant index describing the hydrological variability among Sustained Baseflow regimes is M_L3, the Alt-BFI based on the Desktop Reserve model index of short-term variability, representing "baseflow", the low amplitude and frequently occurring part of the hydrograph (*c.f.* Section 2.4.3 of this Chapter). However, with the exception of the seasonal high flow conditions, the maxima flows (D_H1, D_H2, D_H3 and D_H4) are also dominant indices associated with this streamflow type. Indices of low flow events are less relevant, but researchers could chose from either weekly (D_L3) or monthly (D_L4) minimum flows. Indices of dispersion are well represented in the "high information" selection with the variability in October flows (M_A13), low pulse counts (F_L2) as well as their durations (D_L14), 3-day maximum (D_H8) and the predictability of seasonal flooding (T_H3) being dominant.

Indices representing the main facets of the streamflow regime

The PCA can also be used to identify high information, non-redundant indices which represent all the major streamflow characteristics of magnitude, duration, timing, frequency and rate of change for each streamflow type. This was achieved by selecting the index with the highest loading on each of the significant principal component axes *for each of the seven main characteristics of the streamflow regime*. Selecting an index from each of the significant principal components does not result in an increase in redundancy *per se*. However, as in Olden and Poff (2003), there may be some redundancy among the selected indices representing different flow characteristics (*c.f.* Table 5.15).

Using the Extreme Seasonal streamflow type as an example, for each of the seven major streamflow characteristics, the representative index with the highest loading on each of the first four principal components axes was selected since the PCA had identified four statistically significant principal components for this streamflow regime type (*c.f.* Table 5.14).

The results of this analysis are provided in Table 5.16, which highlights four to six indices for each streamflow type, that are relatively independent from each other, and that represent the seven main streamflow characteristics. Where there are only a few indices in the dataset to represent a particular characteristic of the streamflow regime (*e.g.* the frequency of flow events, $n = 4$), there is high commonality of these indices across all stream types. However, it can be seen that the dominance of some indices is common across most of the stream types.

Magnitude of average flow conditions

M_A1 (average flow in October) is a high information, relatively non-redundant, index for Extreme Seasonal, Short, Unseasonal Floods and Runoff streamflow types as well as for the combined set of All-83 Streams (Table 5.16). Not only does this emphasise the relevance of the first of season flushing flows for streams in summer rainfall areas (a feature that is acknowledged in environmental flow assessment studies in South Africa), but suggests that these flows are also important to other river systems in the country. Several high information, non-redundant, calendar month indices are specific to single streamflow types, for example M_A2 (November), M_A7 (March) and M_A10 (July) for Unpredictable Flow and Flood regimes; M_A12 (September) for Seasonally Predictable Flow and Flood regimes; and M_A11 (August) for Runoff regimes.

In general, indices of dispersion of "monthly" indices contain more information about the hydrological variability associated with South Africa's rivers than indices of central tendency of monthly flows. In particular, M_A16 (representing the variability in flows in January) is a high information, non-redundant, index for the Short, Unseasonal Floods, Seasonally Predictable Flow and Floods, Runoff, and Sustained Baseflow streamflow types. These streamflow types represent very different flow regimes and spatial conditions. However, the incidence of this index, as being of high information, across the dataset also confirms the relevance of maintaining "close to natural" variability in flows

Table 5.16 Hydrological indices with the highest absolute loadings on each of the four to six principal component axes for streamflow types in South Africa. Indices are assigned to seven main streamflow characteristics in accordance with the largest loadings exhibited on each significant component. Superscripts denote the first to sixth principal components. Plain font denotes indices of central tendency (medians). Bold font denotes indices of dispersion (CD). Some indices are highlighted. Purple denotes indices used in the Desktop version of the South African Building Block Methodology (see text); blue denotes Colwell's indices of Predictability; green denotes the seasonal predictability of flooding; yellow denotes the seasonal predictability of non-flooding.

Streamflow Characteristic	Streamflow type						
	Extreme Seasonal	Mixed			Perennial		All Streams
	Extreme Seasonal	Short, Unseasonal Floods	Unpredictable Flow and Floods	Seasonally Predictable Flow and Floods	Runoff	Sustained Baseflow	All-83 Streams
Magnitude of flow events							
Average flow conditions	M_A1^1 $\mathbf{M_A19}^2$ $\mathbf{M_A22}^3$ [illegible]	$\mathbf{M_A24}^1$ M_A1^2 $\mathbf{M_A17}^3$ $\mathbf{M_A16}^4$ $\mathbf{M_A14}^5$ M_A13^6	[illegible] $\mathbf{M_A19}^2$ M_A2^3 M_A7^4 M_A10^5	[illegible] M_A12^2 $\mathbf{M_A21}^3$ $\mathbf{M_A16}^4$ $\mathbf{M_A21}^5$ $\mathbf{M_A14}^6$	M_A1^1 M_A11^2 $\mathbf{M_A21}^3$ $\mathbf{M_A15}^4$ $\mathbf{M_A24}^5$ $\mathbf{M_A16}^6$	$\mathbf{M_A13}^1$ $\mathbf{M_A18}^2$ $\mathbf{M_A15}^3$ $\mathbf{M_A16}^4$	M_A1^1 $\mathbf{M_A21}^2$ [illegible] $\mathbf{M_A24}^4$ $\mathbf{M_A22}^5$ M_A17^6
Low or high flow conditions	[illegible] M_H1^2 [illegible] [illegible]	M_L1^1 [illegible] $\mathbf{M_L2}^3$ M_L1^4 [illegible] M_H1^6	[illegible] [illegible] [illegible] $\mathbf{M_L2}^4$ M_H1^5	[illegible] M_H1^2 [illegible] $\mathbf{M_L2}^4$ M_H1^5 M_H1^6	[illegible] [illegible] $\mathbf{M_L2}^3$ $\mathbf{M_L2}^4$ M_H1^5 $\mathbf{M_L2}^6$	[illegible] [illegible] [illegible] M_H1^4	[illegible] $\mathbf{M_L2}^2$ M_L1^3 M_H1^4 $\mathbf{M_L2}^5$ $\mathbf{M_L2}^6$
Duration of flow events							
Low flow conditions	D_L6^1 D_L3^2 D_L5^3 $\mathbf{D_L13}^4$	D_L12^1 D_L5^2 $\mathbf{D_L9}^3$ $\mathbf{D_L14}^4$ $\mathbf{D_L13}^5$ D_L7^6	$\mathbf{D_L12}^1$ D_L4^2 D_L5^3 D_L6^4 D_L1^5	$\mathbf{D_L11}^1$ $\mathbf{D_L14}^2$ D_L1^3 $\mathbf{D_L8}^4$ D_L7^5 D_L7^6	$\mathbf{D_L8}^1$ D_L3^2 $\mathbf{D_L11}^3$ D_L1^4 D_L7^5 $\mathbf{D_L9}^6$	$\mathbf{D_L12}^1$ D_L4^2 D_L2^3 $\mathbf{D_L14}^4$	D_L5^1 D_L6^2 $\mathbf{D_L12}^3$ $\mathbf{D_L10}^4$ $\mathbf{D_L8}^5$ D_L8^6
High flow conditions	D_H4^1 $\mathbf{D_H8}^2$ $\mathbf{D_H13}^3$ $\mathbf{D_H13}^4$	D_H3^1 D_H5^2 $\mathbf{D_H7}^3$ D_H6^4 $\mathbf{D_H12}^5$ $\mathbf{D_H11}^6$	$\mathbf{D_H12}^1$ $\mathbf{D_H7}^2$ D_H4^3 $\mathbf{D_H9}^4$ D_H1^5	D_H4^1 $\mathbf{D_H12}^2$ $\mathbf{D_H11}^3$ D_H6^4 $\mathbf{D_H8}^5$ D_H13^6	D_H3^1 $\mathbf{D_H10}^2$ D_H13^3 D_H6^4 $\mathbf{D_H8}^5$ D_H6^6	D_H2^1 $\mathbf{D_H12}^2$ $\mathbf{D_H8}^3$ $\mathbf{D_H9}^4$	D_H5^1 D_H10^2 D_H2^3 $\mathbf{D_H12}^4$ D_H8^5 D_H13^6
Timing of flow events	[illegible] T_H1^2 T_L1^3 [illegible]	$\mathbf{T_L2}^1$ $\mathbf{T_L2}^2$ [illegible] [illegible] $\mathbf{T_H2}^5$ [illegible]	$\mathbf{T_L2}^1$ T_H1^2 [illegible] [illegible] [illegible]	[illegible] T_H1^2 [illegible] $\mathbf{T_H2}^4$ T_L1^5 $\mathbf{T_L2}^6$	[illegible] [illegible] T_L2^3 [illegible] T_L1^5 $\mathbf{T_L2}^6$	[illegible] T_L1^2 [illegible] $\mathbf{T_L2}^4$	[illegible] [illegible] [illegible] [illegible] T_H1^5 [illegible]
Frequency of flow events	$\mathbf{F_H2}^1$ F_H1^2 $\mathbf{F_H2}^3$ F_H1^4	F_L1^1 F_H1^2 F_H1^3 F_L1^4 $\mathbf{F_H2}^5$ $\mathbf{F_H2}^6$	F_H1^1 $\mathbf{F_L2}^2$ $\mathbf{F_L2}^3$ $\mathbf{F_H2}^4$ F_L1^5	$\mathbf{F_H2}^1$ $\mathbf{F_H2}^2$ F_L1^3 F_H1^4 $\mathbf{F_L2}^5$ F_L1^6	$\mathbf{F_H2}^1$ F_L1^2 $\mathbf{F_L2}^3$ F_H1^4 $\mathbf{F_L2}^5$ F_H1^6	F_H1^1 $\mathbf{F_L2}^2$ F_H1^3 $\mathbf{F_H2}^4$	F_L1^1 F_H1^2 F_L1^3 $\mathbf{F_L2}^4$ $\mathbf{F_H2}^5$ $\mathbf{F_L2}^6$
Rate of change in flow events	R_A3^1 $\mathbf{R_A4}^2$ R_A6^3 R_A2^4	$\mathbf{R_A6}^1$ $\mathbf{R_A4}^2$ $\mathbf{R_A5}^3$ R_A3^4 R_A3^5 R_A3^6	$\mathbf{R_A6}^1$ R_A3^2 R_A2^3 $\mathbf{R_A5}^4$ R_A2^5	R_A2^1 $\mathbf{R_A4}^2$ $\mathbf{R_A4}^3$ $\mathbf{R_A6}^4$ $\mathbf{R_A5}^5$ $\mathbf{R_A5}^6$	R_A2^1 $\mathbf{R_A4}^2$ $\mathbf{R_A6}^3$ R_A3^4 R_A3^5 R_A1^6	R_A2^1 $\mathbf{R_A5}^2$ R_A3^3 $\mathbf{R_A6}^4$	R_A2^1 $\mathbf{R_A6}^2$ R_A1^3 $\mathbf{R_A4}^4$ $\mathbf{R_A4}^5$ R_A1^6

for this "first-of-high-flow month" for many river systems in the South Africa. Similarly, variability in flows in September (M_A24) is also a common index of high information across the dataset, having relevance for the Short, Unseasonal Floods, and Runoff streamflow types, as well as All-83 Streams. This indicates the importance of natural variability in flows in the driest month of the year for many river systems. Table 5.16 indicates that there are other indices of the variability in monthly flows that are dominant for more than one streamflow type, but they are not discussed further here. However, M_A18 (the variability in flow in March) is specific to Sustained Baseflow, indicating the relevance of the natural variability of in-channel flows at the end of the high flow season for these perennial rivers in summer rainfall regions. Index M_A25 (CDB, the Desktop Reserve model index representing overall variability in the streamflow regime) is also a high information, non-redundant, index for describing the hydrological variability for streams at the more arid end of the environmental gradient of streamflow types, as well as for the combined set of "All-83 Streams".

Magnitude of low or high flow conditions

M_H1 (the HFI representing the median of high flow conditions) is the most common high information, non-redundant, index of the magnitude of either low or high flow conditions across all streamflow types (Table 5.16). This is all the more pertinent since it is the only high flow index of the five indices of these conditions in the dataset, indicating its relevance for describing the hydrological variability for a diversity of South African rivers.

The importance of M_L1 (the IHA "baseflow" index) is shared among the Short, Unseasonal Floods streamflow type and the All-83 Streams, yet the importance of the variability associated with this streamflow component (M_L2) is shared across four of the streamflow types as well as All-83 Streams. However, M_L2 may be not be important for the two streamflow types at either extreme of the streamflow type range, *i.e.* Extreme Seasonal and Sustained Baseflow (Table 5.16).

On the other hand, M_L4 (index Q75, *c.f.* Section 2.4.3 of this Chapter) is also a common dominant index of low flow conditions. M_L4 appears to be an important indicator for all streamflow types except for Extreme Seasonal regimes, where no flow occurs for most of the year. M_L3 (index Alt-BFI, the Desktop Reserve model index representing the baseflow component, and relatively short-term variability in the streamflow regime) is a dominant index for all the distinct streamflow types (Table 5.16).

Duration of low flow conditions

The variability associated with long spells of extreme low flow (described by D_L13, variability in the annual number of days with zero flow) is important for describing the hydrological variability associated with the "harsher" streamflow types (Table 5.16). D_L5 and D_L12 (the 90-day minimum flow and variability thereof) are the most common high information, non-redundant, indices of the duration of low flow conditions across the different stream flow types. However, these indices may not be important for characterising the Seasonally Predictable Flow and Flood, or Runoff streamflow types or the combined set of All-83 Streams. Other high information indices of the duration of low flow disturbance which are shared across streamflow types include D_L6 (days of zero flow) for Extreme Seasonal, Unpredictable Flow and Flood and All-83 Streams; D_L7 (duration of low pulses) for Short, Unseasonal Floods, Seasonally Predictable Flow and Flood as well as Runoff streamflow types and D_L1 and D_L8 (the annual minimum flow and variability thereof) for Seasonally Predictable Flow and Floods and Runoff streamflow types.

On the other hand, D_L2 (3-day minimum flow) is specific to the Sustained Baseflow streamflow type, D_L3 (7-day minimum flow) is specific to Runoff streamflow type while D_L10 (variability in the 7-day average flow) is specific to All-83 Streams (Table 5.16).

In general, indices of the dispersion of the duration of low flow conditions are more appropriate than indices of the central tendency (median) for describing the hydrological variability associated with the Short, Unseasonal Floods and Seasonally Predictable Flow and Flood streamflow types as well as the combined set of All-83 Streams. Indices of the dispersion of the duration of low flow conditions are under-represented in Table 5.16 for the Extreme Seasonal as well as Unpredictable Flow and Flood streamflow types.

Duration of high flow conditions

D_H12 (variability in high flow pulses) is the most common high information, non-redundant, index of the duration of high flow conditions across the different streamflow types and, with the exception of the Extreme Seasonal and Runoff types, is dominant across all streamflow types, including the combined set of All-83 Streams (Table 5.16).

Other dominant indices of the duration of high flow disturbance which are common among different streamflow types include D_H13 (the seasonal predictability of non-flooding) for Extreme Seasonal, Seasonally Predictable Flow and Floods, Runoff and All-83 Streams and D_H8 (variability in the 3-day maximum flow) for Extreme Seasonal, Seasonally Predictable Flow and Floods, Runoff, Sustained Baseflow and All-83 Streams. Thus, with the exception of Sustained Baseflow regimes, D_H8 and D_H13 have an affinity to the same streamflow types. Of the duration of high flow disturbance indices, D_H1 (the annual maximum flow) and D_H10 (variability in the 30-day maximum flow) are both specific, as dominant indices, to one streamflow type, being the Unpredictable Flow and Floods and Runoff streamflow types respectively. Indices of variability in the duration of high flow conditions are as under-represented in Table 5.16 for the Extreme Seasonal streamflow regimes as those of the variability in the duration of low flow conditions.

Timing of flow events

T_A2 (the proportion of predictability attributed to constancy) is a high information index among all streamflow types, with the exception of the Extreme Seasonal streamflow type. With the exception of the Short, Unseasonal Floods and Unpredictable Flow and Flood streamflow types, T_L1 (the Julian date of the annual minimum flow) also has wide commonality as a dominant index across streamflow types. Predictability of flow (T_A1) is dominant index for the Extreme Seasonal, the Short, Unseasonal Floods and Unpredictable Flow and Floods streamflow types, but not for the Seasonally Predictable Flow and Floods, Runoff and Sustained Baseflow streamflow types, where the constancy component of Colwell's Index of predictability is more important. The Julian date of the annual maximum flow (T_H1) and the variability associated with this flow characteristic (T_H2) have varying degrees of commonality across all streamflow regimes, except for the Runoff and Sustained Baseflow streamflow types (Table 5.16). T_H3 (the predictability of flooding) is a dominant index for the streamflow types at the "wetter" end of the environmental gradient of streamflow types, having commonality for both the Runoff and Sustained Baseflow streamflow types, yet is also important for All-83 streams.

Frequency of flow events

Indices of the frequency of flow events are poorly represented in the dataset. However, there are still some salient points to be gained from the analysis of high information, non-redundant indices of these flow events for the different streamflow types. The importance of indices F_H1 and F_H2, relating to the frequency of high flow pulses, is common across all the streamflow regime types (Table 5.16). This is in contrast to the indices relating to the frequency of low flow pulses (F_L1 and F_L2). F_L1 and F_L2 are inconsequential for the Extreme Seasonal streamflow type, since the value of the 25th percentile of average daily streamflows across the records at the sites in this group is invariably zero. Index F_L1 is less relevant than F_H1 for describing Sustained Baseflow streamflow regimes.

Rate of change in flow events

R_A6 (variability around the number of hydrograph reversals) is a high information, non-redundant index of the rate of change in flow events, across all streamflow types as well as the combined set of All-83 Streams (Table 5.16). R_A1 (the rate of rise in river level) is dominant only for the streamflow regimes of the Runoff streamflow type and for the combined set ofAll-83 Streams.

General Observations

Overall, M_L3, M_H1, F_H1, F_H2, and R_A6 have the greatest commonality, as high information indices, non-redundant across all the streamflow types (Table 5.16). Indices M_A3, M_A4, M_A6, M_A8, M_A9 (average of flow in December, January, March, May and June respectively) as well as M_A20 and M_A23 (variability in flow in May and August) do not feature in the list of high information indices (*i.e.* with the highest loading) on each of the four to six significant principal component axes for any of the streamflow types (Table 5.16). However, these may still be important indices for certain stream types (*e.g.* M_A6 has the fourth highest loading on the first principal component axis for the Extreme Seasonal streamflow type, Table 5.15). The salient point that can be gleaned from this feature and from Table 5.16 is that while average monthly flows, for a particular calendar month, may represent an informative index for specific streamflow types, indices of the *variability* in monthly flows are more appropriate for providing information of the different streamflow types comprising the dataset. Unsurprisingly, D_H1 (annual maximum flow) has little commonality as a high information, non-redundant index across the diversity of streamflow regimes found in South Africa.

END OF VOLUME 1 OF 2

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THE HYDROLOGICAL BASIS FOR THE PROTECTION OF WATER RESOURCES TO MEET ENVIRONMENTAL AND SOCIETAL REQUIREMENTS

by

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(Continued from Chapter 5 on page 5-89 at the end of Volume 1 of 2)

4.5.4 The length of record required for stable indices

The investigation of how many years of record are needed to provide reliable hydrological indices of the different streamflow types found in South Africa, as detailed in Section 4.4.4 of this Chapter, produced large quantities of information as a result of the high number of indices (74) across the different streamflow types (six distinct types as well as the combined set of All-83 Streams). However, since the principal aim of this Chapter is the identification of reduced sets of high information hydrological indices which adequately represent the different facets of the streamflow regimes found in South Africa, the main focus of the results reported here relates to the indices of high information identified in Section 4.5.3 of this Chapter. Nonetheless, for the sake of completeness, some of the Figures referred to in this Section show all the 74 indices analysed.

It is important to remember that, following Clausen and Biggs (2000), the analysis of the length of record required for reliable assessments of the hydrological indices applied in this Study was conducted to assess the *overall behaviour* of the indices at the resolutions of the broad, or coarse, scale of the combined set of streams and at the resolution of the streamflow type, and not the variation at individual sites.

First, the indices of central tendency are addressed (*c.f.* Figure 5.7). In the second place, the indices of dispersion (*inter-annual streamflow variability*), *streamflow predictability*, *seasonal predictability* and *overall variability* are considered (*c.f.* Figure 5.7). Together, the Coefficients of Dispersion of these groups of indices describe the temporal and spatial variability of the different streamflow regimes found in South Africa (*c.f.* Section 4.5.3 of this Chapter).

4.5.4.1 Indices of central tendency

Overall behaviour of the hydrological indices

The Coefficients of Dispersion (CDs) of standardised annual values of the 35 intra-annual indices across the entire set of 83 streamflow sites (All-83 Streams), as well as across each of the different streamflow types, are shown in Figures 5.12 to 5.16 (*c.f.* Table 5.11 for numbers of standardised values applied) for the magnitude, duration, timing, frequency and rates of change of streamflow conditions.

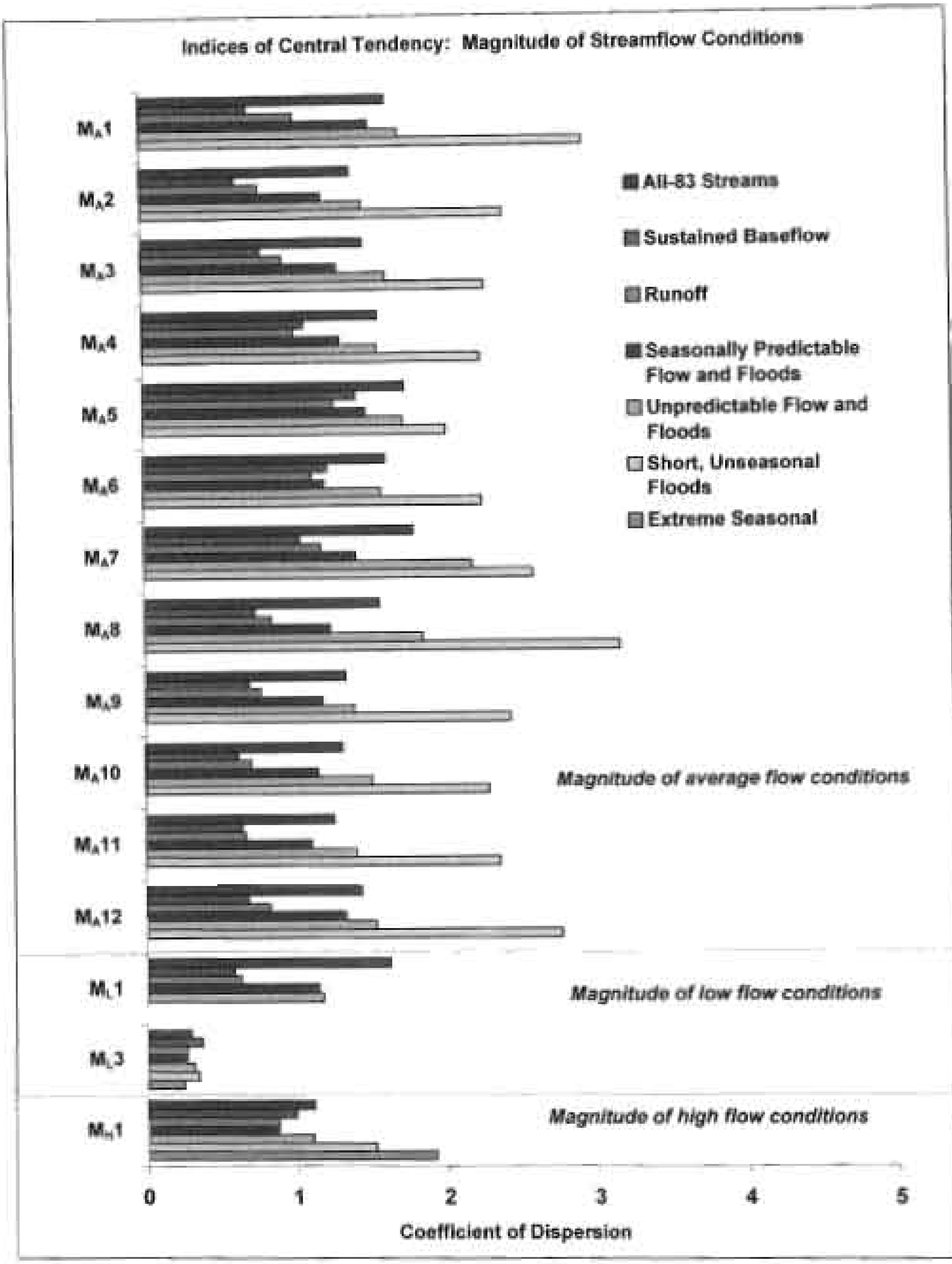

Figure 5.12 Coefficient of Dispersion of standardised annual values of the intra-annual indices describing the magnitude of flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types. See Table 5.11 for sample sizes.

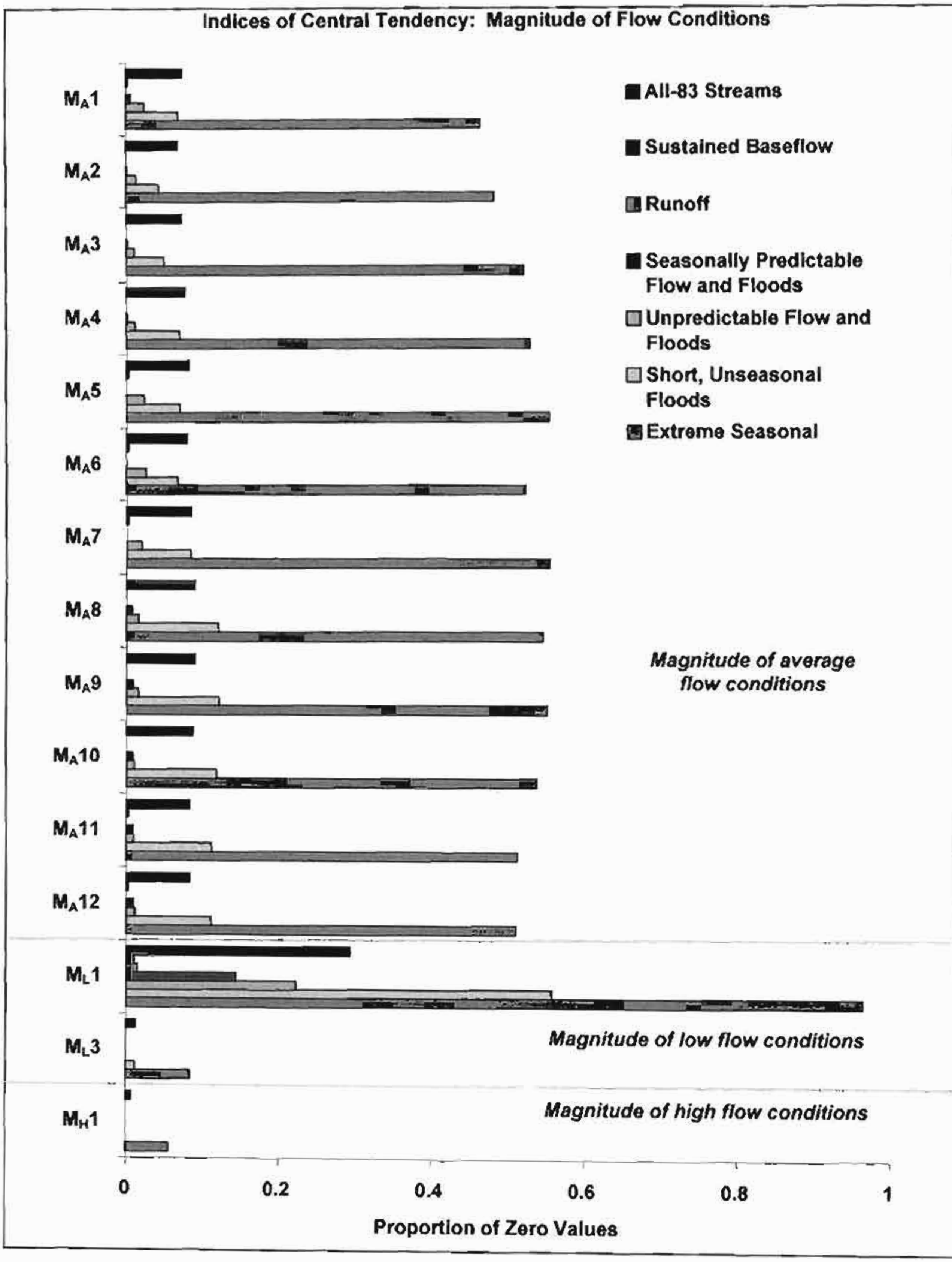

Figure 5.13 Proportion of Zero Values of standardised annual values of the intra-annual indices describing the magnitude of flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types. See Table 5.11 for sample sizes.

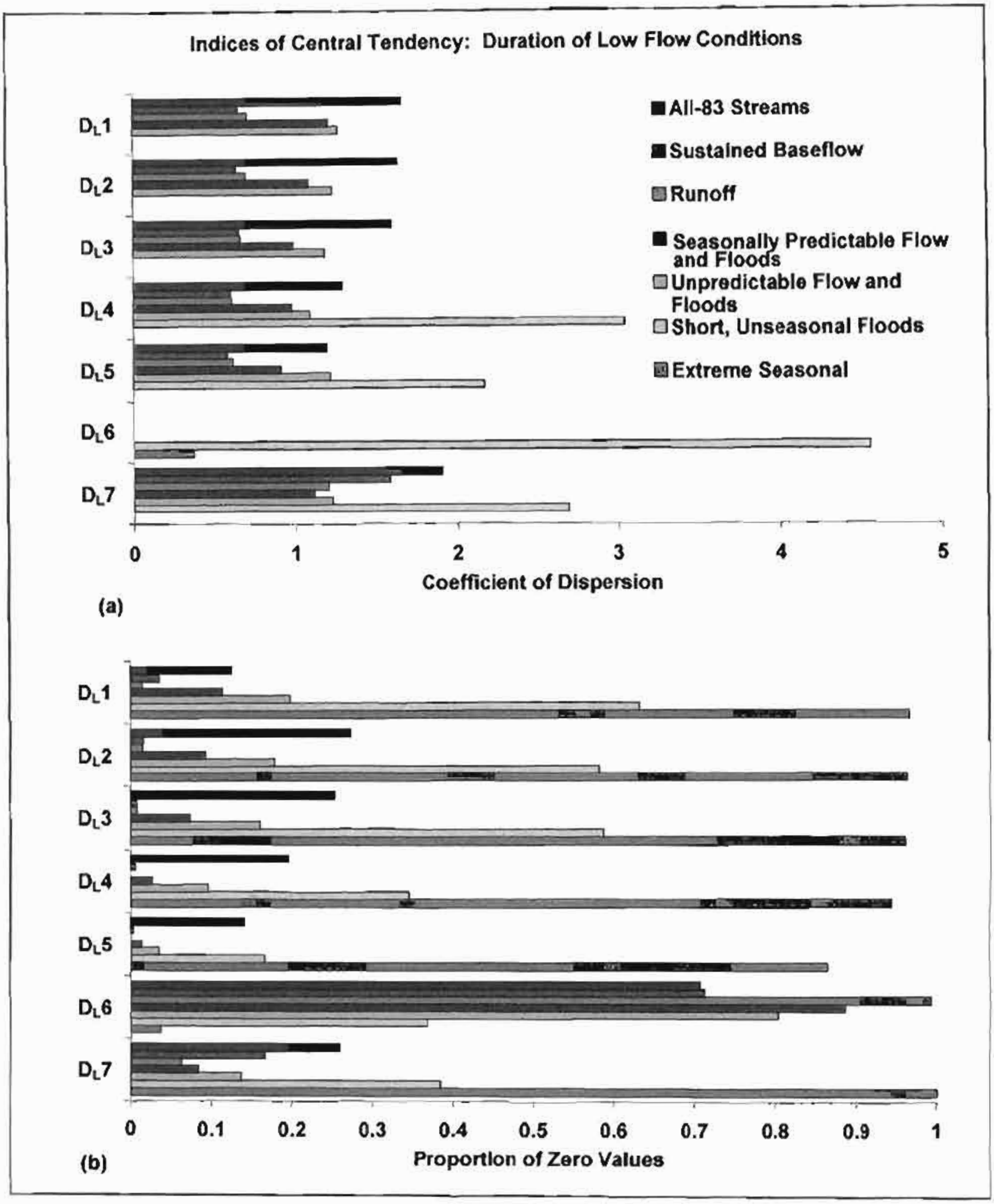

Figure 5.14 (a) Coefficient of Dispersion of standardised annual values of the intra-annual indices describing the duration of low flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types and (b) Proportion of Zero Values. See Table 5.11 for sample sizes.

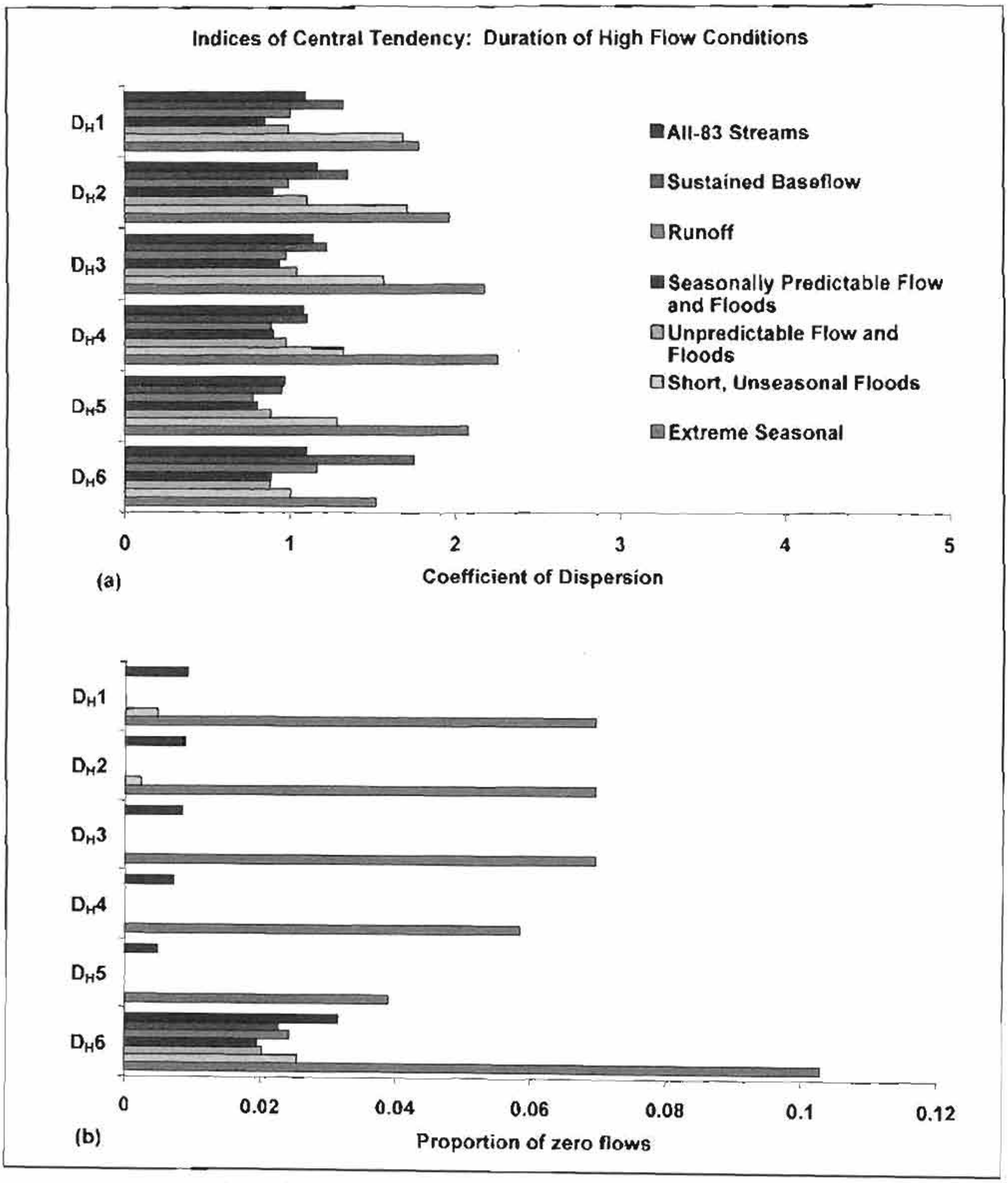

Figure 5.15 (a) Coefficient of Dispersion of standardised annual values of the intra-annual indices describing the duration of high flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types and (b) Proportion of Zero Values. See Table 5.11 for sample sizes.

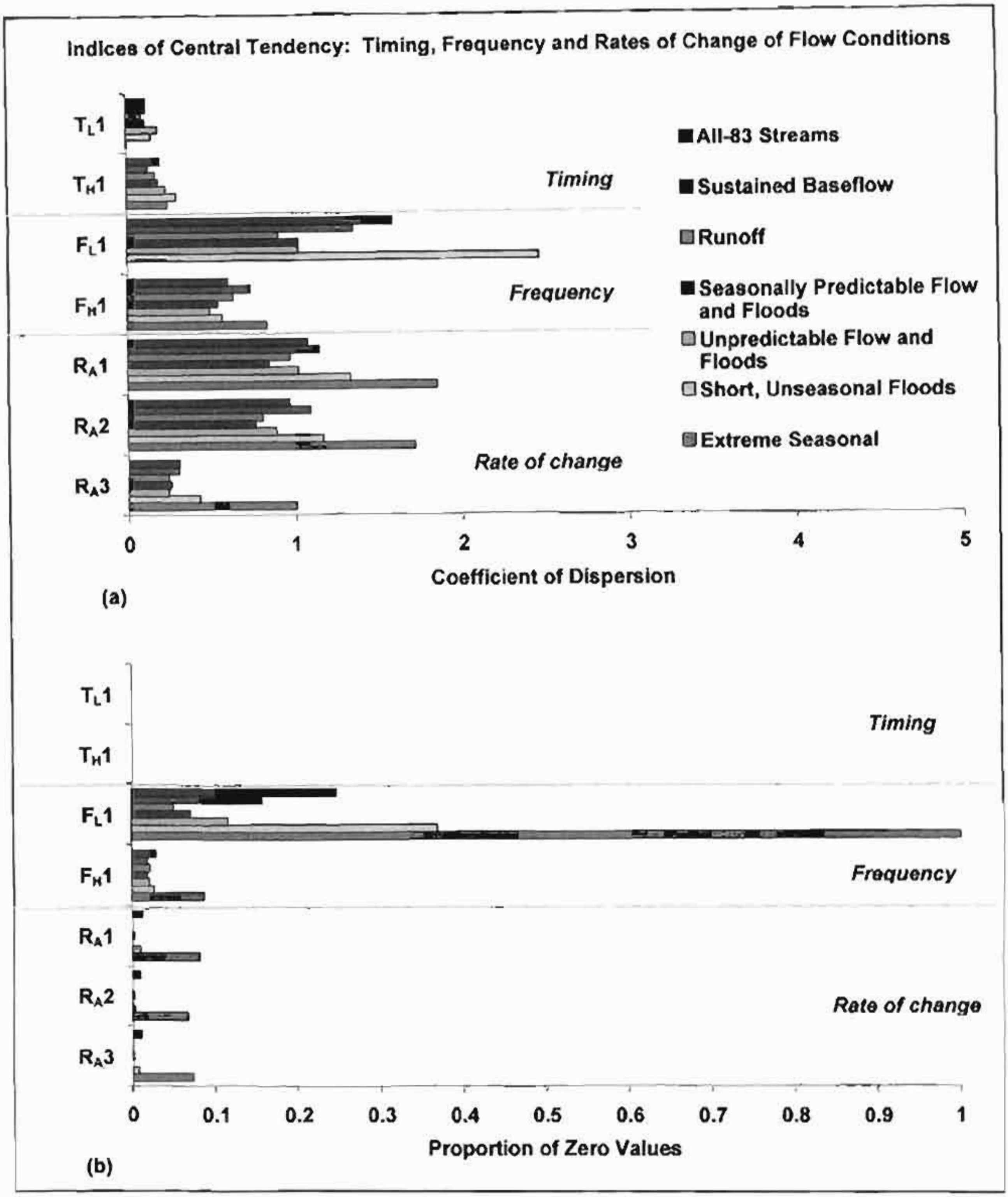

Figure 5.16 (a) Coefficient of Dispersion of standardised annual values of the intra-annual indices describing the timing, frequency and rate of change flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types and (b) Proportion of Zero Values. See Table 5.11 for sample sizes.

The proportion of standardised annual values which are zero (see Clausen and Biggs, 2000) is relatively high for several of the indices describing the magnitude, duration and frequency of low flow conditions (*c.f.* Figures 5.13, 5.14b, 5.15b and 5.16b) as a result of the occurrence of such streamflow conditions at either end of the ephemeral to perennial environmental gradient. Consequently, the derivation of the CD of standardised annual values, applying the method described in Section 4.4.4 of this Chapter, is problematic since, in many instances, the median of the standardised annual values is zero. The indices thus affected are shown in Table 5.17. For All-83 Streams, the most notable is D_L6 (number of days with zero flow, proportion of zero values of 0.71) and, to a lesser extent, the M_L1 (IHA baseflow index, proportion of zero values of 0.29) and F_L1 (number of low pulses, proportion of zero values of 0.25) as indicated respectively on Figures 5.14(b), 5.13, 5.16(b). Indeed, the high proportion of zero values, not only for M_L1 and F_L1, but also for D_L1 to D_L5 and D_L7 (the minima flow conditions and duration of low pulses), among the streams in the Extreme Seasonal group and, to a lesser extent, the Short, Unseasonal flood group presented anomalies in the calculations of their CDs. Consequently these indices are not particularly useful for describing the streamflow patterns in either of these groups, a feature which can be confirmed by their low rankings on each statistically significant principal component axis in the PCAs conducted for these streamflow types. It should be noted that while minimum flow conditions and the duration of low pulses are, undoubtedly, important to the ecology of these streamflow types, the indices M_L1,F_L1, D_L1 to D_L5 and D_L7 are so consistent among the streams in these respective streamflow types (*i.e.* all streams in both of the groups have low values for these conditions), that eco-hydrological studies should focus on different, high information indices of low flow conditions which distinguish between the different streams in each of these groups (*i.e.* D_L6, the number of days with zero flow as shown in Table 5.15).

Alternatively, the calculation can result in extremely high CD values. For example, the CD value for M_A2 (average flow in November) representing the Extreme Seasonal streamflow type is 66.7. In order to provide meaningful schematic representations of this part of the analyses, some of the indices have been omitted from the figures. However, the omissions do not necessarily detract from the results of the analyses, but rather emphasise the sensitivity of different streamflow regimes with regard to obtaining reliable hydrological indices.

Table 5.17 Indices for which the derivation of the CD of standardised annual values, applying the method described in Section 4.4.5 of Chapter 5 is problematic and / or results in extremely high values

Streamflow Characteristic	Streamflow Type						
	Extreme Seasonal	Mixed			Perennial		All Streams
	Extreme Seasonal	Short, Unseasonal Floods	Unpredictable Flow and Floods	Seasonally Predictable Flow and Floods	Runoff	Sustained Baseflow	All-83 Streams
Magnitude of average flow conditions	M_A1 to M_A12	D_L1 to D_L3	-	-	-	-	-
Magnitude of extreme flow conditions	M_L1	M_L1	-	-	-	-	M_L1
Duration of low flow conditions	D_L1 to D_L5; D_L7	D_L1 to D_L5; D_L7	D_L6	D_L6	D_L6	D_L6	D_L6
Duration of high flow conditions	-	-	-	-	-	-	-
Timing of flow conditions	-	-	-	-	-	-	-
Frequency of flow conditions	F_L1	F_L1	-	-	-	-	F_L1
Rate of change of flow conditions	-	-	-	-	-	-	-

The CD values for All-83 Streams indicate that, in general, longer records are required for M_A1 through M_A12, the calendar month indices as measures of the magnitude of average flow conditions (Figure 5.12), M_L1 (the IHA baseflow) as a measure of the magnitude of low flow conditions (also Figure 5.12), the duration (Figure 5.14) as well as the frequency (Figure 5.16) of low flow conditions than the records required for other indices. M_L3 (the Alt-BFI, Figure 5.12), T_L1 and T_H1 (the Julian dates of the annual minimum and maximum flows, Figure 5.16) as well as R_A3 (the number of hydrograph reversals, Figure 5.16) all require much shorter records.

Comparison among the different stream types

As expected, Figures 5.12 to 5.16 show that the inter-annual dispersion of the indices is generally greater for the Extreme Seasonal streams than the other streamflow types and is generally lowest for the streams in the Runoff and Sustained Baseflow streamflow types. Differences in M_A1 to M_A12 (average monthly flows) show that streams in the Sustained Baseflow group also have the least inter-annual dispersion, whereas the streams in Extreme Seasonal group have the greatest (Figure 5.12). Thus, for the most part, longer records are necessary for the assessment of indices which describe the hydrological variability associated with ephemeral and intermittent streams than are required for perennial streams.

There are, however, exceptions to this broad behaviour. Longer records may be required for the reliability of M_L3 (the Alt-BFI) for the Sustained Baseflow streams than the other stream types. Generally, streams in the Sustained Baseflow group also require longer records than those in the Runoff group and the Seasonally Predictable Flow and Floods group for stability of the M_H1, (HFI) and longer records than those in the Runoff, Seasonally Predictable Flow and Floods and Unpredictable Flow and Floods groups for stability of D_L7 (duration of low pulses), D_H1 to D_H5 (multi-day maxima), F_L1 (frequency of low pulses) and R_A1 to R_A3 (the rates of change in flow conditions). Perennial streamflow regimes in the Runoff and Sustained Baseflow groups) generally require longer records than the streams in the mixed streamflow groups (Seasonally Predictable Flow and Floods, Unpredictable Flow and Floods and Short, Unseasonal Floods groups) for reliable estimates of D_H6 (the duration of high pulses) and F_H1 (number of high pulses).

Together these features confirm the sensitivity of streams at both ends of the environmental gradient of the streamflow types (*i.e.* the spectrum ranging from Extreme

Seasonal to Sustained Baseflow) to inter-annual variation in the either the high or low streamflow components.

Comparisons of record lengths

The Coefficients of Dispersion, calculated from standardised annual values of each of intra-annual indices, across different record lengths, are shown on Figures 5.17 to 5.19 for the main streamflow characteristics of a reduced number of 43 DWAF gauging stations (*c.f.* Section 4.4.4 of this Chapter). The figures compare the overall behaviour of the indices, at a broad spatial scale, over the different record lengths and give an indication of the number of years of record required to stabilise each index. Given the differences among the streamflow types, regarding the likelihood of an annual value being different from the long term average in the 36-year record described above, the information provided in the Figures 5.17 to 5.19 may be perceived to be of limited use. However, the information does have value where the "classification" of the streamflow regime is either uncertain or unknown. Figures 5.17 to 5.19 show varying degrees of difference for the length of record required for stable indices, although it does appear that at least one climatic event has been excluded from the 20-year record which, in addition, has greater influence on indices derived from the 36-year record than from the 42-year record. The(se) climatic event(s) have most impact on the inter-annual variability in flow conditions in October, December, January, April and July (M_A1, M_A3, M_A4, M_A8 and M_A10 Figure 5.17). For example, the impacts of (separate) high flow events on the inter-annual statistical properties of flows in October and April, across the different record lengths, are shown in Figure 5.20. Figure 5.20(a) shows that while the median October flows are very similar across the different record lengths (with a value close to 1.00, as they should be in accordance the "standardised computation"), the 75th percentile is greatest across the 36 year record. However, the statistical properties of the 42-year record have a mitigating effect on these events, resulting in a lower value for the 75th percentile. On the other hand, the 42 year record is not sufficient to mitigate the effects of a high flow event on the statistical properties of the monthly flows in April (Figure 5.20b).

Notwithstanding the above findings, the following observations can be deduced for the overall behaviour of the indices of central tendency. There is least difference among the record lengths for M_L3 (the Alt-BFI), indicating that the 20-year record is just as useful as longer records. In general, there is little difference among the record lengths required for

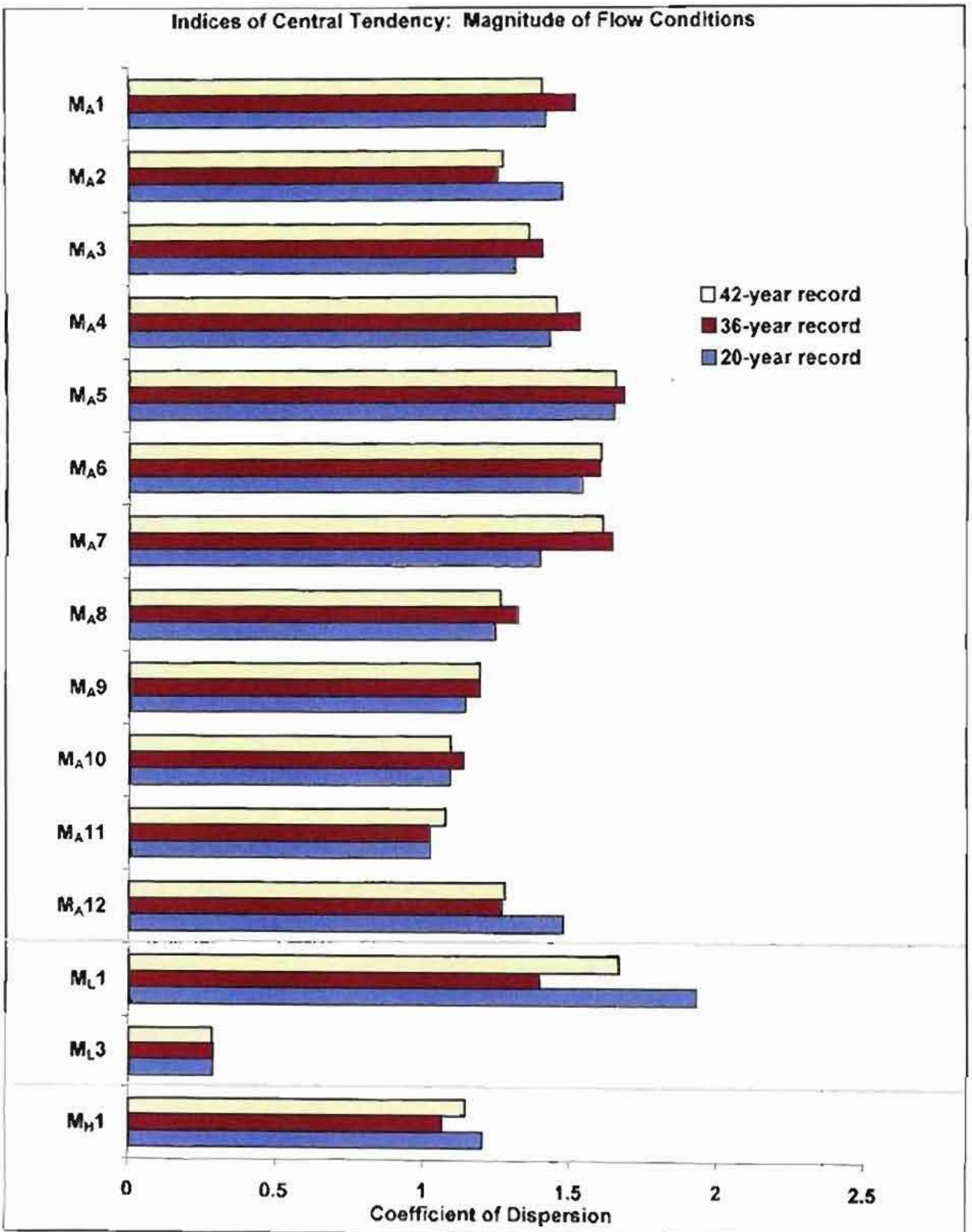

Figure 5.17 Coefficient of Dispersion of standardised annual values of the intra-annual indices describing the magnitude of flow conditions across (a) the 42-year record 1959 to 2000, (b) 36-year record 1965 to 2000 and (c) 20-year record 1981to 2000) for 43 DWAF gauging stations. See Table 5.11 for sample sizes.

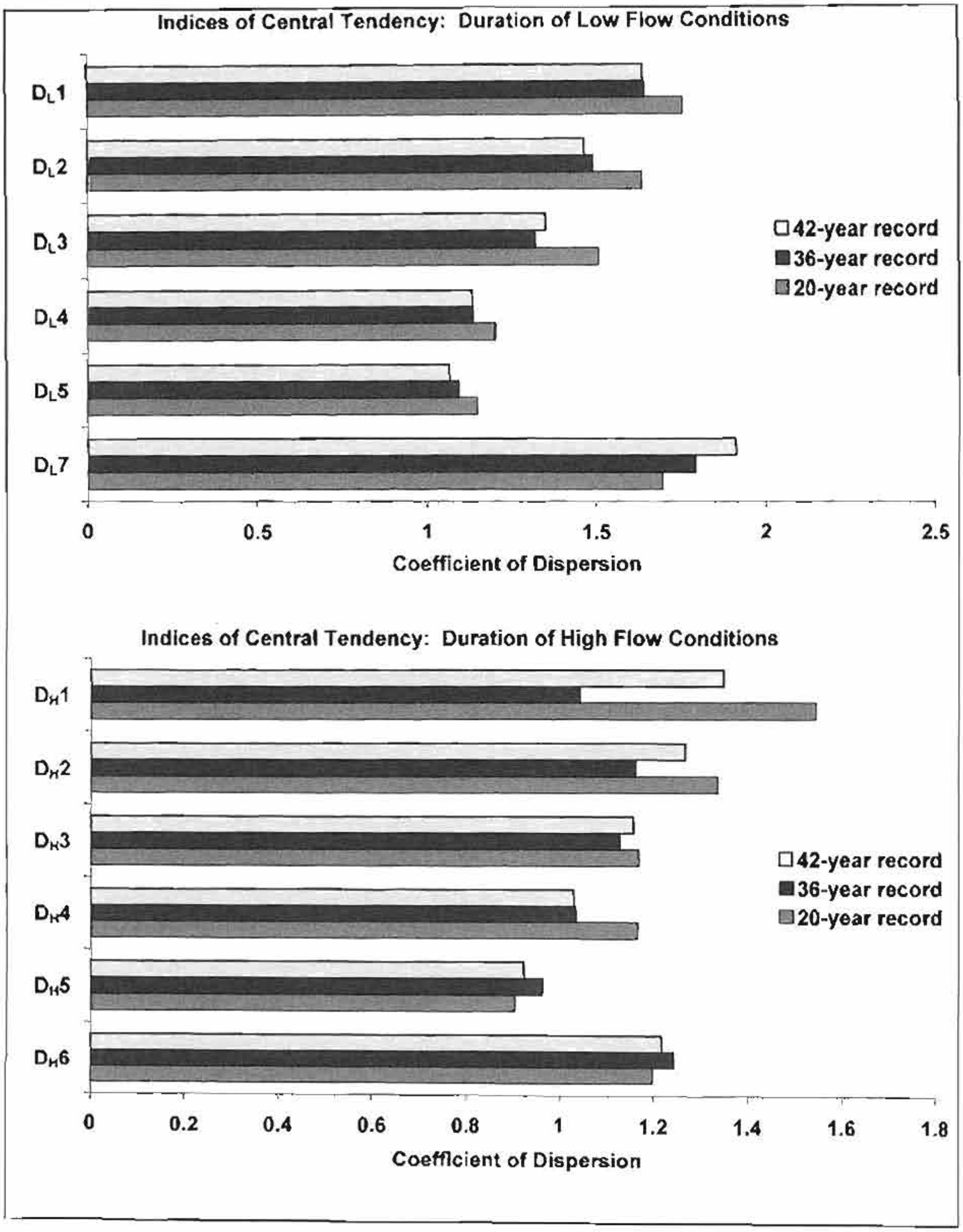

Figure 5.18 Coefficients of Dispersion of standardised annual values of the intra-annual indices describing the duration of flow conditions across the (a) 42-year record 1959 to 2000, (b) 36-year record 1965 to 2000 and (c) 20-year record 1981 to 2000 for 43 gauging stations. See Table 5.11 for sample sizes.

stable indices of the magnitude of average flow conditions, particularly for M_A9 (June), M_A10 (July) and M_A11 (August), and, in general 36 years of record would be sufficient to obtain reliable estimates of monthly flow conditions.

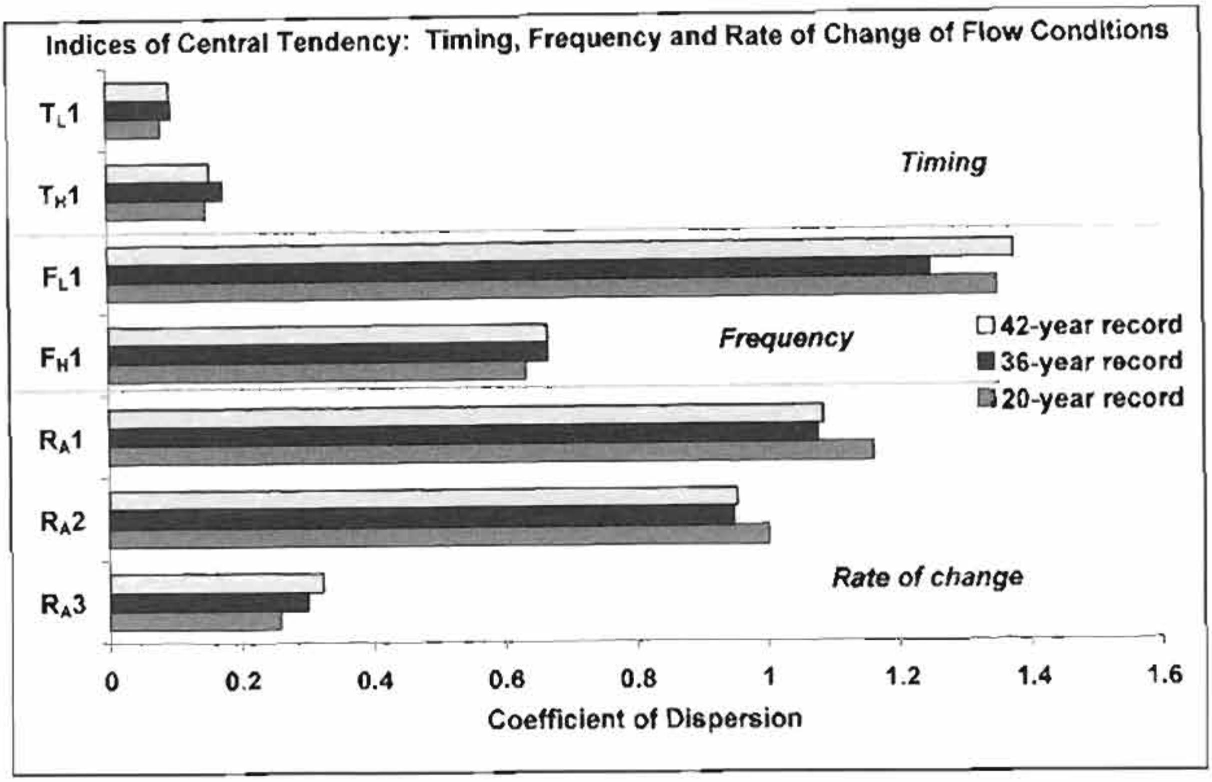

Figure 5.19 Coefficients of Dispersion of standardised annual values of the intra-annual indices describing the timing, frequency and rate of change of flow conditions across the (a) 42-year record 1959 to 2000, (b) 36-year record 1965 to 2000 and (c) 20-year record 1981 to 2000 for 43 DWAF gauging stations. See Table 5.11 for sample sizes.

At this broad spatial scale, greater differences exist among the record lengths required for indices of both the low and high flow conditions. M_L1 (the IHA baseflow index) and M_H1 (the high flow index) both require more than 36 years of record to stabilise. More than 20 years of record are also required for reliable indices of the minimum and multi-day extreme low flow conditions (D_L1 to D_L5), whereas the length of record required for the assessment of D_L7 (the duration of low pulses) is less clear and may require more than 42 years to stabilise. While 36 years of record is sufficient for the assessment of D_L1 to D_L4, D_L5 (the 90-day minimum flow) generally requires longer records. Analysis of the difference among the record lengths was performed for D_L6 (number of days with zero flow), but is not included in Figure 5.18 since the high proportion of zero values (of

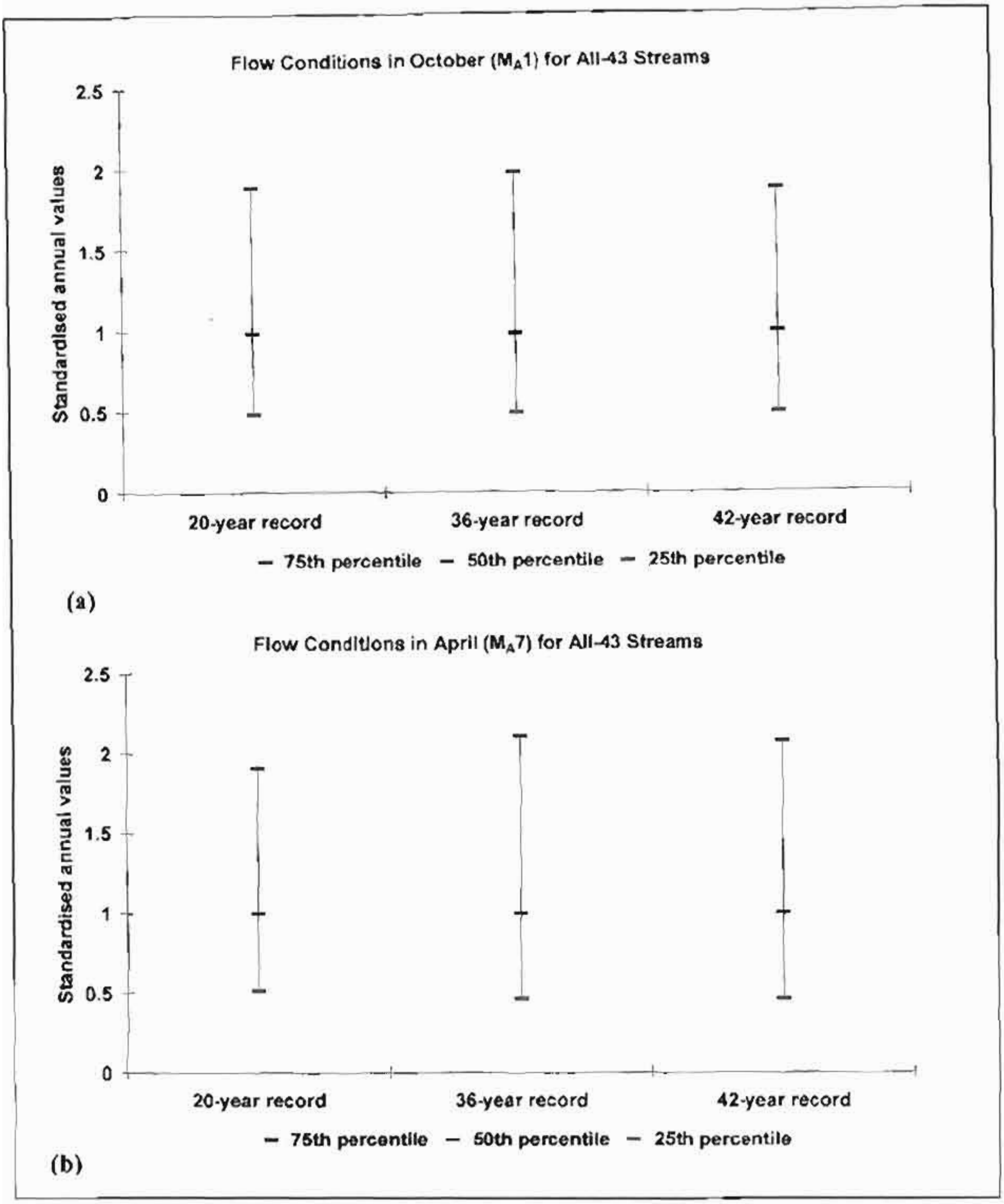

Figure 5.20 Variation across three different record lengths, backdated from 2000, of standardised annual values of monthly flow conditions in (a) October (M_A1) and (b) April (M_A7) for the reduced set of 43 DWAF Gauging stations (All-43 Streams). Median values for the record length are shown, together with the 25th and 75th percentiles, to indicate the extent of inter-annual variation. See Table 5.11 for sample sizes.

standardised annual values), at this spatial resolution of "All-43 Streams", hindered any useful discussion the behaviour of this index.

More than 20 years of record are also necessary to stabilise the minimum and shorter multi-day high flow conditions (D_H1 to D_H4). However, Figure 5.18 indicates that while 36 years of record may be sufficient for estimates of D_H3 and D_H4, longer records may be required for the shorter maxima high flow conditions D_H1 and D_H2. Thus, the length of record required for reliable indices of the high flow conditions reduces with increases with reductions in the time step analysed and 20 years may be sufficient for estimates of D_H5 (the 90-day maximum flow) and for (D_H6) the duration of high flow pulses.

At this broad spatial scale, records in excess of 20 years are required for assessments of R_A1 and R_A2 (the rates of rising and falling river levels), yet the length of record required for R_A3 (the number of reversals) is unclear, requiring at least 36 years. Twenty years of record is sufficient for assessments of T_L1 (Julian date of minimum flow) and T_H1 (Julian date of the maximum flow) as well as F_H1 (number of high pulses), although more than 36 years of record are preferable for assessments of F_L1 (number of low pulses).

The analysis to identify the length of record necessary to obtain stable indices of central tendency for each of the distinct streamflow types identified by this Study (Section 4.3.2 of this Chapter) resulted in large amounts of information. Consequently, the results reported here are restricted to those high information indices of central tendency identified for each of the different streamflow types (*c.f.* Section 4.5.3 of this Chapter). It is important to note that the analysis was conducted on the high information indices of central tendency identified by the PCAs performed for the indices derived from the *36-year record*. As in Clausen and Biggs (2000), it is acknowledged that had the PCAs been performed for indices derived from a different record length, different indices *could* have been identified as explaining the dominant patterns of hydrological variation. The Coefficients of Dispersion of standardised annual values of the relevant intra-annual indices are shown in Figures 5.21 to 5.23 and 5.25 to 5.28, for each of the streamflow types. The inter-annual variability associated with the high information indices for the reduced set of 43 DWAF gauging sites,"All-43 Streams", (*c.f.* Section 4.4.4 of this Chapter) has been included (*c.f.* Figure 5.21). However, since these indices form a subset of those described in the

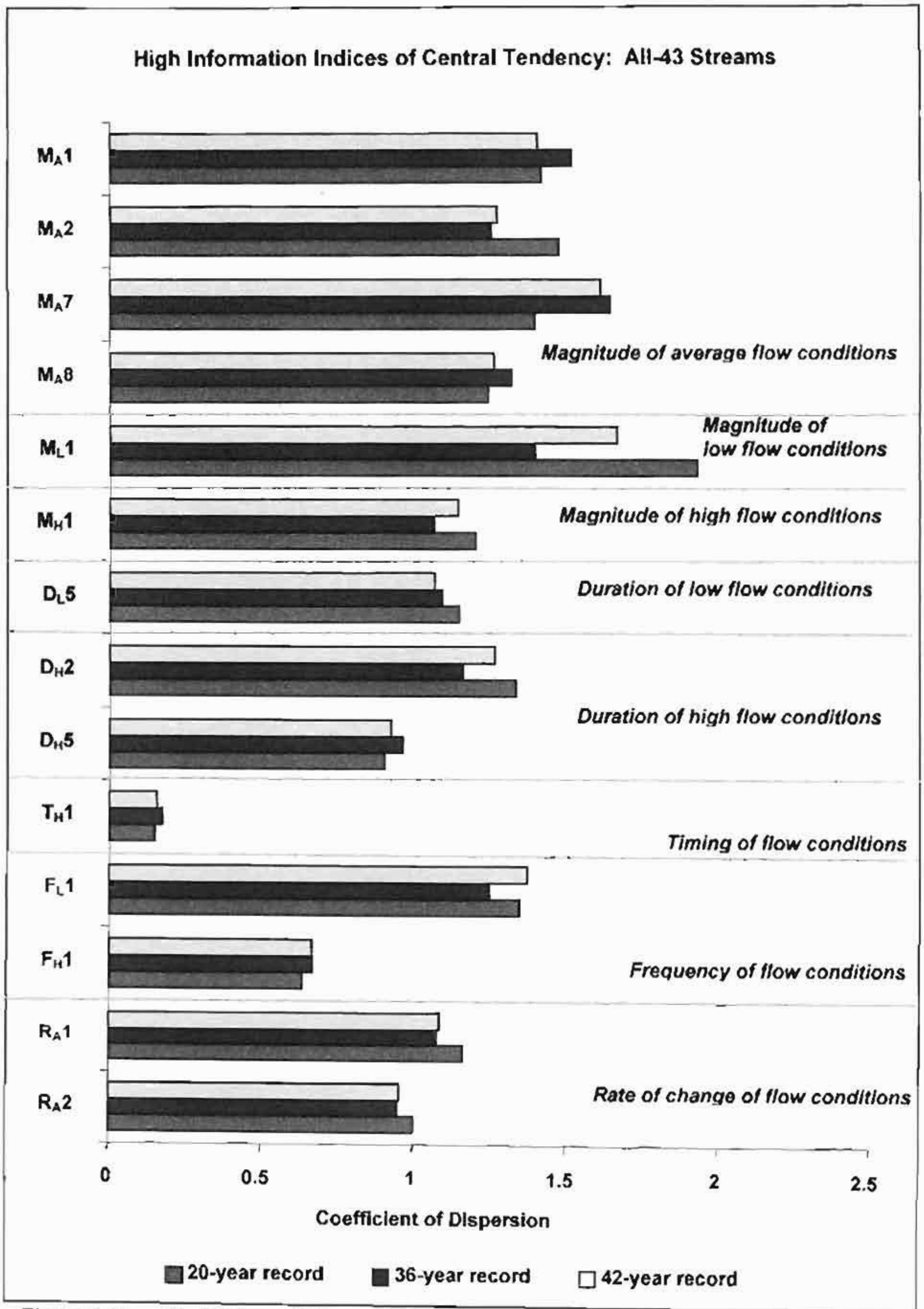

Figure 5.21 Coefficient of Dispersion of standardised annual values of the intra-annual indices representing high information indices for the reduced set of 43 DWAF gauging stations (All-43 Streams), across different record lengths. See Table 5.11 for sample sizes.

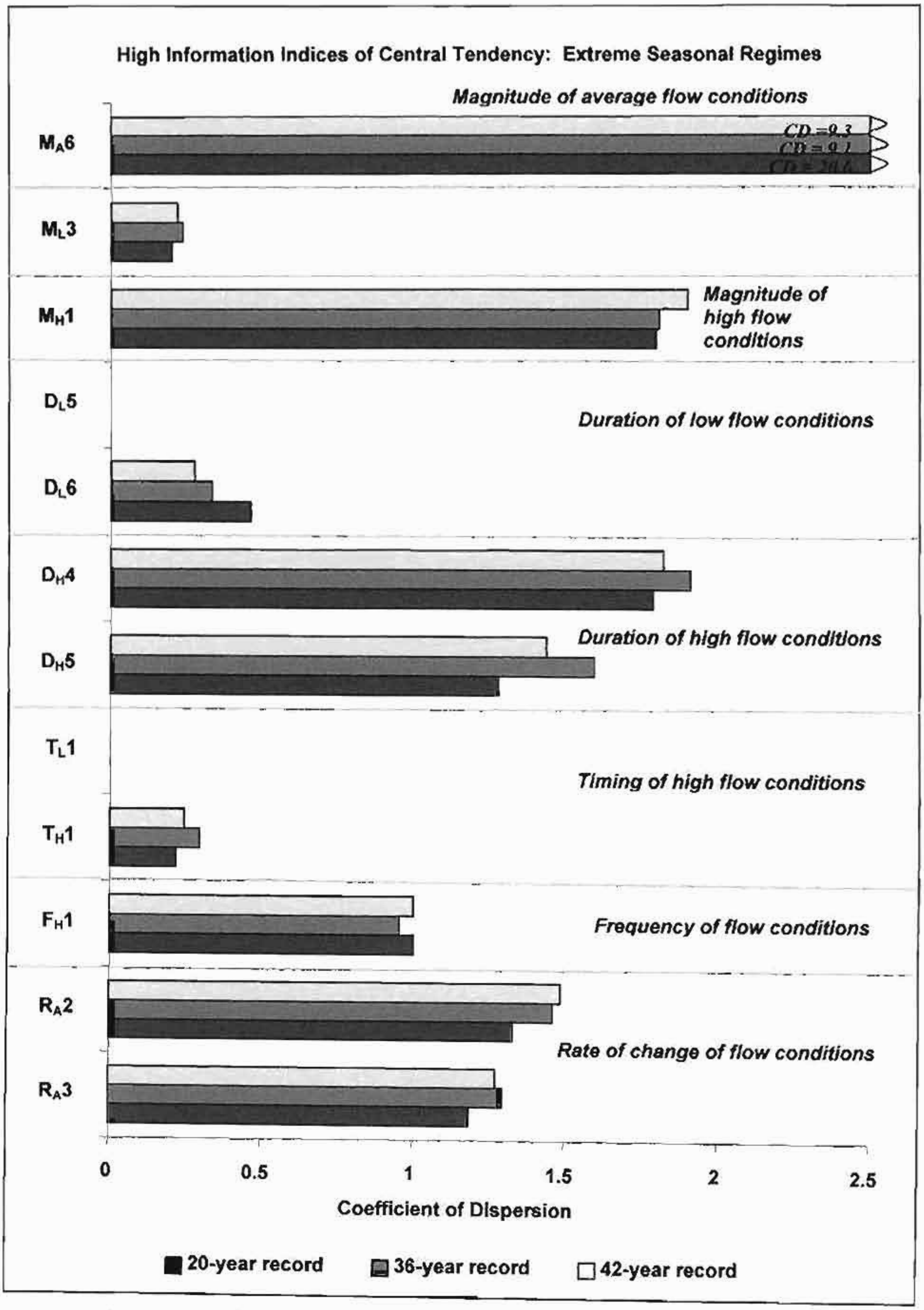

Figure 5.22 Coefficient of Dispersion of standardised annual values of the intra-annual indices representing high information indices for Extreme Seasonal regimes found in South Africa (across different record lengths). See Table 5.11 for sample sizes.

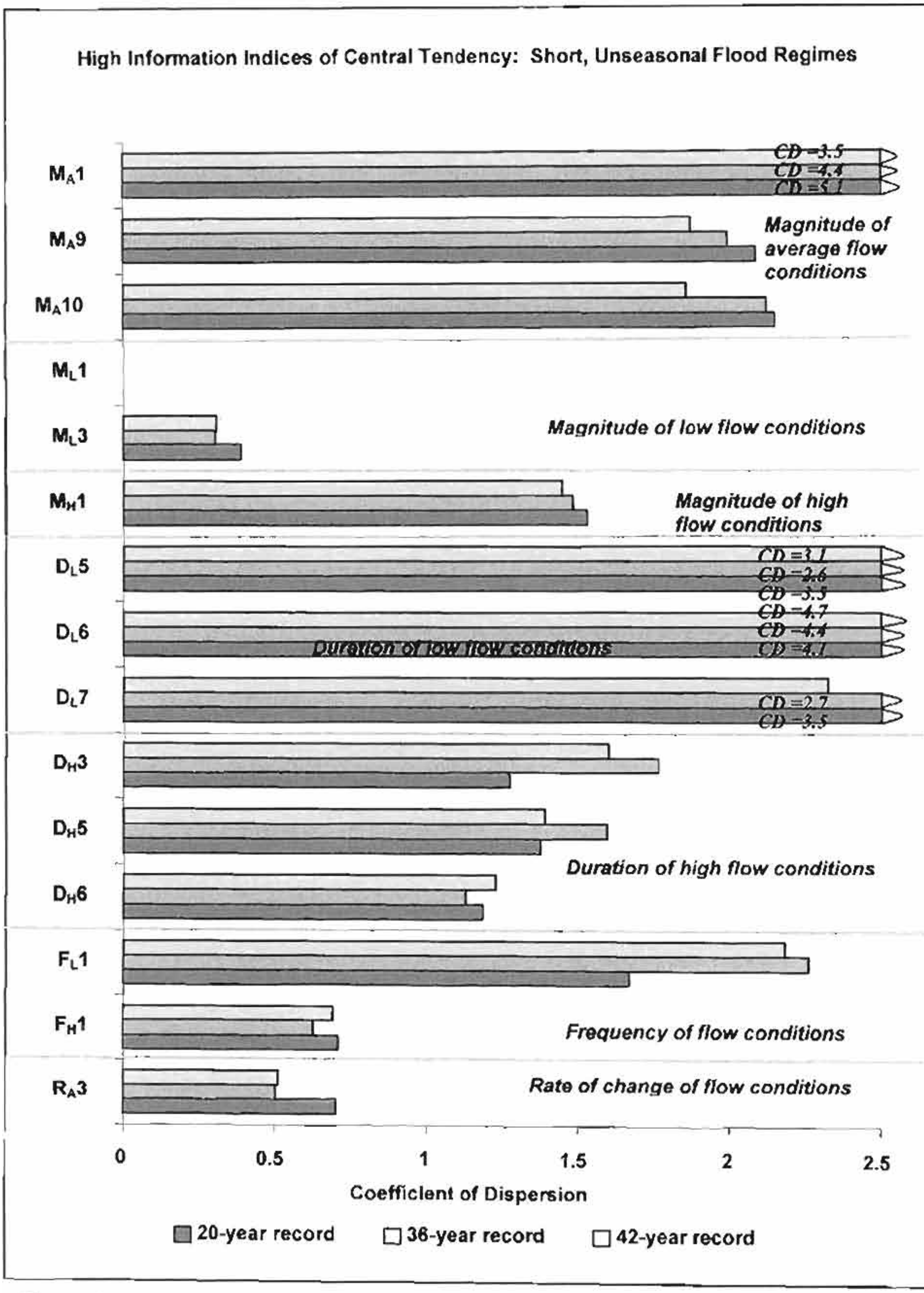

Figure 5.23 Coefficient of Dispersion of standardised annual values of the intra-annual indices representing high information indices for Short, Unseasonal Flood regimes found in South Africa (across different record lengths). See Table 5.11 for sample sizes.

immediately preceding paragraphs, they need no further discussion in this section; they are included solely for completeness.

Extreme Seasonal regimes

Figure 5.22 shows that most of the high information indices of central tendency identified for Extreme Seasonal regimes require more than 36 years of record to stabilise. Exceptions to this are M_A6 (average flows in March) and M_L3 (the Alt-BFI), both of which can be assessed reliably from 36 years of record, and probably even shorter records for M_L3. Section 4.5.3 of this Chapter indicated that M_A1 (average flows in October) is a high information index for Extreme Seasonal regimes. However, the values of the CD of standardised annual values for each of the record lengths investigated for M_A1 were far in excess of the values for any of the other indices and consequently not useful to this part of the study save to highlight the problems that decision makers face in selecting appropriate indices for the magnitude of even "average" flow conditions for extreme seasonal, or arid, river systems; much longer records are required for ephemeral than other streamflow types. Both D_L5 (the 90-day minimum) and T_L1 (the Julian date of the annual minimum) are shown on Figure 5.22 as having CD values of zero. These indices are so consistent for the harsh regimes of these streams that they do not require as many as 20 years for their reliable assessment. However, the reasons for their consistency differ. D_L5 is consistent as a result of the high proportion of standardised values of zero, across all three record lengths, whereas T_L1 often falls on the same Julian day (1 October) as a result of the computation of this index in the IHA software.

Short, Unseasonal Flood regimes

Figure 5.23 indicates that while longer records are likely to produce more reliable assessments of the average monthly flow conditions of Short, Unseasonal Flood regimes, there is little difference among the record lengths for both M_A9 and M_A10 (average flow in June and July), although more than 36 years of record would be preferable. Certainly, more than 36 years are required for assessments of M_A1 (average flow in October). The stability of the indices of the magnitude of low flow or of high flow conditions are also improved with longer records, yet 36 years of record is adequate for M_L3 (the Alt-BFI baseflow index) and probably M_H1 (the HF1, high flow index). Nonetheless, the high proportion of standardised annual values of zero for M_L1 (the IHA baseflow index) for each of the different records indicates that this index is consistent, regardless of the length of record.

At least 36 years of record are required for the stability of indices of D_H3 and D_H5 (the 7-day and the 90-day maximum flows), F_H1 and D_H6 (the number of high pulses and their durations). More than 36 years are also required for D_L7 (the duration of low pulses) and probably also for D_L5 (the 90-day minimum flow) and F_L1 (number of low pulses). The length of record required for the stability of D_L6 (the number of days with zero flow) is unclear. The most likely reason for this is that most of the streams in this group were formed from a streamflow type recording quasi-perennial-seasonal flow (*c.f.* Section 4.3.1 of this Chapter). Nonetheless, the 42-year record, which contains more low flow events, is preferable for estimates of D_L6 (*c.f.* Figure 5.24). Thirty six years of record are sufficient for assessments of R_A3 (number of hydrograph reversals).

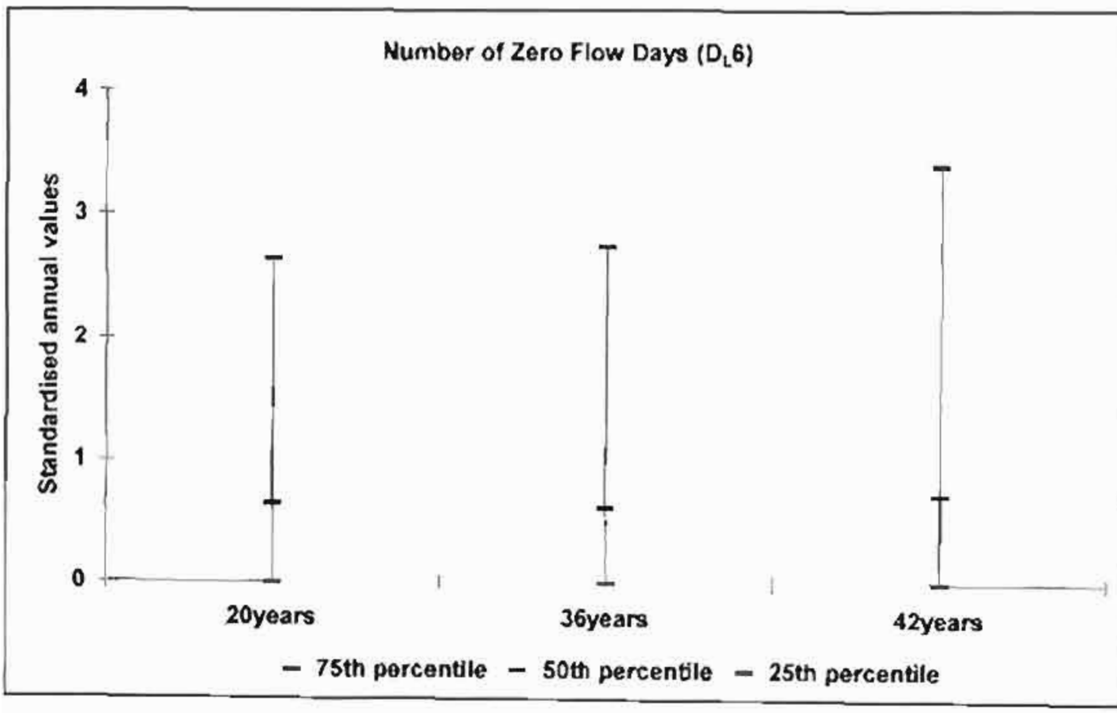

Figure 5.24 Variation across three different record lengths, backdated from 2000, of standardised annual values of the Numbers of Days with Zero Flow for the reduced set of Short, Unseasonal Flood regimes. Median values of standardised annual values across each record length are shown, together with the 25th and 75th percentiles, to indicate the extent of inter-annual variation. See Table 5.11 for sample sizes.

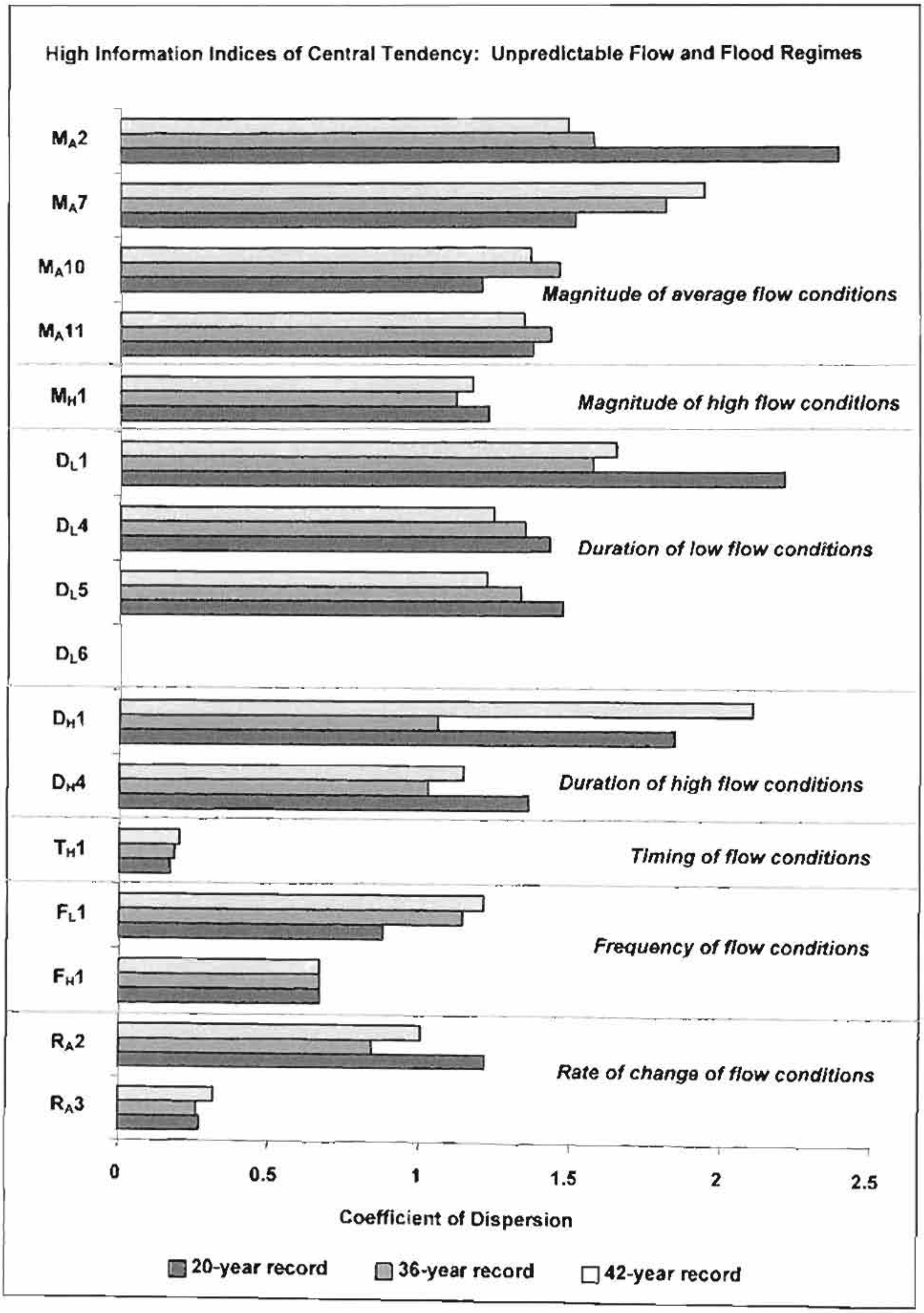

Figure 5.25 Coefficient of Dispersion of standardised annual values of the intra-annual indices representing high information indices for of Unpredictable Flow and Flood regimes found in South Africa (across different record lengths). See Table 5.11 for sample sizes.

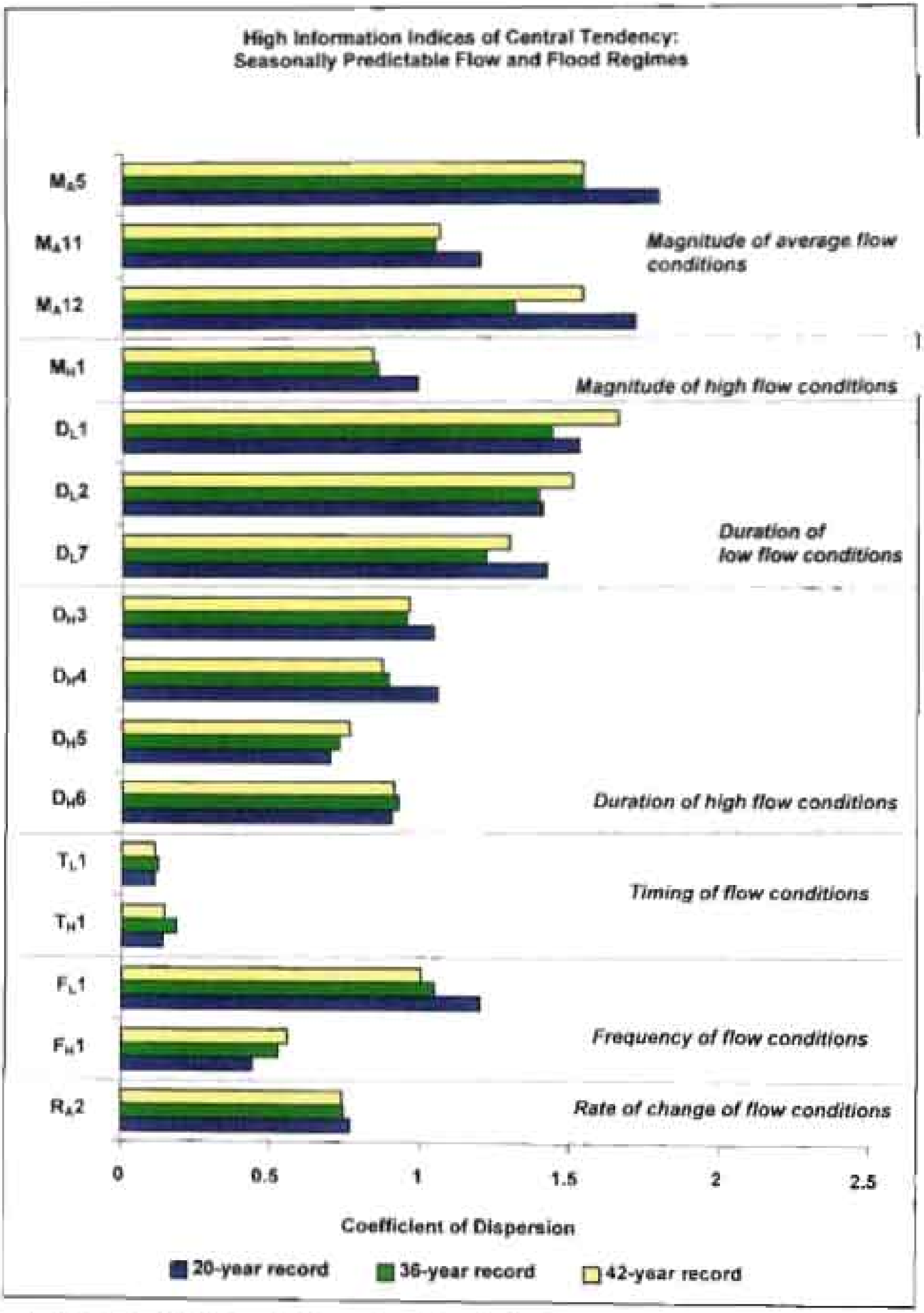

Figure 5.26 Coefficient of Dispersion of standardised annual values of the intra-annual indices representing high information indices for Seasonally Predictable Flow and Flood regimes found in South Africa (across different record lengths). See Table 5.11 for sample sizes.

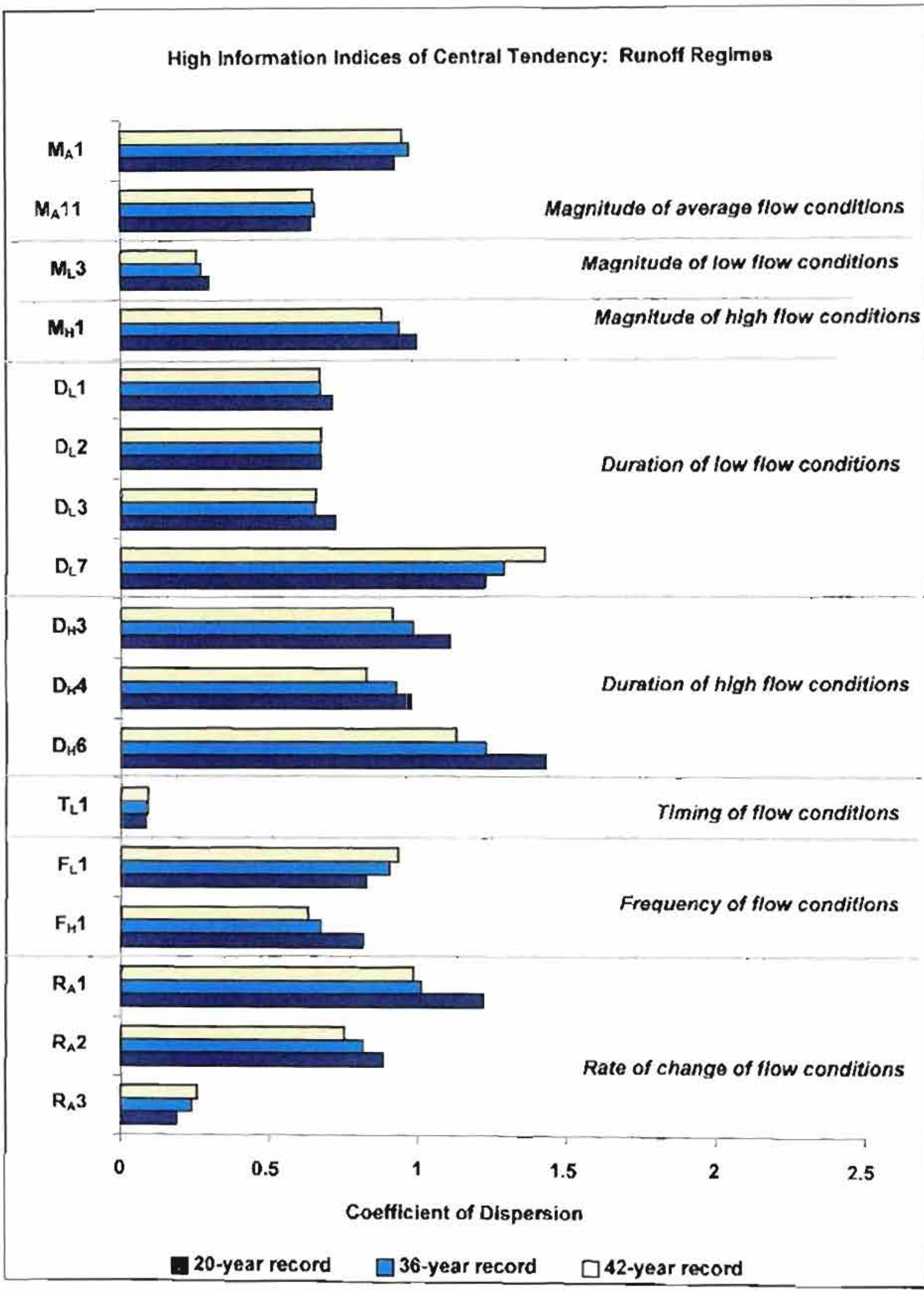

Figure 5.27 Coefficient of Dispersion of standardised annual values of the intra-annual indices representing high information indices for Runoff regimes found in South Africa (across different record lengths). See Table 5.11 for sample sizes.

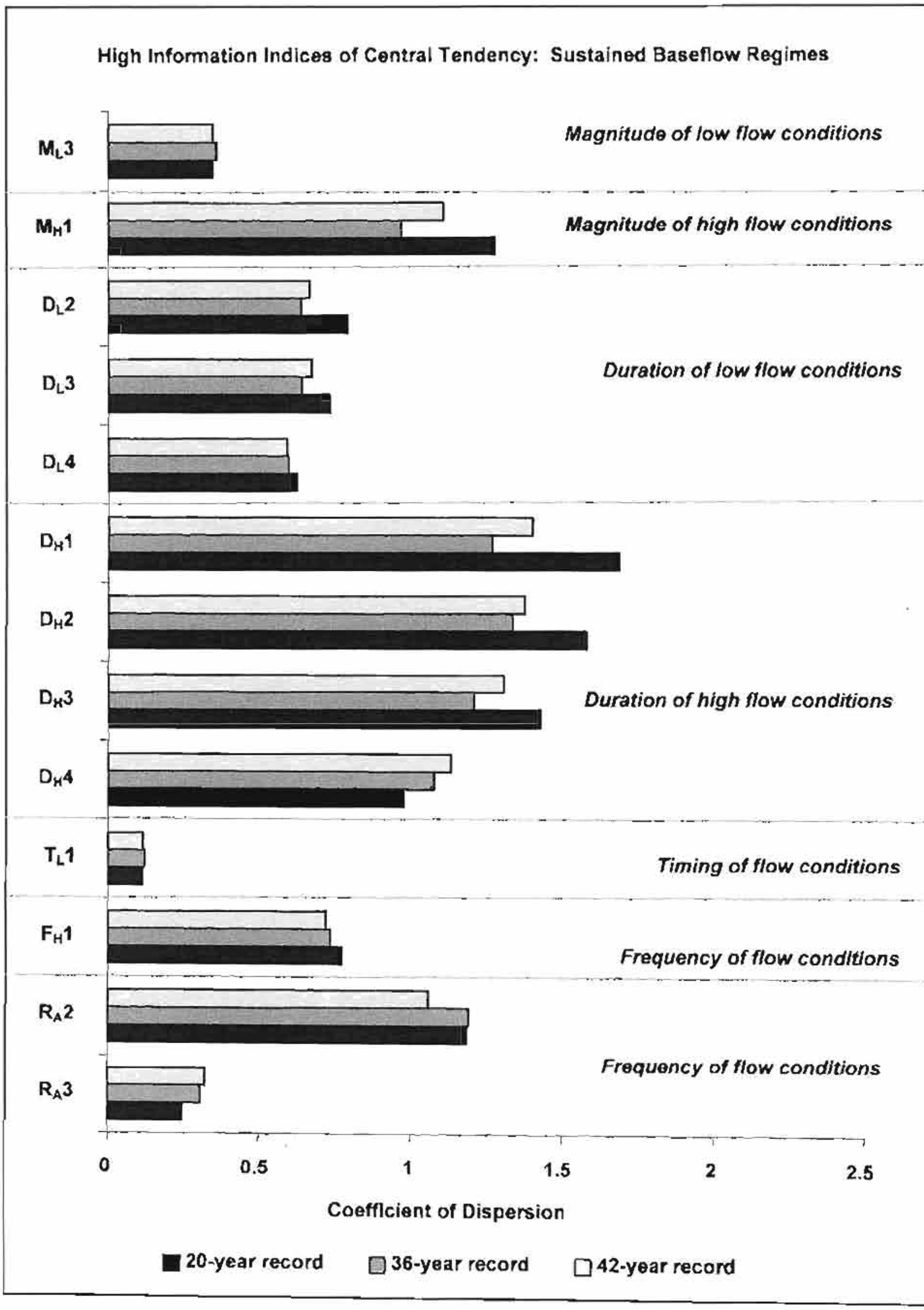

Figure 5.28 Coefficient of Dispersion of standardised annual values of the intra-annual indices representing high information indices for Sustained Baseflow regimes found in South Africa (across different record lengths). See Table 5.11 for sample sizes.

Unpredictable Flow and Flood regimes

Figure 5.25 shows that more than 36 years are required for consistent indices of M_A2, M_A10 and M_A11 (average flows in November, July and August), and probably more than 42 years for M_A7 (average flows in April). T_H1 (the Julian date of the maximum flow) F_H1 and R_A3 (numbers of both high pulses and hydrograph reversals) can all be assessed reliably from the 20-year record. More than 36 years of record are required for the stability D_L1, D_L4 and D_L5 (1-day, 30-day and 90-day minimum flows), R_A2 (falling river level), M_H1 (the HFI high flow index), D_H1 and D_H4 (the annual maximum and 30-day maximum flow). D_L6 (number of days with zero flow) can be assessed from less than 20 years since the high proportion of standardised annual values of zero for D_L6 for each of the different records indicates that this index is consistent, regardless of the length of record. However, more than 42 years of record would be preferable for the assessment of F_L1 (number of low pulse counts).

Seasonally Predictable flow and flood regimes

Figure 5.26 indicates that many of the high information indices of central tendency identified for the Seasonally Predictable Flow and Flood regimes can be assessed reliably from 36 years of record. These are M_A5 and M_A11 (flows in February and August), D_H3 and D_H4 (7-day and 30-day maximum flows), M_H1 (the HFI, high flow index) and R_A2 (the rate of the falling hydrograph), whereas T_L1 and T_H1 (Julian dates of the annual minimum and maximum flow) and D_H6 (high pulse duration) can be assessed adequately from the 20-year record. However, more than 36 years are required for M_A12 (average flows in September), D_L1 and D_L2 (1-day and 3-day minimum flows), D_H4 (30-day maximum flow) and F_L1 (number of low pulses). In addition, D_H5 (90-day maximum flow) and F_H1 (number of high pulses) may need more than 42 years to stabilise and for their adequate assessment.

Runoff regimes

Figure 5.27 indicates that the 20-year record is just as reliable as longer records for assessments of M_A11 (average flows in August), T_L1 (Julian date of the annual minimum flow) and D_L2 (3-day minimum flow) of Runoff regimes. Thirty-six years of record are sufficient for the assessment of both D_L1 and D_L3 (1-day and 7-day minimum flows) and also for M_A1 (flows in October). There is little difference among the record lengths regarding values of the CD of M_L3 (Alt-BFI) and while 20 years of record would suffice for reliable estimates of this index, Figure 5.27 indicates that longer records would be preferable. Thirty-six years of record are sufficient for assessments of R_A1 (rise in river level), whereas the high flow components of these streamflow regimes (i.e. M_H1, D_H3, D_H4, D_H6 and F_H1) all require more than 36 years of records for reliable estimates. F_L1 (the number of low pulses) and D_L7 (their durations) as well as R_A3 (number of hydrograph reversals) may require even longer for reliable assessment.

Sustained Baseflow regimes

M_L3 (magnitude of low flow conditions represented by the Alt-BFI) and T_L1 (Julian date of the minimum flow) at the sites of streams in the Sustained Baseflow group can be assessed just as reliably from the 20-year record as from longer records (Figure 5.28). However, more than 20 years of record are required for reliable assessments of D_L2, D_L3 and D_L4 (3-day, 7-day and 30-day minimum flows), although for each of these indices 36 years of record are just as reliable as 42 years. Thirty six years of record are sufficient for reliable assessments of F_H1 (number of high pulses) and R_A3 (hydrograph reversals). However, records of more than 36 years are required for the stability of R_A2 (the fall rate) and are probably also advisable for D_H1, D_H2 and D_H3 (the 1-day, 3-day and 30-day maximum flows) as well as the magnitude of high flow conditions, M_H1 (represented here by HF1). The record length necessary for reliable assessment of the D_H4 (30-day maximum flow) for this streamflow type is unclear and may require more than 42 years.

Summary of results of the analysis of the record length required for reliable indices of central tendency

The results of the comparisons of record length required to stabilise the high information hydrological indices of central tendency identified for each of the streamflow types described in the immediately preceding sections are summarised in Table 5.18. It is important to note that since the focus of the discussion related to only the high information indices, some indices described in this Study are omitted from the Table. That is not to say that the stability of these indices over different records lengths is not important, but rather that they are less relevant to the main aim of this Study, *viz,* the identification of a subset of indices that explain the major sources of the variation in all the indices describing the flow regimes of different streamflow types. In addition, the number of standardised annual values applied in the sensitivity analyses of the indices of central tendency to record

Table 5.18 Record lengths required to stabilise the high information hydrological indices of central tendency for different streamflow regime types. Blanks represent less useful indices.

Hydrological Index	Streamflow Flow Type						
	Extreme Seasonal	Mixed			Perennial		All Streams
	Extreme Seasonal	Short, Unseasonal Floods	Unpredictable Flow and Floods	Seasonally Predictable Flow and Floods	Runoff	Sustained Baseflow	All-43 Streams
M_A1							
M_A2							
M_A5							
M_A6							
M_A7							
M_A8							
M_A9							
M_A10							
M_A11							
M_A12							
M_L1							
M_L3							
M_H1							
D_L1							
D_L2							
D_L3							
D_L4							
D_L5							
D_L6							
D_L7							
D_H1							
D_H2							
D_H3							
D_H4							
D_H5							
D_H6							
T_A1							
T_H1							
F_L1							
F_H1							
R_A1							
R_A2							
R_A3							
Key	<20 years sufficient	20 years sufficient	36 years is sufficient	>36 years preferable	>42 years preferable		

length, as well as the number of high information indices, varied among the different streamflow groups.

Despite these anomalies of sample size, there are still some relevant general observations regarding the high information indices of central tendency that can be drawn from the results shown in Table 5.18.

(a) Dominant indices of the *magnitude of average flow conditions* require at least 36 years of record, and in most instances longer records, to stabilise. This is particularly relevant for the "mixed" streamflow types (the Groups comprising some streams with quasi-perennial-seasonal and others with perennial flow).

(b) While 20 years of record is more than adequate for the assessment of indices of the *magnitude of low flow conditions* for streams at the moderate end of the environmental gradient, 36 years of record is preferable for streams at the arid end of the range.

(c) With the exception of streams in the Seasonally Predictable Flow and Flood group, more than 36 years are preferable for assessments of indices of the *magnitude of high flow conditions*.

(d) With the exception of streams in the Runoff and Sustained Baseflow groups, more than 36 years of record are preferable for assessment of indices of the *duration of low flow conditions*.

(e) Records in excess of 36 years are preferable for assessment of the *duration of high flow conditions* for all streamflow regimes, except those in the Seasonally Predictable Flow and Flood group.

(f) For the most part, 20 years of record are more than adequate, across the diversity of streamflow types, for the stability of indices of the *timing of flow conditions*.

(g) The length of record required for indices of the *frequency of either high or low flow events* varies among the different streamflow groups. In general, shorter records are required for the assessment of the high flow events than are required for low flow events.

(h) The length of record required to stabilise the indices of *rates of change of flow conditions* also varies across the different streamflow types. In general, 36 years of record are adequate for streamflow regimes in the "mixed" groups. However, longer records are preferable for streams at either end of the environmental gradient.

This summary indicates that, depending on the streamflow type, *more than 36 years of record is preferential* for reliable assessment of many of the indices examined. While this information may seem inconsistent with the main research component of this Study *i.e.* the application of PCA of hydrological indices derived from streamflow records spanning only 36 years of record, it is important to re-emphasis the following points.

(a) This Study set out to investigate the longest common time period among the maximum number of DWAF gauging stations which were recording reasonably natural streamflows (*c.f.* Section 3.2.1). This resulted in only 83 DWAF stations, recording streamflows for the 36-year period from 1965 to 2000, being included in a "working database". Clearly a longer record length would have been desirable. However, a compromise had to be reached between the number of DWAF gauging stations which could be utilised and the length of common record.

(b) The investigation of the record length required to ascertain reliable indices was initiated to examine the suggestion by the developers of the IHA suite of ecologically relevant hydrological indices that a minimum of 20 years of data is required to minimise the effects of inter-annual climatic variation on the IHA parameter statistics (Richter *et al.*, 1997). Thus, it was considered acceptable to proceed with a working database of 74 hydrological indices derived from streamflow records spanning 36 years of record at the 83 DWAF stations.

(c) Inadequacies regarding relatively long streamflow records of good or better quality, *and* which also represent reasonably natural flows, are a common problem in environmental flow assessments for South African rivers. The data used in this Study represent the best available observed streamflow records for the purposes of the PCA. However, it is important to emphasise that had the PCA been performed for indices derived from a different record length, different indices *could* have been identified as explaining the dominant patterns of hydrological variability (Clausen and Biggs, 2000).

(d) The emergence of information that *some* of the high information indices identified by the PCA may be more reliably assessed from records longer than 36 years, for *some* of the different streamflow types, does not detract from the usefulness of the Study. Rather, the results provided in Table 5.18 can be applied with varying degrees of confidence, depending on the relevance of any particular index to the eco-hydrological study in question, and on the region where the study is performed.

4.5.4.2 Indices of streamflow variability and streamflow predictability

Overall behaviour of the hydrological indices

Where possible in this Section, the indices of (a) dispersion of each of the 33 IHA intra-annual indices and (b) overall variability (*c.f.* Figure 5.7) are referred to collectively as "*indices of streamflow variability*", whereas the indices of predictability and of seasonal predictability (*c.f.* Figure 5.7) are referred to collectively as "*indices of streamflow predictability*".

The Coefficients of Dispersion of the hydrological indices of streamflow variability and of streamflow predictability in different streamflow characteristics across the entire set of 83 streamflow sites (All-83 Streams), as well across each of the different streamflow types, are shown in Figures 5.29 to 5.33 (*c.f.* Table 5.13 for sample sizes applied) for the magnitude, duration, timing, frequency and rates of change of streamflow conditions. The proportion of zero values in each instance is also shown.

The CD values for "All-83 Streams" indicate that using measures of streamflow variability and of streamflow predictability to describe the broad spatial patterns of hydrological variability shows greater disparity among the sites for indices of the magnitude of flow conditions (Figure 5.29) and the duration of low flow conditions (Figure 5.31) than for indices of the duration of high flow conditions (Figure 5.32) and the timing, frequency and rate of change of flow conditions (Figure 5.33). The most notable exception to this feature is that there is considerably greater disparity among the sites for D_H13 (seasonal predictability of non-flooding) than for any other index. Indices T_A1 and T_A2 (relating to the predictability and constancy of the flow regime), T_H3 (the seasonal predictability of flooding) as well as R_A4 and R_A5 (variability in the rise and falls rates of river levels respectively show least disparity among the sites.

Some of the indices of streamflow variability and of streamflow predictability had high proportions of zero values across the diversity of the different streamflow regimes found in South Africa. These are D_L13 (variability in the number of zero flow days, 0.74) and, to a lesser extent, D_H13 (seasonal predictability of non-flooding, 0.30) and M_L2 (variability in the IHA baseflow index, 0.24), as indicated in Figures 5.31, 5.32 and 5.30) respectively.

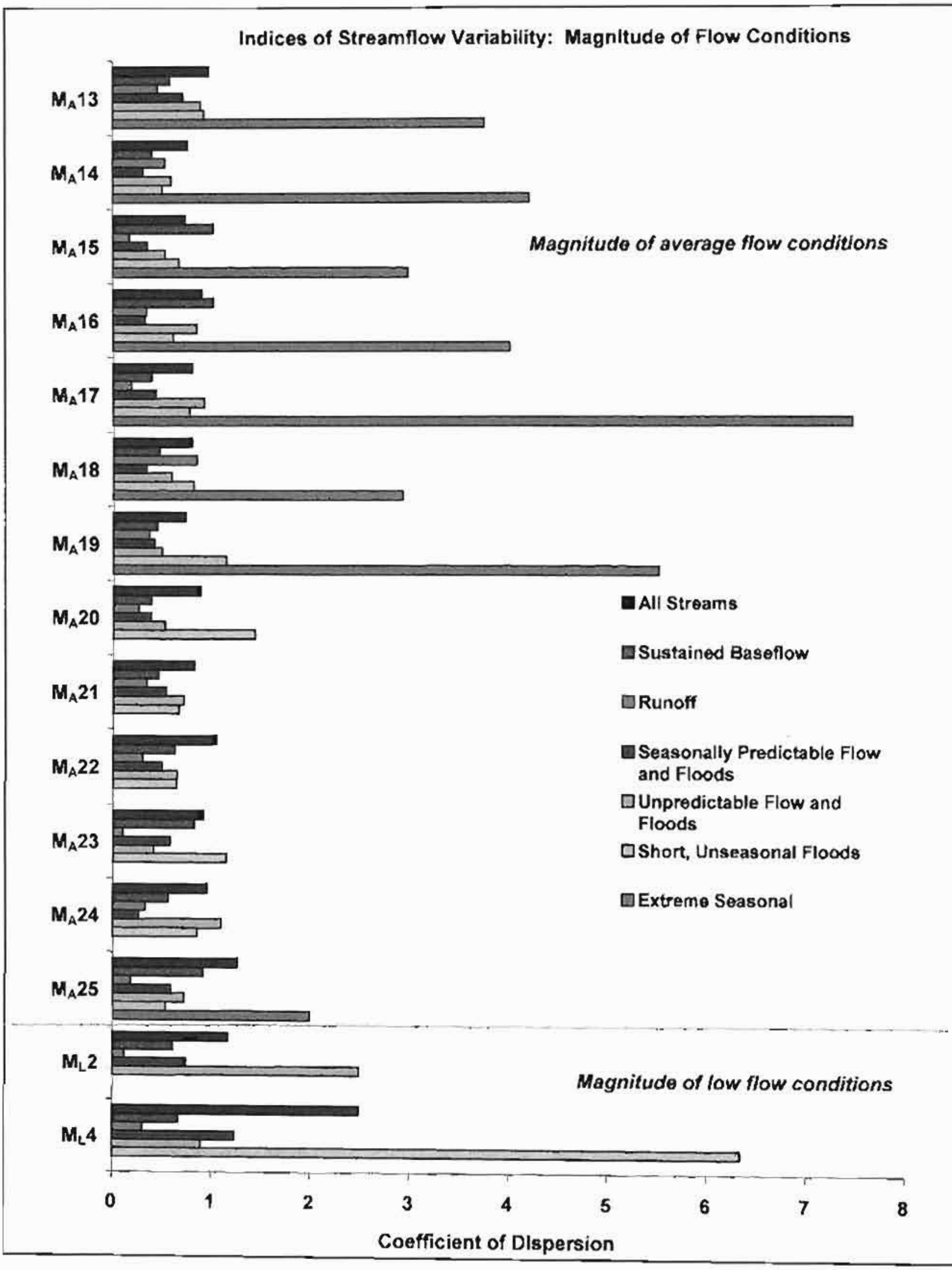

Figure 5.29 Coefficient of Dispersion of the indices of streamflow variability in the magnitude of streamflow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types. See Table 5.13 for sample sizes.

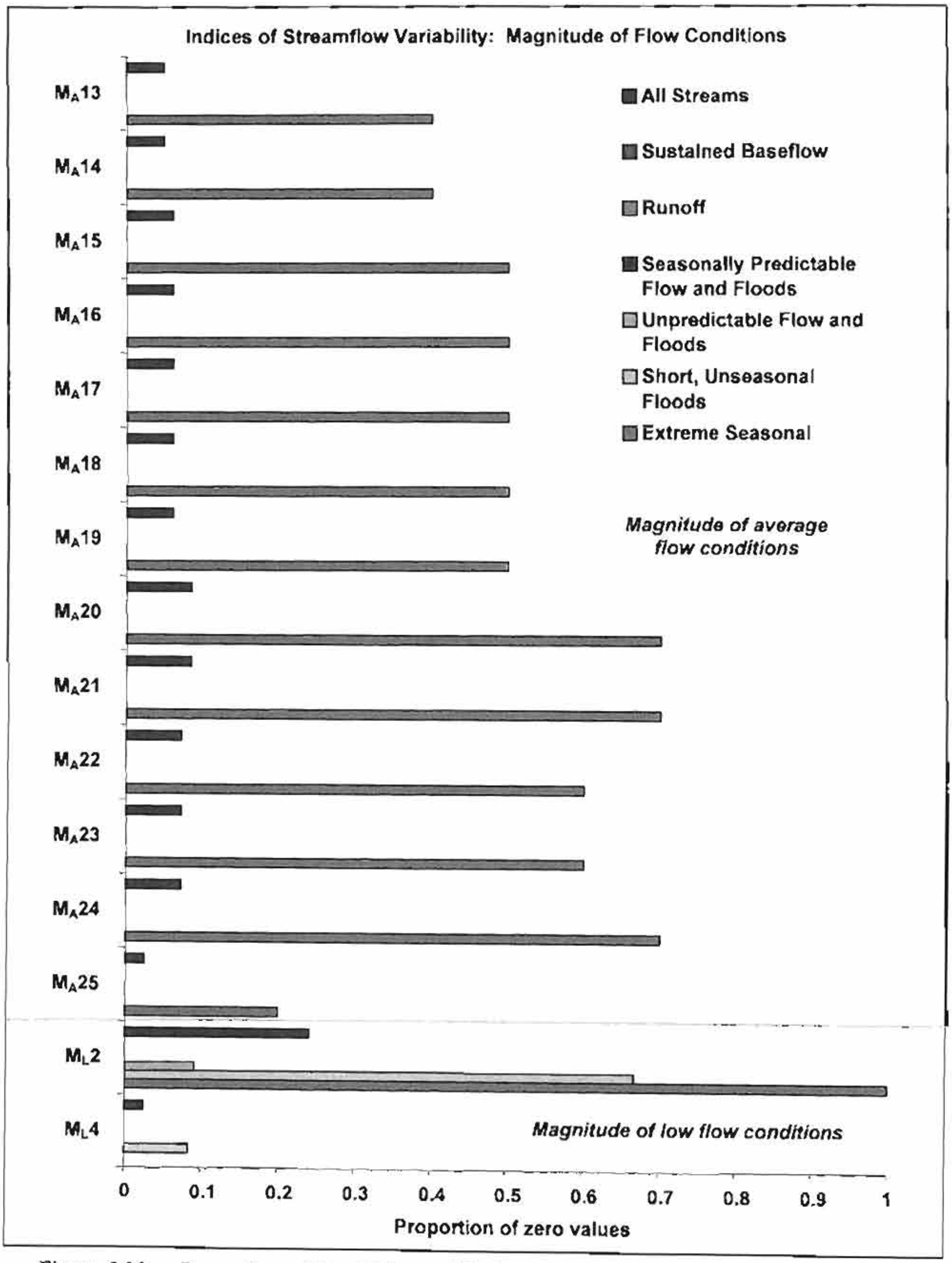

Figure 5.30 Proportion of Zero Values of the indices of streamflow variability in the magnitude of flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types. See Table 5.13 for sample sizes.

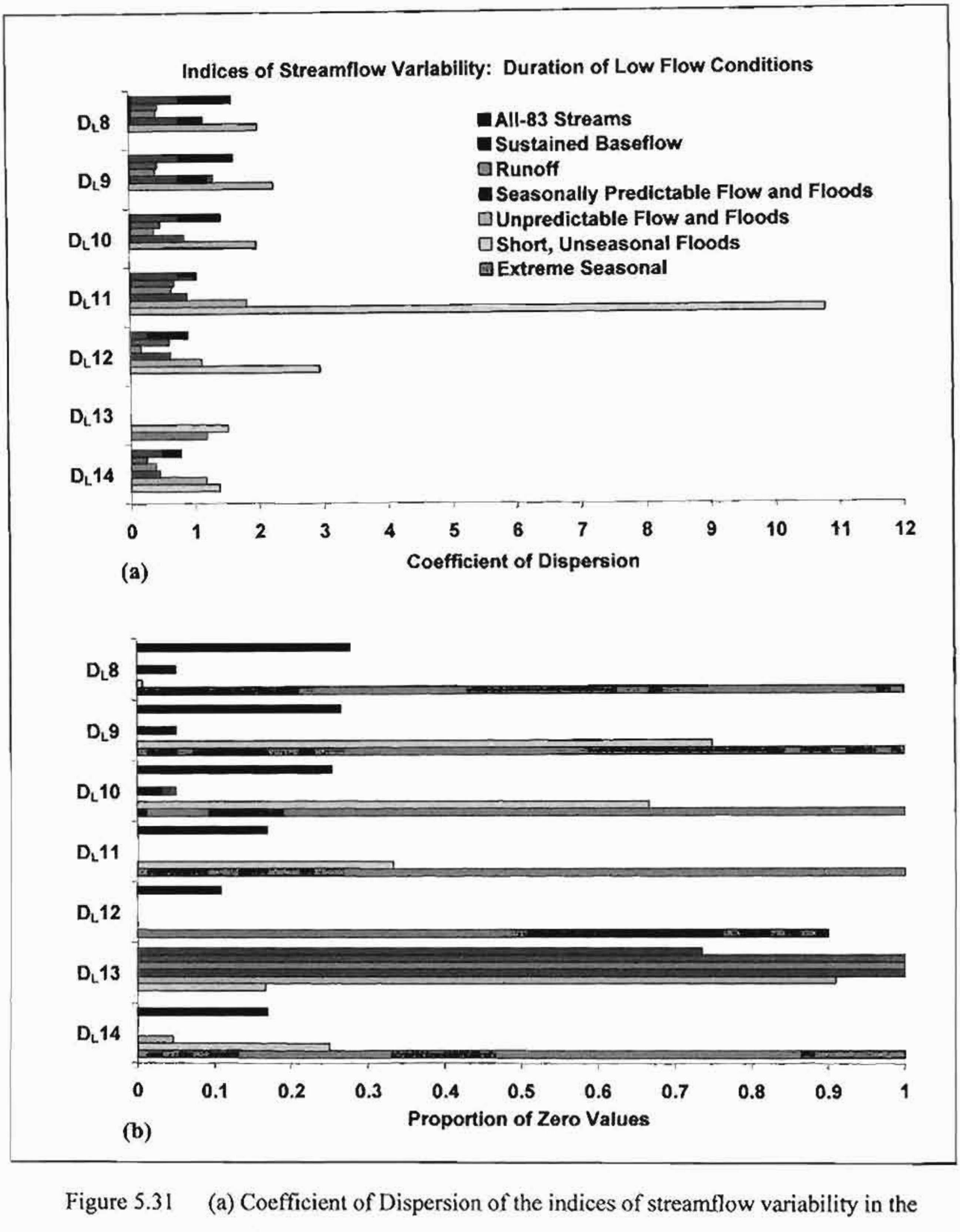

Figure 5.31 (a) Coefficient of Dispersion of the indices of streamflow variability in the duration of low flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types and (b) the Proportion of Zero Values. See Table 5.13 for sample sizes.

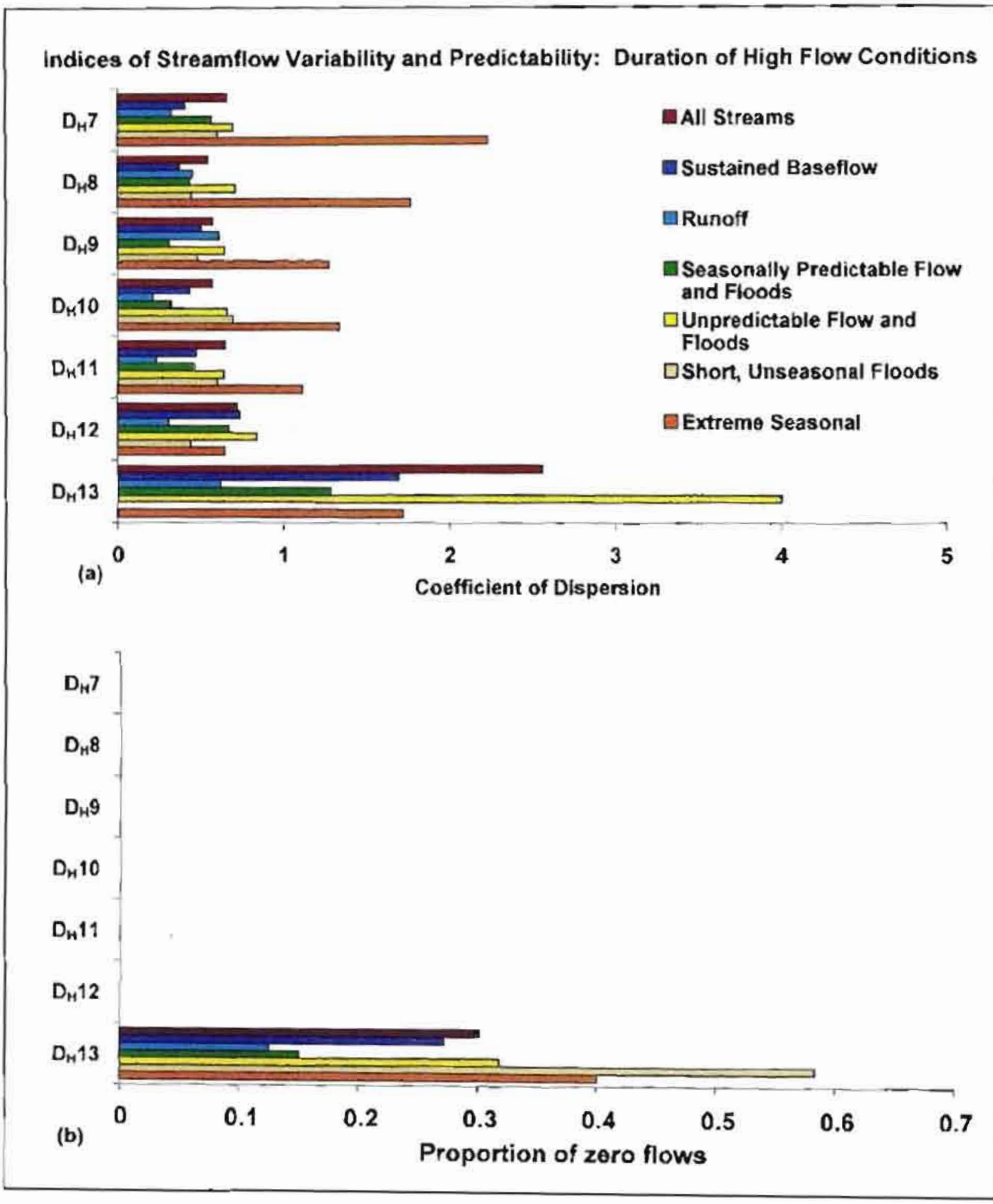

Figure 5.32 (a) Coefficient of Dispersion of the indices of streamflow variability and of streamflow predictability of the duration of high flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams), as well as the different streamflow types and (b) the Proportion of Zero Values. See Table 5.13 for sample sizes.

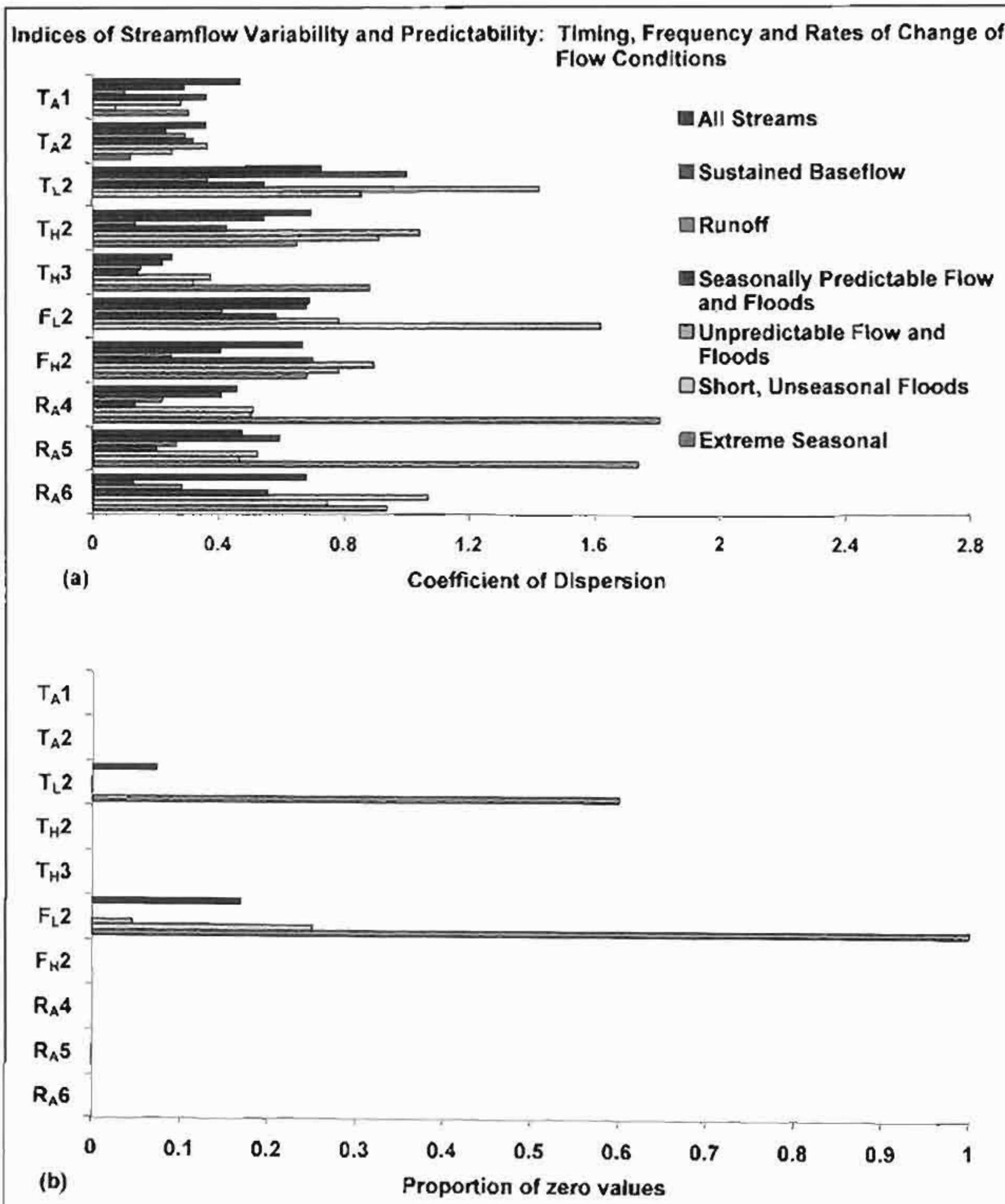

Figure 5.33 (a) Coefficient of Dispersion of the indices of streamflow variability and of streamflow predictability of the timing, frequency and rate of change of flow conditions across the 36-year record 1965 to 2000 for the entire set of 83 DWAF gauging stations (All-83 Streams) as well as the different streamflow types and (b) the Proportion of Zero Values. See Table 5.13 for sample sizes.

This finding confirms the results of the PCA and highlights their limited usefulness for describing streamflow patterns across this broad spatial scale.

Comparison of the indices of streamflow variability and of streamflow predictability among different streamflow types

Figures 5.29 to 5.33 show that the CDs of the indices of streamflow variability and of predictability are generally greater for the Extreme Seasonal streams than the other streamflow types and are generally lowest for the streams in the Runoff and Sustained Baseflow streamflow types. The only notable exception to this feature is that there is lower disparity of T_A2 (related to the constancy, or invariance, of the streamflow regime) among both the Short, Unseasonal Flood and the Extreme Seasonal groups than for the other streamflow types.

Differences in the CDs of each of the indices M_A13 to M_A24 across the diversity of streamflow types show that streams in the Runoff group also have the lowest among site disparity in the monthly streamflows, whereas the streams in Extreme Seasonal group have the greatest (Figure 5.29). This measure of streamflow variability is indicative of the difficulties encountered when assessing the monthly flow patterns of streamflow regimes at the arid or ephemeral extremity of the spectrum of streamflow regimes found in South Africa. This supports the observation in Section 4.5.4.1 that longer records are necessary for the identification of consistent indices which describe the streamflow patterns associated with ephemeral and intermittent streams than are required for perennial streams.

The CD values for M_L2 (variability in the IHA Baseflow Index), M_A25 (the CDB), D_L12 (variability in the 90-day minimum flow) and D_H10 to D_H12 (variability in the longer multi-day maxima flows) are relatively low for the Runoff Group, indicating the reliability of these indices for assessments of both low and high flow conditions in Runoff regimes

Values of 1 (*i.e.* the highest possible value) for the Proportion of Zero Values were computed for the M_L2 (variability in the IHA Baseflow Index), D_L8 to D_L12 (variability in the minimum and multi-day minima) and D_L14, as well as F_L2 (variability in the number of low pulses and their durations), for the Extreme Seasonal Group and for D_L13 (variability in the number of days with zero flow) for the Seasonally Predictable Flow and Floods, Runoff and Sustained Baseflow Groups. There is minimal variability in these low flow conditions across the respective streamflow regimes. Consequently, these indices are not useful for describing the streamflow patterns within these respective regime types. Again, this feature can be confirmed by their low rankings on each statistically significant principal component axes in the PCA.

Comparisons of record lengths required for describing streamflow variability and of streamflow predictability among streamflow types

The Coefficients of Dispersion, calculated for the indices of streamflow variability and of streamflow predictability, for the reduced number of 43 DWAF gauging stations across different record lengths is shown on Figures 5.34 to 5.36. The Figures compare the *overall behaviour* of the hydrological indices of streamflow variability and of streamflow predictability over the different record lengths at a broad spatial scale and give an indication of the number of years of record required to stabilise each index.

Again the impact of at least one climatic event being excluded from the 20-year record is evident in Figures 5.34 to 5.36. Moreover, in the impacts of such events on high flow and low flow conditions in the streamflow regime is much more pronounced for indices of streamflow variability and of streamflow predictability than it is for the indices of central tendency (*c.f.* Section 4.5.4.1 of this Chapter). However, the following observations can be deduced for the overall behaviour of the indices of streamflow variability and of streamflow predictability.

There is least difference among the record lengths for M_A24 (the variability in flows in September), indicating that, for these monthly flows, the 20 year record is just as useful as longer records for environmental flow assessments of a diversity of streamflow regimes. This is contrary to the deduction in Section 4.5.4.1 of this Chapter where it was reported that at least 36 years of record would be required for reliable estimates of M_A12 (the *average* of flows in September).

There is also little difference among the record lengths when assessing the reliability of the M_A25 (the index of overall variability, CDB) for the diverse range of streams found in South Africa and 20 years of record may be sufficient for desktop assessments of unmonitored streams.

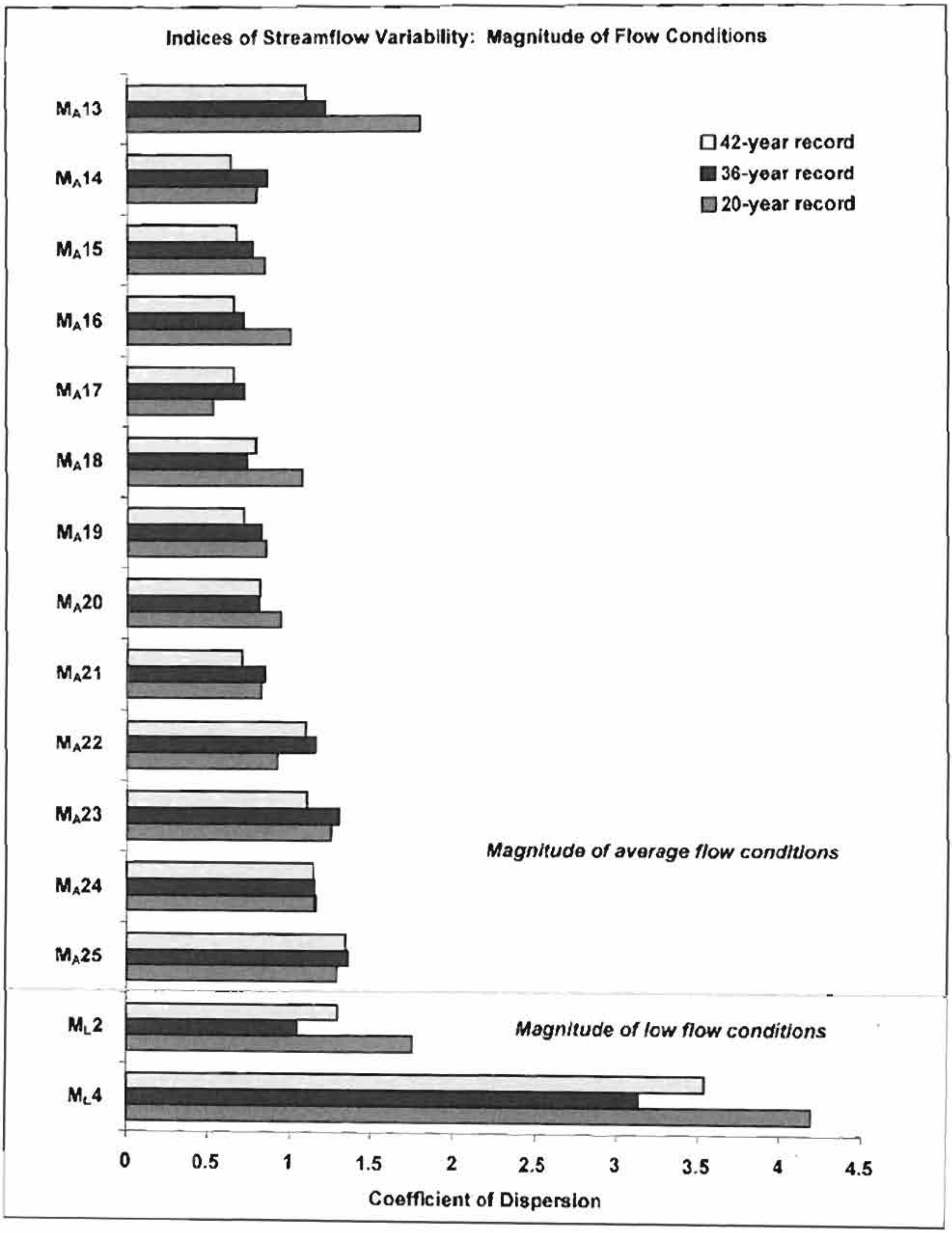

Figure 5.34 Coefficient of Dispersion of the indices of streamflow variability in the magnitude of flow conditions across (a) the 42-year record 1959 to 2000, (b) the 36-year record 1965 to 2000 and (c) the 20-year record 1981 to 2000, for 43 DWAF gauging stations. See Table 5.13 for samples sizes.

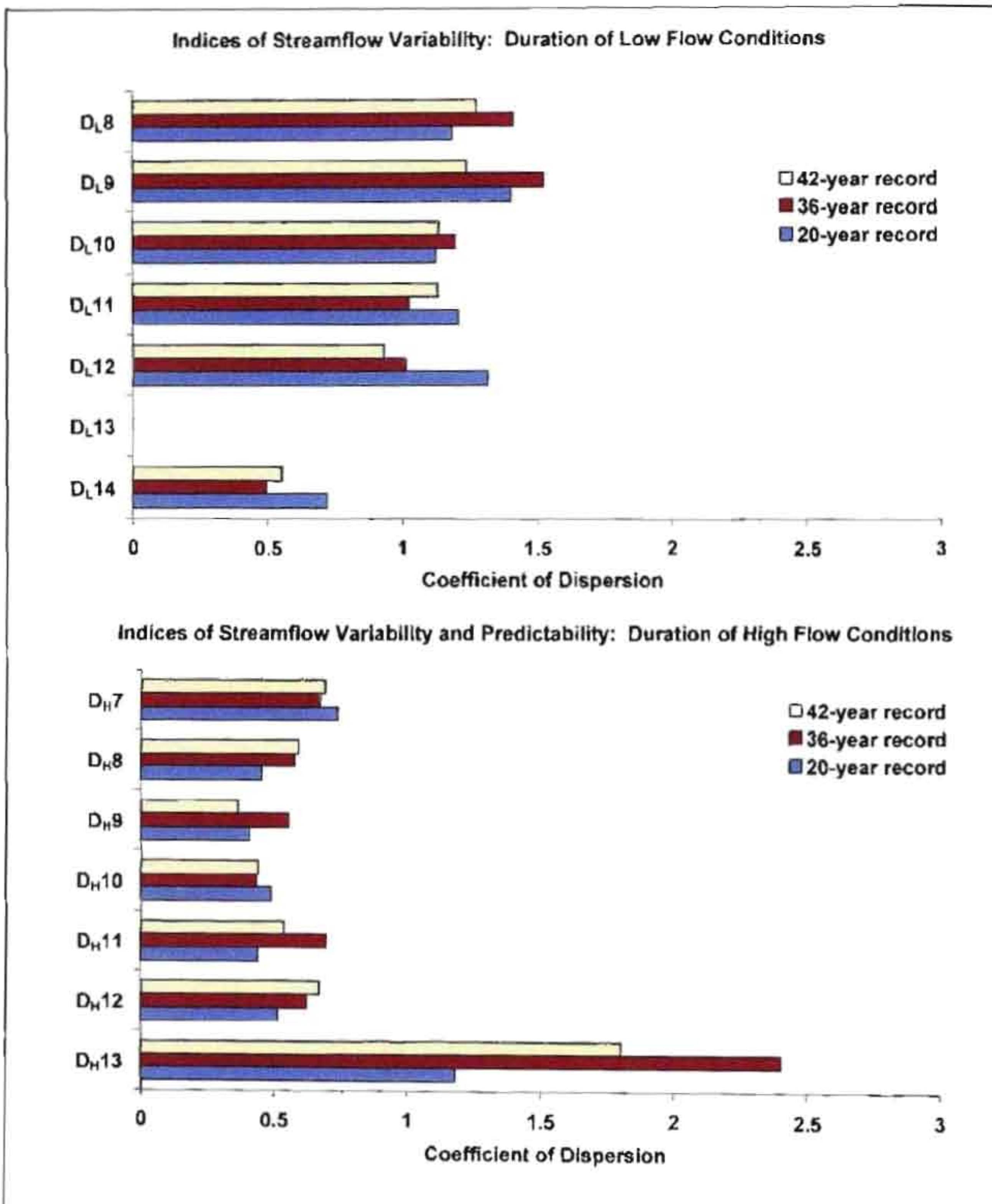

Figure 5.35 Coefficients of Dispersion of the indices of streamflow variability and of streamflow predictability of the duration of flow conditions across the (a) 42-year record 1959 to 2000, (b) 36-year record 1965 to 2000 and (c) 20-year record 1981 to 2000 for 43 gauging stations. See Table 5.13 for sample sizes.

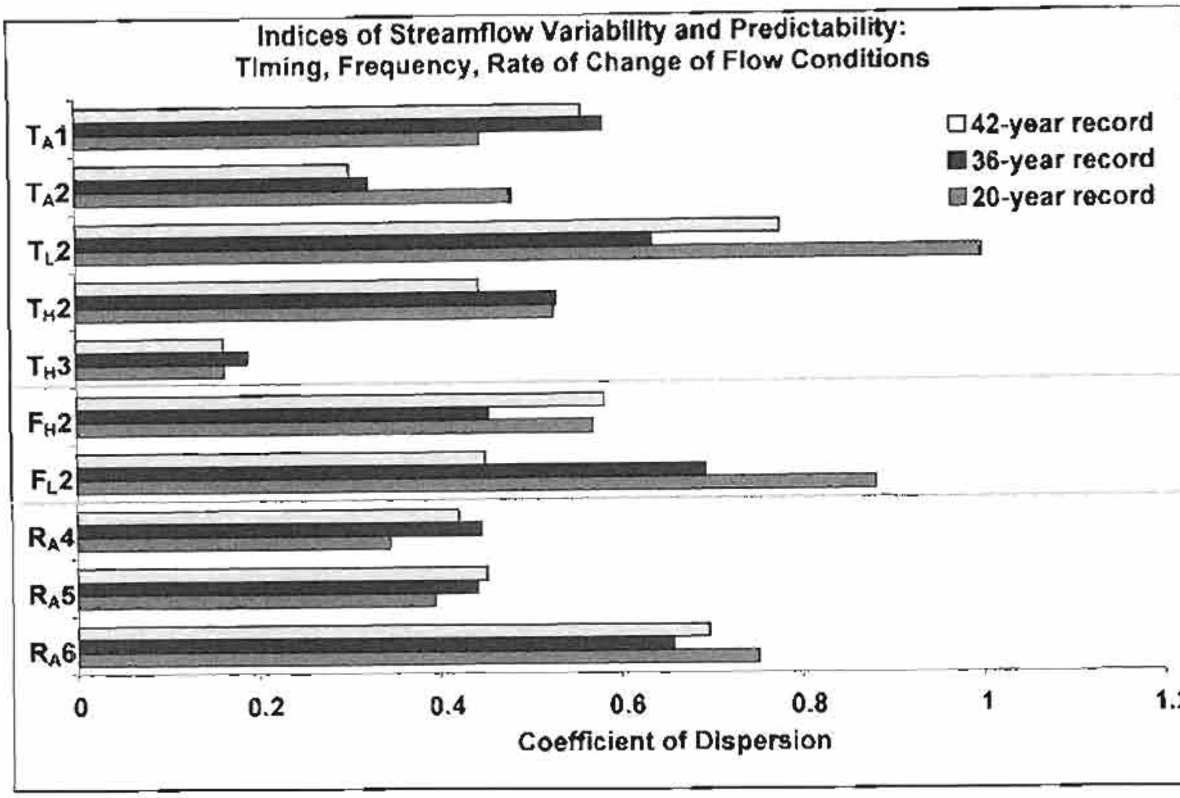

Figure 5.36 Coefficients of Dispersion of the indices of streamflow variability and of streamflow predictability of the timing, frequency and rate of change of flow conditions across the (a) 42-year record 1959 to 2000, (b) 36-year record 1965 to 2000 and (c) 20-year record 1981 to 2000 for 43 DWAF gauging stations. See Table 5.13 for sample sizes.

Figure 5.34 shows that, with the exception of M_A13 and to a lesser extent M_A16 and M_A18 (the variability in flows in October, January and March), there are only small differences in the overall behaviour of the variability in monthly flow conditions over different record lengths. In general, more than 36 years of record is preferable for reliable assessments of the variability within each of the calendar months at this broad spatial scale, although 36 years is sufficient for M_A20 and M_A22 (variability in flows in May and July). Greater differences exist among the indices of variability in the magnitude of low flow conditions in the flow regimes. Both the indices M_L2 (variability in the IHA Baseflow Index) and M_L4 (Q75) require more than 36 years for even coarse assessments, at this spatial resolution.

Figure 5.35 indicates that more than 36 years of record are required for assessments of the variability in the minimum and multi-day minima flows (D_L8 to D_L12) as well as the duration of low pulses (D_L14). There is no difference among the record lengths for D_L13 (the variability in the number of days with zero flow) across the reduced set of 43 Streams. In each instance the CD value of D_L13 is zero, indicating that, in general, this is a consistent index of the variability in this extreme low flow condition.

Figure 5.35 indicates that with the exception of D_H10, (the 30-day maximum), more than 36 years of record are required for estimates of variability in the 1-day maximum and the multi-day flows. This is similar to the findings of the record length requirements of the central tendency of the maximum and shorter multi-day high flow conditions in Section 4.5.4.1 of this Chapter. However, the variability in the duration of the high pulse, D_H12, requires more than 42 years for reliable assessments. Thirty-six years of record is sufficient for estimates of T_H3 (the seasonal predictability of flooding) across the diversity of sites, but not for D_H13 (seasonal predictability of non-flooding), which needs more than 36 years of record.

Twenty years of record are just as reliable as 36 years of record for estimates of T_H2 (variability in the date of maximum flow) for the different streamflow regimes, although more than 36 years are preferable (Figure 5.36). Records of 36 years are sufficient for assessments of T_A1 and T_A2 (indices of predictability and constancy of the streamflow regime) as well as for R_A4, R_A5 and R_A6 (variability in the rates of rising and falling river levels and the number of hydrograph reversals). However, records in excess of 36 years are required for assessments of T_L2 (variability in the date of the minimum flow), F_L2 and F_H2 (variability in the numbers of both low and high pulses).

The analysis to identify the length of record necessary to obtain reliable indices of streamflow variability and of streamflow predictability for each of the different streamflow types also resulted in large amounts of information pertaining to the 39 indices tested. The results reported here are restricted to those indices of high information identified for the different streamflow regime types (*c.f.* Section 4.4.3 of this Chapter).

The Coefficients of Dispersion of the high information indices of streamflow variability and of streamflow predictability for each of the streamflow regime types are shown in Figure 5.37 to 5.43. The variability exhibited by the high information indices of the reduced set of 43 DWAF gauging stations (All-43 Streams) has been included

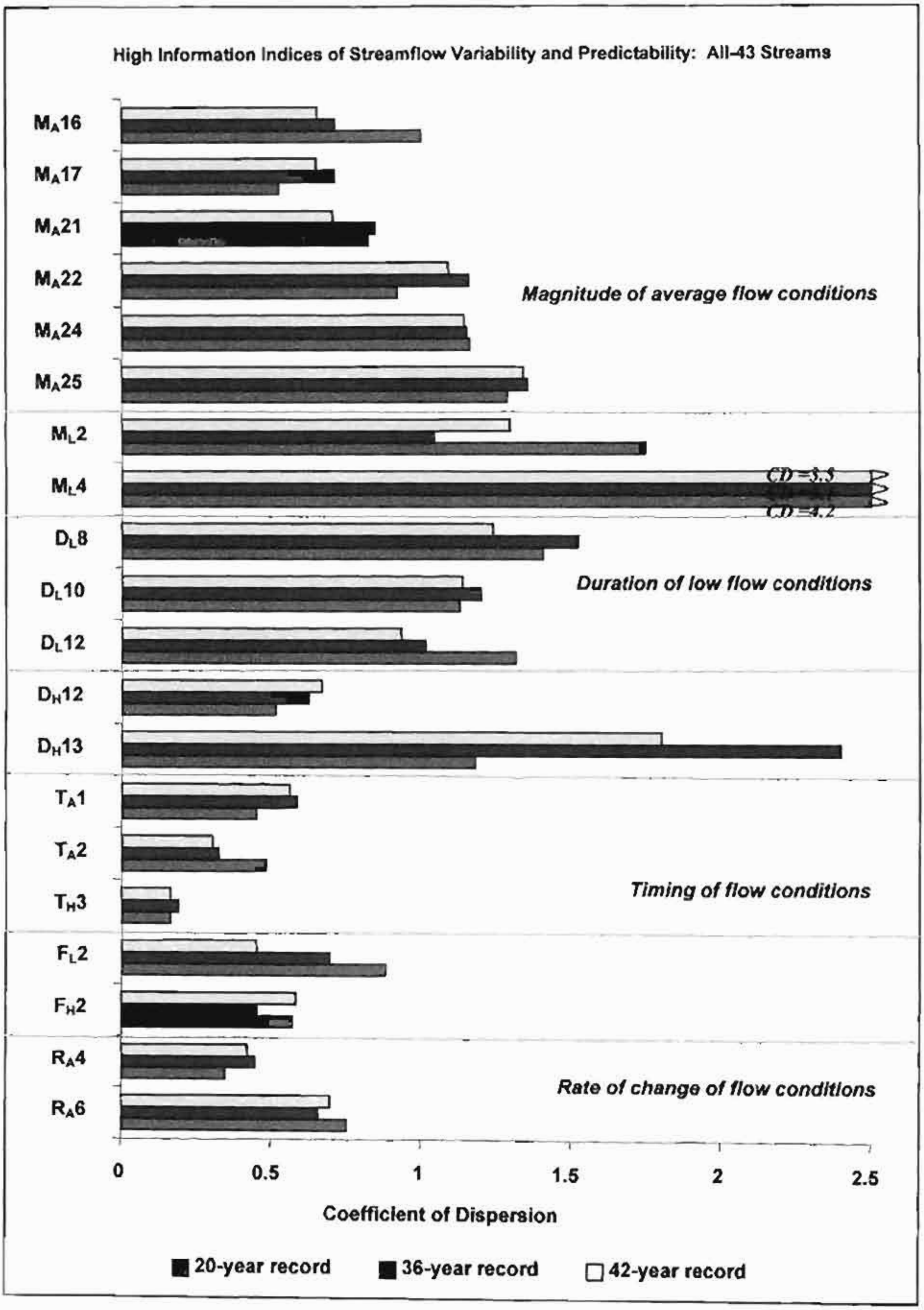

Figure 5.37 Coefficient of Dispersion of the high information indices of streamflow variability and of streamflow predictability of the flow conditions of the reduced set of 43 DWAF gauging stations, "All-43 Streams" (across different record lengths). See Table 5.13 for sample sizes.

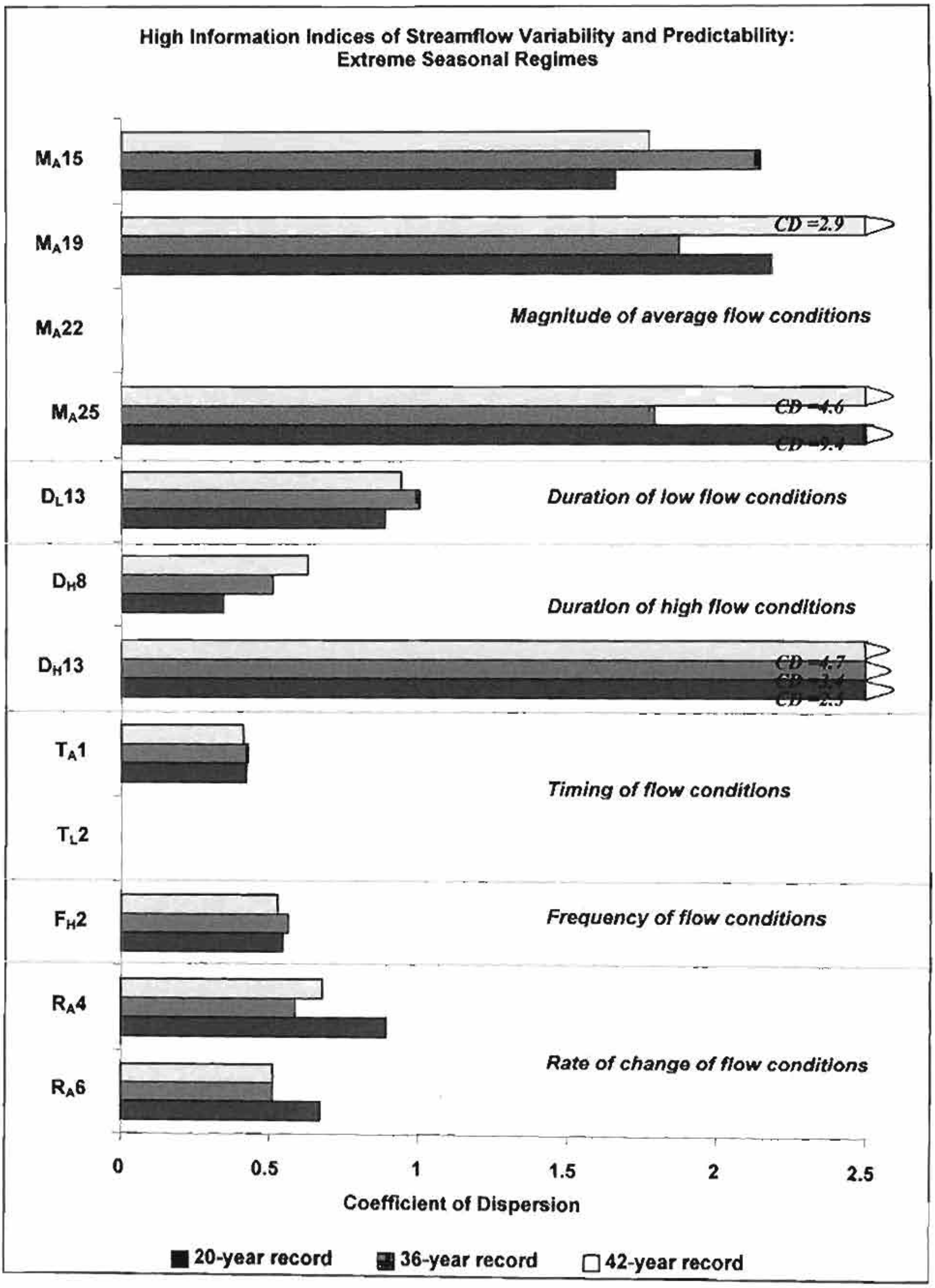

Figure 5.38 Coefficient of Dispersion of high information indices of streamflow variability and of streamflow predictability of the flow conditions of Extreme Seasonal Regimes found in South African (across different record lengths). See Table 5.13 for sample sizes.

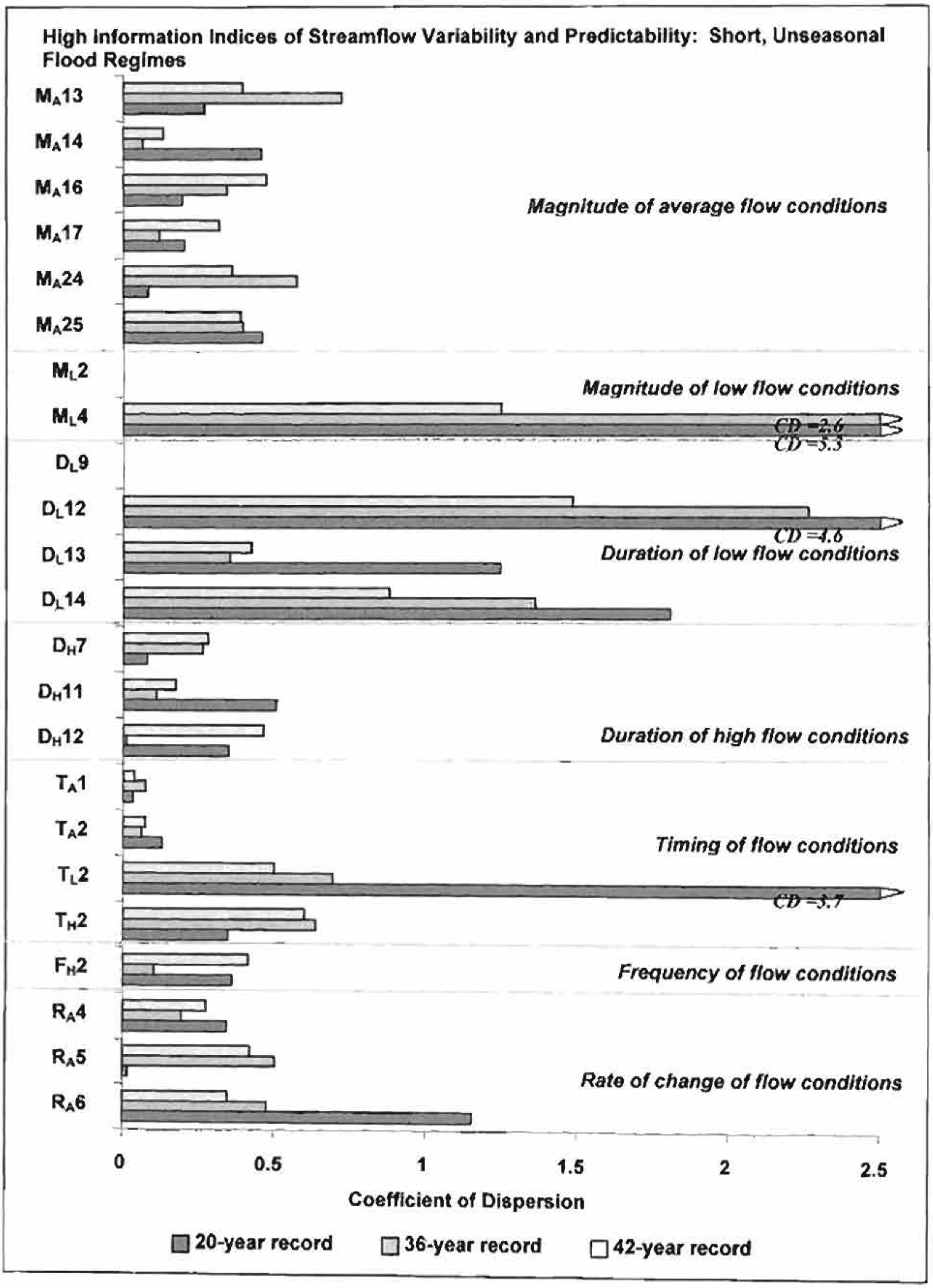

Figure 5.39 Coefficient of Dispersion of high information indices of streamflow variability and of streamflow predictability of the flow conditions of Short, Unseasonal Flood Regimes found in South African (across different record lengths). See Table 5.13 for sample sizes.

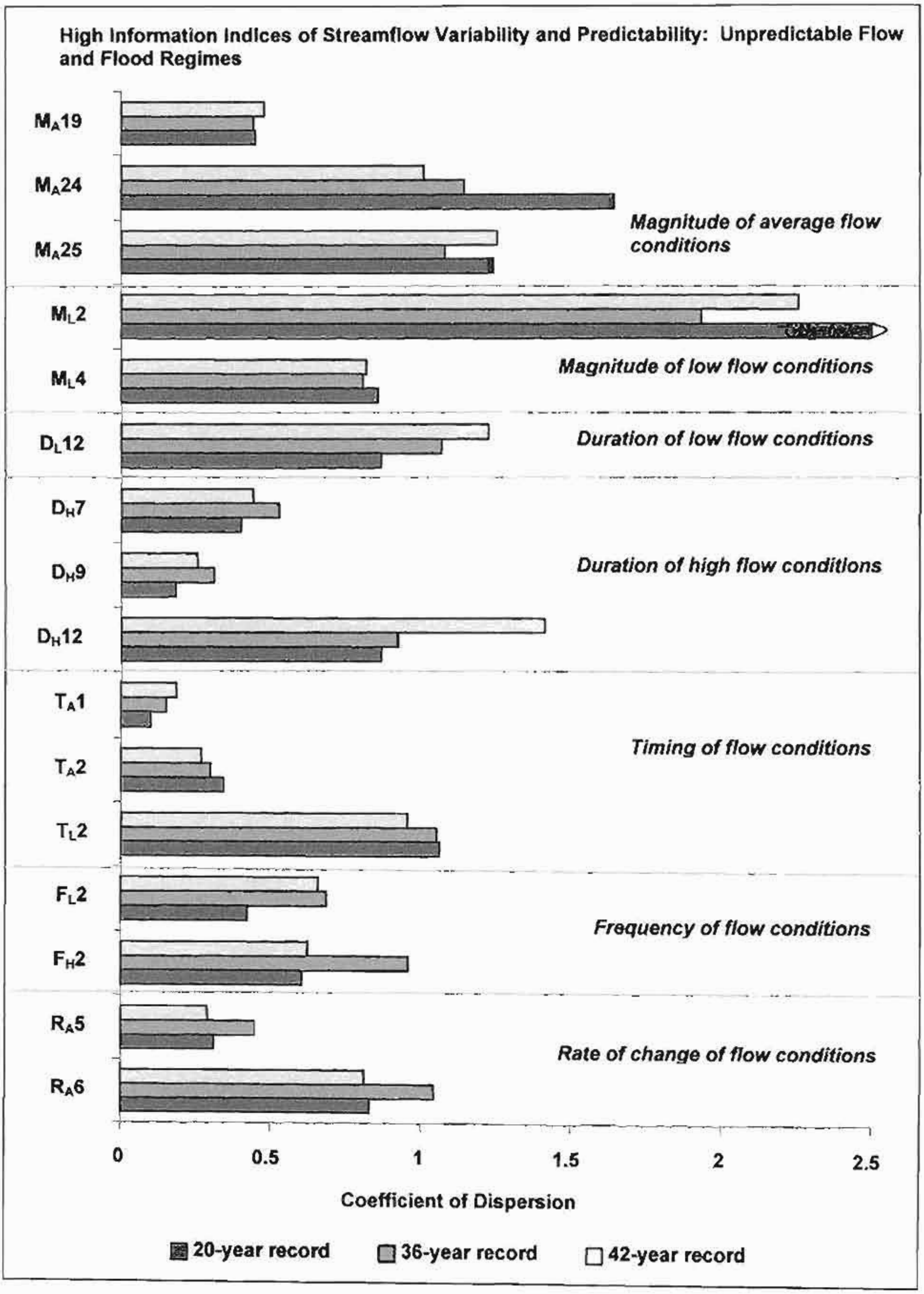

Figure 5.40 Coefficient of Dispersion of high information indices of streamflow variability and of streamflow predictability of flow conditions of Unpredictable Flow and Flood Regimes found in South African (across different record lengths). See Table 5.13 for sample sizes.

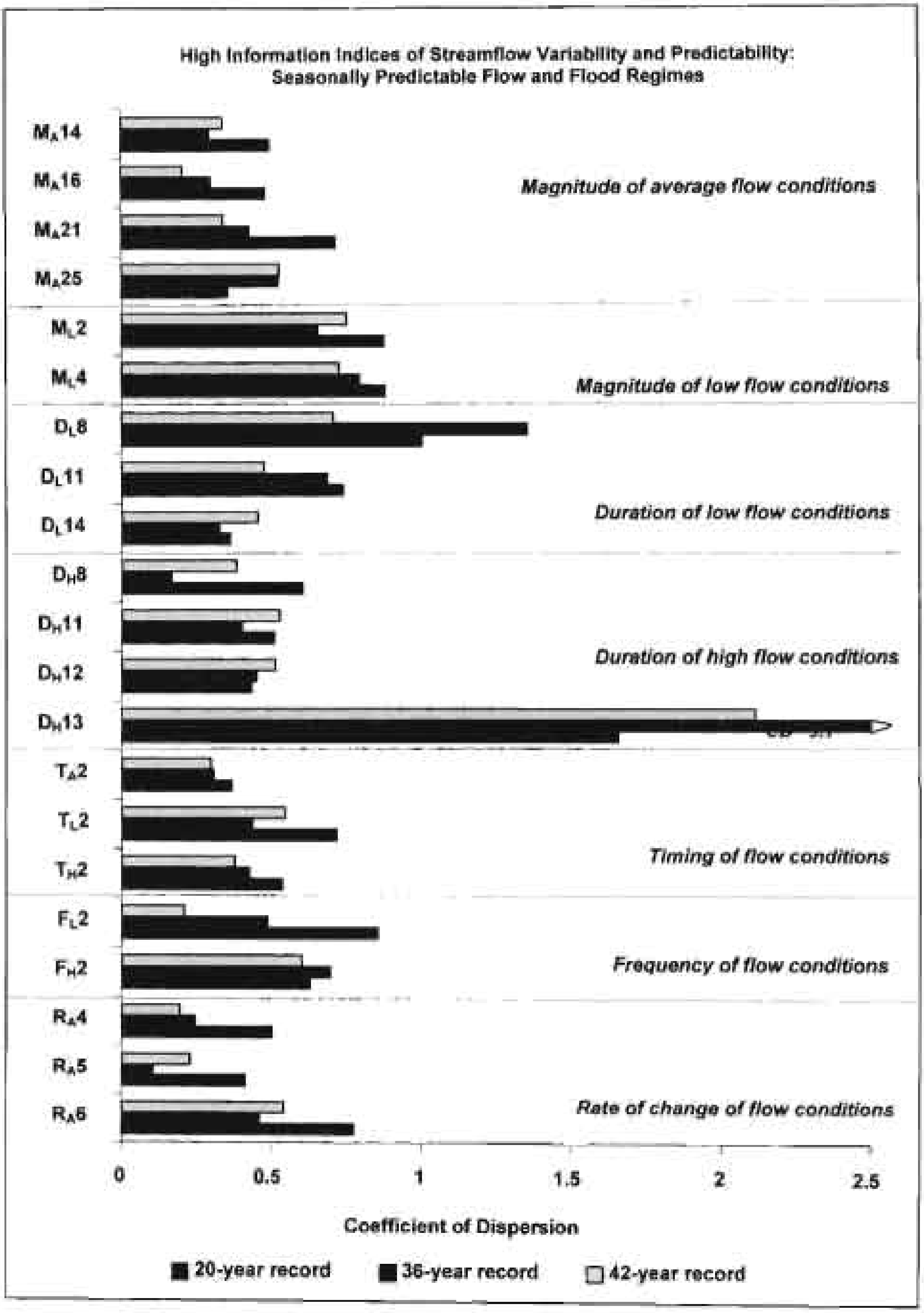

Figure 5.41 Coefficient of Dispersion of high information indices of streamflow variability and of streamflow predictability of the flow conditions of Seasonally Predictable Flow and Flood Regimes found in South African (across different record lengths). See Table 5.13 for sample sizes.

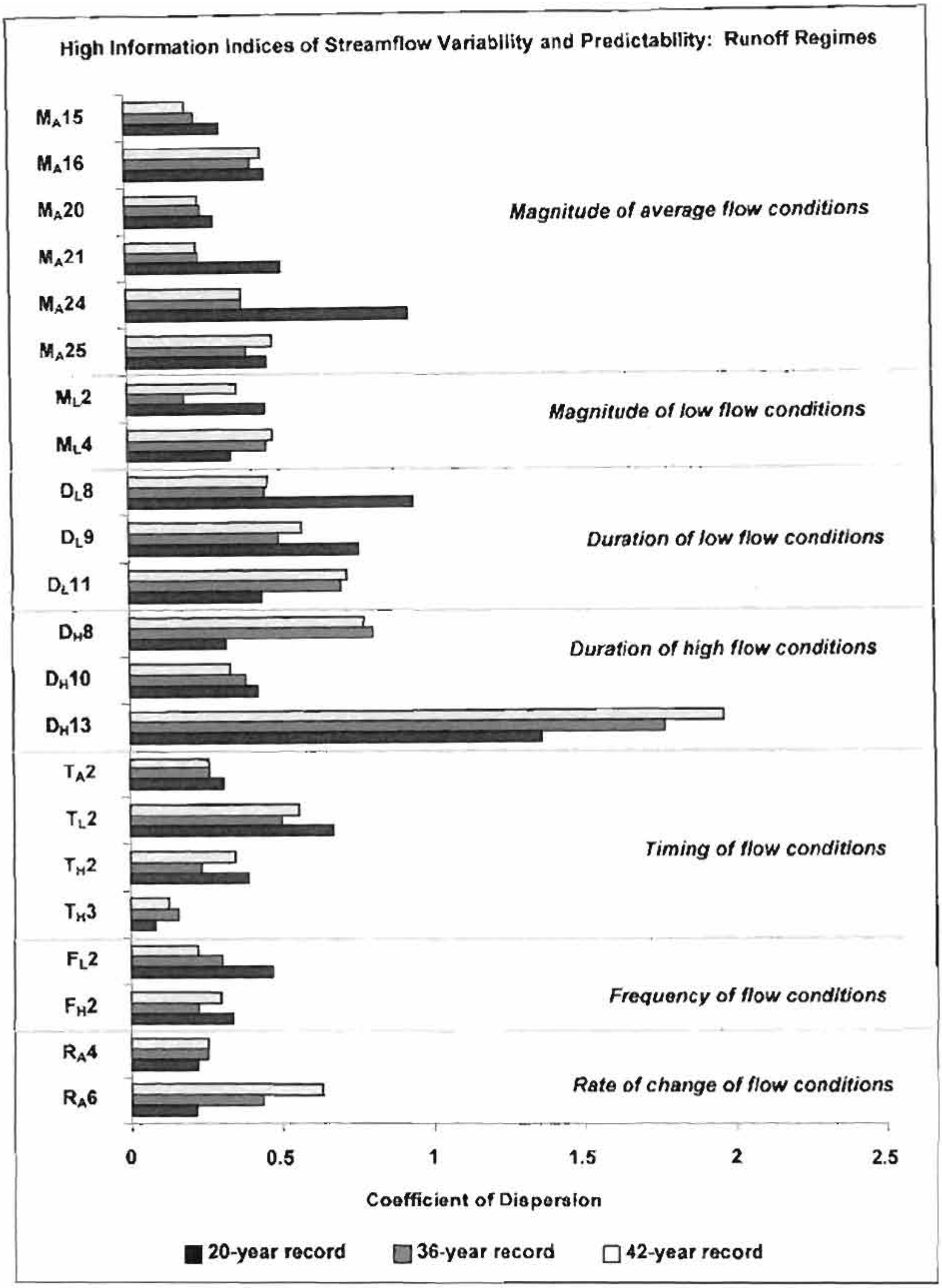

Figure 5.42 Coefficient of Dispersion of high information indices of the streamflow variability and of streamflow predictability of the flow conditions of Runoff Regimes found in South Africa (across different record lengths). See Table 5.13 for sample sizes.

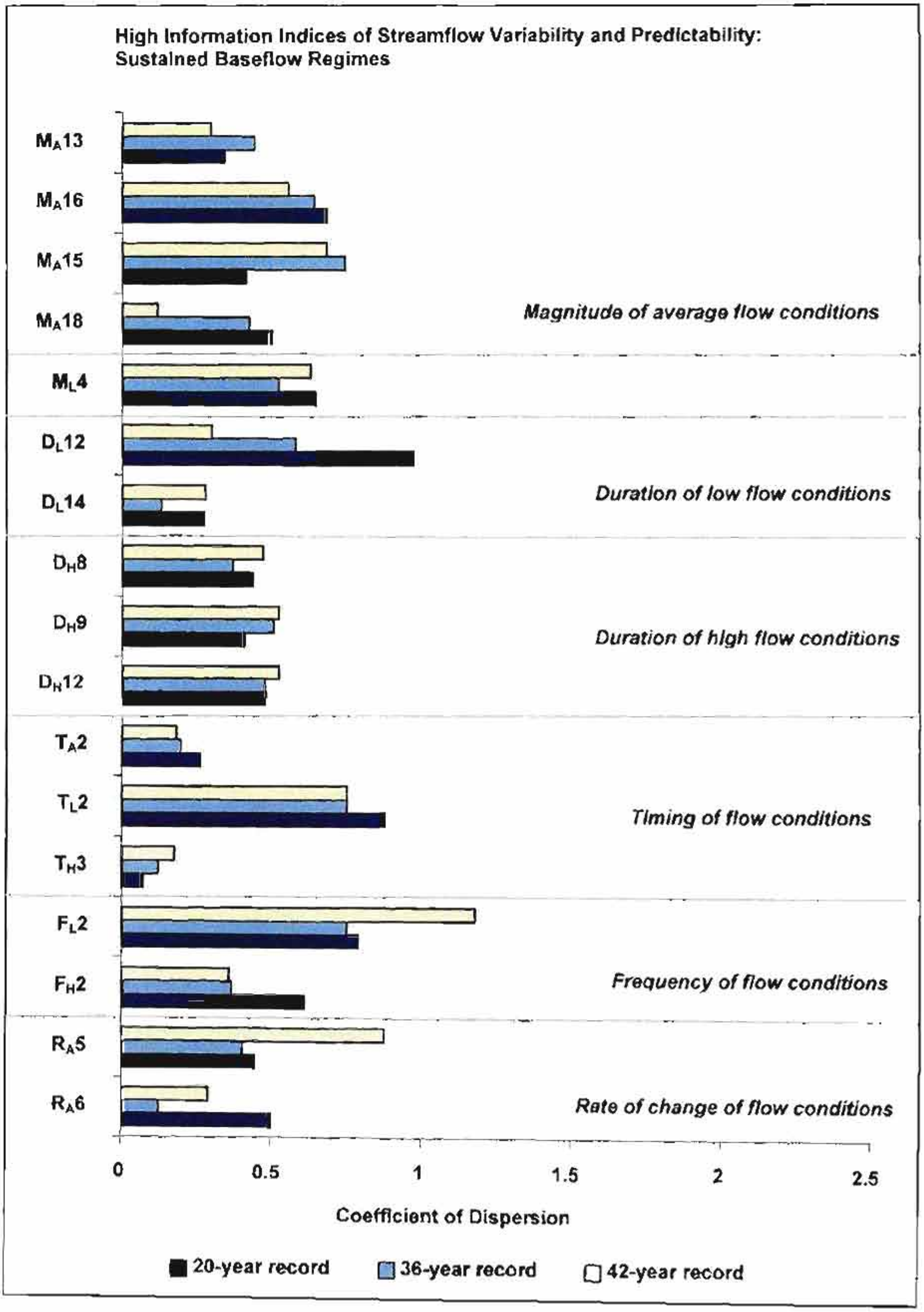

Figure 5.43 Coefficient of Dispersion of high information indices of streamflow variability and of streamflow predictability of the flow conditions of Sustained Baseflow Regimes found in South African (across different record lengths). See Table 5.13 for sample sizes.

(Figure 5.37). However, since these indices form a subset of those described in the immediately preceding paragraphs, they need no further discussion in this section; they are included solely for completeness.

Extreme Seasonal regimes

Figure 5.38 shows that the CDs of the "at-site" values of the high information hydrological indices of streamflow variability and of streamflow predictability for this group are most consistent across the different records lengths for M_A22 (variability in flows in July), T_A1 (predictability of flows), T_L2 (variability in the date of the minimum flow), F_H2 (variability in the number of high pulses) and D_L13 (variability in the number of days with zero flow) and to a lesser extent for R_A4 and R_A6 (variability in rise rate and the number of hydrograph reversals). Twenty years of record are adequate for assessments of T_A1 (the predictability of flows), and F_H2 (variability in the number of high pulses). In addition, the CDs for both M_A22 (variability in flows in July) and T_L2 (variability in the date of the minimum flow) suggest that these indices are so consistent for the ephemeral regimes of these streams (CD values of 0 for each of the records analysed at each of the sites) that less than 20 years is sufficient for their reliable assessment, at this "stream type" scale.

The 36-year record was found to be just as reliable as the 42-year record for the assessments of R_A6 (variability in the number of hydrograph reversals). However, records in excess of 36 years are preferable for M_A15 and M_A19 (variability required for flows in December and April), M_A25 (the CDB index based on the Desktop Reserve model CVB index of overall variability), D_L13 (variability in the number of days with zero flow) and R_A4 (variability associated with rising river levels). Reliable assessments of both D_H8 (variability in the 3-day maximum flow) and D_H13 (the seasonal predictability of non-flooding) require longer than 36 years of record.

Short, Unseasonal Flood regimes

Figure 5.39 indicates that, in general, longer records are needed to produce reliable assessments of the high information indices of streamflow variability and of streamflow predictability of streams in this group than the records required for streams in the Extreme Seasonal group. Only M_L2 (variability in the IHA Baseflow Index) and D_L9 (variability in the 3-day minimum) can be assessed from less than 20 years of record. Nonetheless, 36 years of record can still provide reliable assessments of M_A25 (CDB, the Desktop Reserve model overall index of variability), D_H7 (variability in the annual maxima), T_A1 and T_A2 (predictability and constancy of flows) and T_H2 (variability in the date of the annual maximum flow). The remaining high information indices of the streamflow variability and of streamflow predictability of these streamflow regimes all require more than 36 years of record to stabilise and comprise the following:

(a) the indices of variability in monthly flows at the end of winter (M_A24, M_A13 and M_A14, *i.e.* September, October and November) as well as the during summer (M_A16 and M_A17, *i.e.* January and February),

(b) M_L4 (Q75),

(c) R_A4, R_A5 and R_A6 (variability in the rate of change of flow conditions), and

(d) indices of high flow conditions (*i.e.* D_H11 and D_H12, *i.e.* variability in the 30-day maximum flow and number of high pulses) as well as

(e) low flow conditions (*i.e.* D_L12, D_L13 and D_L14, *i.e.* variability in the 90-day minimum flow, number of days with zero flow as well as number of low pulses).

The greatest discrepancies among the record length requirements for reliable indices of these streams is for M_L4 (Q75), D_L12 (variability in the 90-day minimum flow), T_L2 (variability in the date of the annual minimum flow) and to a lesser extent D_L14 (variability in the duration of low pulses).

Unpredictable Flow and Flood regimes

High information indices of streamflow variability and of streamflow predictability of the streamflow regimes comprising the ***Unpredictable Flow and Flood*** group show little discrepancy among the record lengths required to stabilise most indices. Nonetheless, only M_A19 (variability in the flows in April) and M_L4 (Q75), can be assessed reliably from only 20 years of record. Figure 5.40 shows that M_A25 (the index of overall variability) and M_L2 (variability in the IHA Baseflow index) both require more than 36 years of record to stabilise.

Thirty-six years of record are sufficient for the assessment of D_H7 and D_H9 (variability in both the 1-day and the 7-day maximum flow) as well as T_A1 and T_A2 (predictability and constancy of flows) and F_H2 (variability in the number of high pulses) among these streams. The remaining high information indices of streamflow variability of these streamflow regimes all require more than 36 years of record to stabilise, including M_A24

(flows in September), D_H7 and D_H9 (both the 1-day and the 7-day maximum flows), T_L2 (date of the minimum flow), F_H2 (the number of high pulses); and R_A5 and R_A6, (the rate of falling river levels and number of hydrograph reversals respectively). D_L12 (90-day minimum flow) and D_H12 (duration of high pulses) both require even longer records.

Seasonally Predictable Flow and Flood regimes

Figure 5.41 indicates that most of the high information indices of streamflow variability and of streamflow predictability of ***Seasonally Predictable Flow and Flood*** regimes need more than 36 years of record to stabilise. Only M_A25 (the CDB index of overall variability) and T_A2 (related to the constancy of the flow regime) could be reliably assessed from 36 years of record. The other high information indices all require more than 36 years to stabilise. The greatest discrepancy, relating to CD values, among the record lengths for these streams is for index D_H13 (the seasonal predictability of non-flooding). Moreover, the CD values of D_H13 for the streams in this Group are considerably larger than those of the other high information indices, indicating the difficulty encountered when making assessments of this streamflow characteristic of Seasonally Predictable Flow and Flood regimes.

Runoff regimes

Figure 5.42 indicates that the 36-year record is sufficient for assessments of M_A15, M_A20, M_A21 and M_A24 (variability in flows in December, May, June and September), M_L4 (the Q75), D_L8 and D_L11 (variability in the 1-day minimum and the 30-day minimum flows), D_H13 (seasonal predictability of flooding), T_A2 (related to the constancy of the flow regime), and R_A4 (variability in the rise rate of flows). However, most of the indices of streamflow variability and of streamflow predictability of streams in the Runoff group require more than 36 years of records for reliable assessments. For indices representing the magnitude of flow conditions these are M_A16 (variability in January) as well as M_A25 (the CDB index of overall variability). Indices of high flow conditions (D_H8 and D_H10, variability in the 3-day and of the 30-day maximum flows; T_H2, variability in the date of the annual maximum flow and F_H2, variability in the number of high pulses) and of low flow conditions (D_L9, variability in the 3-day minimum flow; T_L2, variability in the date of the annual minimum flow and F_L2, variability in the number of low pulses) also, generally, require more than 36 years to stabilise for the streams in this Group.

High seasonal predictability of non-flooding and frequent hydrograph reversals are two of the three indices (short frequent floods being the third) which distinguish the "flashy" nature of the streams in this Group (*c.f.* Table 5.10), yet indices R_A6 (variability in the number of hydrograph reversals) and D_H13 (seasonal predictability of non-flooding) probably require more than 42 years for reliable assessments of their values.

Sustained Baseflow regimes

Thirty-six years of record are sufficient for assessments of T_A2 (the proportion of predictability attributed to constancy), T_L2 (variability in the date of the annual minimum flow), and F_H2 (variability in the number of high pulses) of streams in the ***Sustained Baseflow*** group (Figure 5.43). However, more than 36 years of record would be beneficial in the assessment of all the other high information indices of streamflow variability and of streamflow predictability. Index T_H3 (seasonal predictability of flooding) may require more than 42 years of record to stabilise.

The results of the comparisons of record length required to stabilise the high information hydrological indices of streamflow variability and of streamflow predictability, for each of the streamflow types described in the immediately preceding sections, are summarised in Table 5.19.

Summary of results of the analysis of the record length required for reliable indices of streamflow variability and of streamflow predictability

The focus of the discussion above relates to only the high information indices of streamflow variability and of streamflow predictability and, consequently, some indices have been omitted from the Table for the same reasons as those provided in Section 4.5.4.1. Again, the number of high information indices varied among the different streamflow groups, as did the number of stream sites within each of the distinct groups. Despite these differences of sample size, there are still some relevant general observations can be drawn from the results shown in Table 5.19.

(a) Dominant indices of the streamflow variability in the *magnitude of average flow conditions* generally require at least 36 years of record, and for a number of indices, longer records to stabilise. However, the streamflow patterns distinguishing the Runoff Group indicate that 36 years of record is sufficient for reliable assessments of these flows conditions in winter months.

Table 5.19 Record lengths required to stabilise the high information hydrological indices of streamflow variability and of streamflow predictability for different streamflow regime types. Blanks represent less useful indices (according to the results of the PCA).

Hydrological Index	Streamflow Flow Type						
	Extreme Seasonal	Mixed			Perennial		All Streams
	Extreme Seasonal	Short, Unseasonal Floods	Unpredictable Flow and Floods	Seasonally Predictable Flow and Floods	Runoff	Sustained Baseflow	All-43 streams
M_A13		>30 years preferable				>30 years preferable	
M_A14		>30 years preferable		>30 years preferable			
M_A15	>30 years preferable				>30 years preferable	>30 years preferable	
M_A16		>40 years preferable		>30 years preferable	>30 years preferable	>30 years preferable	>30 years preferable
M_A17		>30 years preferable					>30 years preferable
M_A18						>30 years preferable	
M_A19	>30 years preferable		20-year record sufficient				
M_A20					30-year record sufficient		
M_A21				>30 years preferable	30-year record sufficient		>30 years preferable
M_A22	<20 years sufficient						>30 years preferable
M_A24		>30 years preferable	>30 years preferable		30-year record sufficient		20-year record sufficient
M_A25	>30 years preferable	30-year record sufficient	>30 years preferable	30-year record sufficient	>30 years preferable		30-year record sufficient
M_B2		<20 years sufficient	>30 years preferable	>30 years preferable	>30 years preferable		>30 years preferable
M_B4		>30 years preferable	20-year record sufficient	>30 years preferable	>30 years preferable	>30 years preferable	>30 years preferable
D_L8				>30 years preferable	30-year record sufficient		>30 years preferable
D_L9		<20 years sufficient			>30 years preferable		
D_L10							>30 years preferable
D_L11				>30 years preferable	30-year record sufficient		
D_L12		>30 years preferable	>40 years preferable			>30 years preferable	>30 years preferable
D_L13	>30 years preferable	>30 years preferable					
D_L14		>30 years preferable		>30 years preferable		>30 years preferable	
D_H7		30-year record sufficient	>30 years preferable				
D_H8	>40 years preferable			>30 years preferable	>30 years preferable	>30 years preferable	
D_H9			>30 years preferable			>30 years preferable	
D_H10					>30 years preferable		
D_H11		>30 years preferable		>30 years preferable			
D_H12		>30 years preferable	>40 years preferable	>40 years preferable		>30 years preferable	>40 years preferable
D_H13	>40 years preferable			>30 years preferable	>40 years preferable		>30 years preferable
T_A1	20-year record sufficient	30-year record sufficient	30-year record sufficient				30-year record sufficient
T_A2		30-year record sufficient	30-year record sufficient	30-year record sufficient	30-year record sufficient	30-year record sufficient	30-year record sufficient
T_L2	<20 years sufficient	>30 years preferable	>30 years preferable	>30 years preferable	>30 years preferable	30-year record sufficient	
T_H2		30-year record sufficient		>30 years preferable	>30 years preferable	>30 years preferable	
T_H3					>30 years preferable	>40 years preferable	30-year record sufficient
F_L2			30-year record sufficient	>30 years preferable	>30 years preferable	>30 years preferable	>30 years preferable
F_H2	20-year record sufficient	>30 years preferable	>30 years preferable	>30 years preferable	>30 years preferable	30-year record sufficient	>30 years preferable
R_A4	>30 years preferable	>30 years preferable		>30 years preferable	30-year record sufficient		30-year record sufficient
R_A5		>30 years preferable	>30 years preferable	>30 years preferable		>30 years preferable	
R_A6	30-year record sufficient	>30 years preferable	>30 years preferable	>30 years preferable	>40 years preferable	>30 years preferable	30-year record sufficient
Key	<20 years sufficient	20-year record sufficient	30-year record sufficient	>30 years preferable	>40 years preferable		

(b) With the exception of the streams in the Short, Unseasonal Floods Group, more than 36 years of record are preferable for reliable assessments of the streamflow variability associated with the indices representing the *magnitude of low flow conditions*.

(c) While 36 years of record is generally sufficient for the assessment of indices defining the variability associated with *duration of low flow conditions* for streams at the moderate end of the environmental gradient, longer records are preferable for the streams in the transitional portion, as well as the arid end of the range.

(d) Indices defining the variability in the *duration of high flow conditions* generally require in excess of 36 years of record across all the distinct streamflow types.

(e) Thirty-six years of record are generally more than adequate for the stability of indices representing the predictability of the *timing of average flow conditions*.

(f) With the exception of the streams at the arid end of the environmental range, more than 36 years of record are preferable for assessing indices representing the streamflow variability in the *timing of both high and low flow conditions*.

(g) In general, more than 36 years of record are preferable for assessing indices of variability in the *frequency of either high or low flow events*. However, the variability in the *frequency of high flow events* can be assessed reliably from shorter records for streamflow regimes at either end of the environmental range.

(h) In general, more than 36 years of record are preferable for assessing indices of the streamflow variability in the *rates of change of flow conditions*.

As in the analysis of the record length required for reliable indices of central tendency, this summary indicates that, depending on the streamflow type, *more than 36 years of record is preferential* for reliable assessment of many of the indices of streamflow variability and of streamflow predictability examined. Again, while this information may seem inconsistent with the main research component of this Study *i.e.* the application of PCA of hydrological indices derived from streamflow records spanning only 36 years of record, the points emphasised previously are also relevant to the findings of the summary immediately above, for the same reasons.

4.6 Summary

PCA was applied to a relatively large hydrological dataset (74 hydrological indices for 83 stream sites) to highlight dominant patterns of intercorrelation among the indices and to identify reduced sets of indices which explain a dominant proportion of statistical variation in the entire set of the indices and which adequately represent the different facets of the streamflow regimes found in South Africa. The PCA provided information on both the streamflow sites and the hydrological indices. Six stream types were identified which ranged over a general environmental gradient from "extreme seasonal", or "harsh arid", conditions represented by ephemeral river systems, through quasi-perennial-seasonal regimes to the more moderate regimes found in perennial systems. Statistical significance of the principal component axes was evaluated using the Broken-stick model. The number of statistically significant principal component axes ranged from four (*e.g.* the Extreme Seasonal group) to six (*e.g.* the Runoff group) and together explained 75.74% of the variation for the combined set of stream types, "All-83 Streams". Two main clusters of inter-correlated indices were identified from the two-dimension ordination illustrating the major patterns of inter-correlation among the 74 indices of the combined set of "All-83 Streams". One cluster comprised indices of central tendency, whereas the other comprised indices of dispersion. For each streamflow type, 12 high information indices that describe the majority of variation provided by the entire set of 74 indices were identified for use in eco-hydrological studies. However, choices may be necessary, as subsets of indices representing the same principal component axis are not independent from each other. A more beneficial approach may be to select the seven indices with the highest loadings on the first principal component axis, but with the condition that each index represents one of the (seven) main streamflow characteristics. This would account for the majority of the variation among the indices while providing a representation of each of the main streamflow characteristics for any further analysis or ecological flow assessment. If required, and given the particular ecological study, these indices could be supplemented with choices of high information indices, which are relatively independent from each other, within each streamflow characteristic, from the remaining significant principal component axes. Some dominant indices are unique to streamflow types (*e.g.*M_L3, the Desktop Reserve model baseflow index for the Sustained Baseflow type), whereas others share a broader commonality among the different streamflow types (*e.g.* D_H8, the variability in the 3-day annual maximum flow). There are differences among the streamflow types regarding the length of record required for reliable assessments of each of the high information hydrological indices. However, depending on the streamflow type, *more than 36 years of record is preferential* for reliable assessment of many of the indices examined.

5 DISCUSSION ON THE ANALYSES OF THE WORKING DATABASE

Streamflow Records

Measurements of streamflow characteristics are important ecological indicators (Poff, 1989; 1996; Richter *et al.*, 1996; 1997). Using the long term record of daily streamflow to derive these measurements (hydrological indices) has advantages over other environmental datasets since the records are usually longer, more complete and more reliable than a collection of data representing an isolated biotic survey. However, the "long term" records should be of "good" or "better" quality. Defining "long term" "good" or "better" records differs for different regions (*c.f.* Appendix 5A).

Grouping rivers by region, or streamflow type, requires data (observed or modelled) on "pristine" or reference flow patterns. This presents problems where gauging stations include the impacts of land use change in the record of observed streamflows, since inclusion of the impacts of "present" land use "could result in rivers being grouped without meaning, and without reflection of local climatic and other conditions" (Joubert and Hurly, 1994). Also, the selection of gauging stations that record streamflows from unimpounded or unaltered catchments is often restricted to the headwater regions where rainfall is higher than in areas of lower reaches (Joubert and Hurly, 1994).

The biggest reservation of this Study lies with the reliability of the DWAF streamflow records. This concern was also raised by Joubert and Hurly (1994) who acknowledged that the records of the DWAF gauging stations used in their study (the 1994-Study) may not be representative of "natural" streamflows. The techniques employed in Section 3 of this Study to screen the DWAF streamflow records using tests for stationarity are considered to be an improvement on the "double-mass" analysis adopted by Joubert and Hurly (1994). However, establishing lack of trend and stationarity as well as homogeneity does not satisfy concerns that the records are, indeed, adequate representations of natural streamflows at the sites. In addition, as stated in Section 3.4 of this Chapter, an inherent assumption in the alternative methods applied to test for the absence of trend and for stability of the median as well as dispersion is that the rainfall over the sub-continent is free from any climatic change. Nonetheless, the most critical concern is that information relating to high flows is not always captured, a problem that arises where the DWAF gauging plate is set too low in the river or when high flows disturb the gauging plate, rendering any subsequent measurements erroneous. Where these features are obvious in the record, or where it is known that high flows "overtop" the gauging plate, the record data are of limited value. Every effort was applied in this Study to ensure that such anomalies were minimised.

Missing days in the record are problematic when deriving hydrological indices. Given the relatively short record of most of South Africa's gauging network, and in particular of a common time span required for comparison between a sufficiently large sample of records, it makes sense to maximise the potential of available records, even where there are "missing days". For example, where there is a clear seasonal climatic pattern, reduction in rainfall events leads to a gradual reduction in daily streamflows and infilling values on missing days is not an unreasonable practice.

In general, the records representing the DWAF streamflow gauging network are well maintained and accessible. However, it is pertinent to note that several of the gauging stations used in the analyses of "natural" streamflows in the 1994-Study have not been updated since the early to mid-1990s (*c.f.* Table 5.1). The historical study of any streamflow regime requires *long-term* records and the curtailment of these records is a matter for concern. While it can be argued that hydrological models are developed to fill this gap, verification of simulated streamflows with observed streamflow data is essential for the scientific defensibility of the simulations. Moreover, as the demand to simulate the complexities associated with whole ecosystems at finer temporal resolution grows, there will be even greater need to verify the results produced by hydrological models.

Notwithstanding these very important concerns, South Africa's streamflow records are routinely monitored and edited by the DWAF. Together with the screening procedure described in Section 3.4 of this Chapter, this author is reasonably confident that the records applied in this Study are the best available representation of the natural streamflow regimes found in different hydrogeographic regions of South Africa.

Grouping Rivers by Flow Type

While a streamflow classification based on hydrological indices of ecological interest was not the primary aim of this Study, regrouping the rivers that were previously clustered together (Joubert and Hurly, 1994) was appropriate, since the analysis presented here was performed using a *common record length* among the DWAF gauging stations which, in addition, extended 11 years beyond the most up-to-date record used in the 1994-Study. Principal Component Analysis generates information on both objects (the stream sites) and descriptors (hydrological indices) and a clear environmental gradient of streamflow patterns and conditions, ranging from those in "harsh-arid" (*i.e.* the Extreme Seasonal group) to more "moderate" (*i.e.* the Sustained Baseflow group) hydrogeographic regions, emerged as a result of plotting the scores for each of the 83 gauging stations for the first two principal components from the 74 x 74 correlation matrix of hydrological indices. Consequently, the streamflow classification presented here represents an updated hydrogeographical description for the relatively undisturbed rivers in South Africa. The usefulness of this classification is that the ecologically relevant hydrological indicators identified as being of high information (Olden and Poff, 2003) for each of the different streamflow types have value for application in ecohydrological studies of similar, yet unmonitored, rivers. Nonetheless, despite the usefulness of the streamflow classification derived for this Study, it is recognised that further research for grouping rivers by flow type may benefit from a different statistical approach such as cluster analysis.

The Hydrological Indices

Where other resources are limited, the long-term streamflow record of daily average flows can be used to derive a relatively large dataset of ecologically relevant hydrological indices. Measures of the central tendency and the dispersion of the inter-annual statistics of intra-annual streamflow "characteristics", or "indices", (*i.e.* monthly flows; flood frequency, peak discharge, and baseflows), are used routinely in environmental flow assessments of the streamflow patterns required to maintain ecological functioning. Consequently, desktop approaches such as the IHA method (Richter *et al*, 1996) which is supported by Windows-based computer software, and within which a suite of ecologically relevant hydrological indices is calculated, have received attention in recent years (Olden and Poff, 2003).

Up to the time of writing, Olden and Poff (2003) provide the most comprehensive examination of the degree of intercorrelation among existing (171) hydrological indices, including the IHA indices, in the literature. Their analyses found that "the IHA method adequately represents the majority of variation explained by the entire population of 171 indices relating to the major components of the streamflow regime for streams in the USA and thus captures the majority of the information available" (Olden and Poff, 2003, page 113). In their Study, Olden and Poff sought to ascertain whether the high information hydrological indices are transferable between differing hydrogeographical regions, both continentally and globally. Their findings were that "while particular hydrological indices may be transferable among particular streamflow types, in general, the choice of hydrological indices should reflect the specific hydro-climatic characteristics of the study region" (Olden and Poff, 2003, page 113).

Until now (2005) there has been no comprehensive examination of the usefulness of the IHA method and indices for the diversity of streamflow types found in South Africa. Thus, this Study provides the first examination of the degree of intercorrelation among not only the IHA indices, but also the Desktop Reserve model indices (which are based on the results from detailed IFR determinations by groups of aquatic and riparian specialists and used in the desktop approach to the ecological flow requirements of rivers systems in South Africa) for the differing hydrogeographic regions in the sub-continent. The approach used in this Study follows that adopted by Olden and Poff (2003). While this cannot be considered to be innovative, there are few similar studies (but see Clausen and Biggs, 2000 for a study of 62 rivers in New Zealand). In addition, it was considered most appropriate to rise to the challenge of testing the suitability of the IHA indices for South African river systems by applying multivariate analysis (in this instance PCA) to identify subsets of high information hydrological indices that explained a dominant proportion of statistical variation in a relatively comprehensive set of indices (including the Desktop Reserve model indices). Moreover, it is pertinent to ascertain whether the IHA indices are transferable and meaningful across the streamflows regimes found in South Africa.

The merit of the PCA presented here is its ability to identify a subset of hydrological descriptors that contains the most information in a relatively large set of hydrological data (74 indices) for the diversity of streamflow types found in South Africa. Following Olden and Poff (2003), this Study shows that by applying PCA, the total number of ecologically relevant hydrological indices examined can be reduced from 74 to between four and six indices (if one takes the index with the highest loading on each statistically significant principal component axis, Table 5.15). This elementary selection could have value for the preliminary stages of environmental flow (or impact) assessments, since at least three, and up to four, of the seven (*c.f.* Section 4.4.3 of this Chapter) major characteristics of the streamflow regime would be represented (*e.g.* magnitude of low flow conditions; duration of low flow conditions and duration of high flow conditions for streams in the Sustained Baseflow group, *c.f.* Table 5.15). Alternatively, the total number of indices could be reduced from 74 to seven, if one selected the index with the highest loading on the first principal component axis for each of the major streamflow characteristic (Table 5.16). If additional indices within each streamflow characteristic are required, they could be selected from each of the other significant principal component axes without any incurring any substantial increase in redundancy (Olden and Poff, 2003). The latter approach would provide a greater representation of the most important facets of the natural streamflow regime and which are essential to sustaining the ecological functioning of each of the differing streamflow types. However, as emphasised by Olden and Poff (2003), *where possible, decisions regarding the indices should be made in conjunction with ecological information.* In this way, indices describing a streamflow component which is closely related to an ecological concern (*e.g.* if the timing of flows is important for a particular life stage) can be selected from the high information indices identified by the PCA.

The results presented in Section 4.5.3 of this Chapter provide a framework for the selection of high information, non-redundant hydrological indices that represent the major gradients of variation described by the entire set of indices (74) of the different streamflow types. However, the selection of high information, non-redundant indices should also be guided by the particular ecological study (Olden and Poff, 2003). The high information indices for each group were discussed in the Section 4.5.3. Consequently, only the salient points need be repeated here.

CDB (M_A25), based on the Desktop Reserve model index of *overall variability*, possesses a large amount of information about the major gradients of variation among the indices, across different stream types and river systems in South Africa. This finding substantiates the relationship between the CVB index and the IFR for the ecological functioning of South Africa's rivers as described by Hughes and Hannart (2003). The CDB index is an important statistical measure of the overall variability required by ecosystems in harsh or intermittent streams, where the value is generally high (Hughes and Hannart, 2003). The merit of this index is that it features as being of high information for the diversity of streamflow conditions comprised in the "All-83 Streams" group. Consequently, the CDB index may be important for broad spectrum environmental studies. In addition, Alt-BFI (M_L3), based on the Desktop Reserve model BFI index of short-term variability, contains the most information in the entire set of 74 indices for the streamflow regimes comprising the Sustained Baseflow group. The baseflow time series used to calculate the BFI has been associated with the groundwater component of the South African Reserve (*c.f.* Section 3.3 of Chapter 4). Thus, maintaining the natural variability in the baseflow component for rivers in this Group is beneficial for sustaining ecological functioning.

On the other hand, of the set of 74 indices, the IHA Baseflow index M_L1 has limited relevance for most South African rivers. This index does not reflect the baseflow regime but rather an extreme low flow. The calculation of this index is based on the proportion of the 7-day minimum flow to the mean daily flow for the year, and for many rivers has a low, or in some instances, zero value. It could be argued that this finding is a shortcoming of the approach used in this Study, since PCA should ideally not be conducted on datasets, with many zero values (Legendre and Legendre, 1998). However, where zero values are the actual values or observations, their inclusion is acceptable (Ndlovu, 2004). Notwithstanding these concerns, M_L1 is a dominant index of hydrological variability for the combined set of "All-83 Streams" and could also be used for large scale environmental studies.

As expected, hydrological indices which describe either high flow or low events are important for all the streamflow types examined, particularly those relating to the duration of extreme events. Various high flow events are important for the Runoff and Sustained Baseflow, Seasonally Predictable Flow and Flood and Extreme Seasonal groups, including the annual flood pulse (*i.e.* D_H1), as well as the multi-day, monthly and seasonal floods (*i.e.* D_H2, D_H3, D_H4 and D_H5) which trigger critical ecological processes and functioning and generate the ecosystem goods and services most desired by society (Bayley, 1995). In addition, the dispersion-based index, D_H8, is an important measure of variability in short-duration high flows, which may act as a climatic cue for spawning of species (Bunn and Arthington, 2002) or seed dispersal (Shafroth *et al.*, 1995). On the other hand, various spells of low flow conditions (*i.e.* D_L1, D_L2, D_L3 or D_L4) are important for all groups except for those at the harsh-arid end of the environmental range, where the degree of cessation of flow and variability in the seasonal low flow conditions (D_L6 and D_L12) are more important.

The degree of cessation of flow (D_L6) is statistically important for the Extreme Seasonal, Short Unseasonal Floods and Unpredictable Flow and Floods Regimes. These three groups represent the arid end of the environmental gradient of the streamflow types found in South Africa. Aquatic and riparian biota in these environments have adapted to harsh conditions, where the cessation of streamflow is required to stress ecosystems, thereby increasing the genetic diversity of biological populations and communities and enhancing ecosystem resilience. Consequently, this hydrological index may be useful for describing the duration of extreme low flows required for streams in relatively arid regions. Moreover, the low magnitude of flow conditions in October (M_A1) may also be critical for harsher environments (typically streams in the Extreme Seasonal and Short Unseasonal Flood groups) for similar reasons. On the other hand, the higher magnitude of flows in October (M_A1) in the Runoff streamflow regimes may be important for the first of season flushing flows. This concurs with the emphasis afforded to October flushing flows in the BBM assessment of environmental flow requirements for rivers in these hydrogeographic environments.

With perhaps the exception of streams comprising the Extreme Seasonal group, the dispersion-based indices of the variability in monthly flows (M_A13 to M_A24) are important across the entire spectrum of streamflow types. This substantiates the need for environmental flow assessments to maintain the inter-annual dynamics of the magnitude of monthly flows, a feature which is fundamental to the concept of the EFA in South Africa (*c.f.* Section 3.2 of Chapter 4).

This Study indicates that the timing of the date of the minimum flow (T_L1 for central tendency of the date and T_L2 for variability in the date) is relevant only for streams in the Extreme Seasonal group. However, it has to be noted that this is most likely a function of the derivation of this index by the IHA software which recognises the first day of the hydrological year (1 October) as the Julian date of the minimum flow in any given year, even although the minimum flow event occurs in the main winter months of May to August for many of South Africa's rivers. Consequently, this IHA index also has limited use for ecohydrological studies of many South African river systems. However, T_H1 (the timing of the Julian date of the maximum flow) is relevant for the Seasonally Predictable Flow and Flood regimes, indicating that at least for these streams, this IHA index is useful. Surprisingly, indices associated with the predictability of streamflow events (*i.e.* T_A1 and T_A2) could be useful measures of the variability associated with the timing of events for streams in the Short, Unseasonal Floods and Unpredictable Flow and Flood groups respectively, in the context that they are "predictably unpredictable".

Dispersion-based indices describing the variability in high flow pulses (F_H2) and rising river levels (R_A4) are important indicators of the flow conditions required by the Extreme Seasonal streams where inter-species competition and "hardy opportunism" is the survival strategy exhibited by the biota in these ecosystems (Davies *et al.*, 1994). This concurs with the findings of Olden and Poff (2003). Although their streamflow classification differed to that used in this Study, there are similarities between the Extreme Seasonal group identified in this Study and their "flashy intermittent" streams, where they found indices of high flow pulses and the rate of change in flow conditions to be dominant. In addition, high flow pulses (F_H1) and falling river levels (R_A2) are dominant indices of streams in the Runoff group in this Study. These indices are useful measures of the streamflow conditions characteristic of these "naturally flashy" streams where ecosystems have evolved to disturbance in the streamflow regime.

For the most part, hydrological indices representing the central tendency (*i.e.* the inter-annual average) of the different streamflow characteristics are sufficient measures of ecological importance for the different streamflow types. However, indices representing the dispersion (*i.e.* the inter-annual variability) in different streamflow conditions were shown to have greater relevance for streams in the Short, Unseasonal Flood group.

Comparison with Other Studies

There are limits to the comparisons that can be made between this Study and the Olden and Poff study, since not only are there differences among the streamflow classifications, but also regarding the number of indices applied. This Study used far fewer indices and was limited mainly to the IHA indices. An additional difference between the two studies was the inclusion in this Study of the indices included in the South African Desktop Reserve model indices.

However, one of the findings of the Olden and Poff study was recognition of the omission from the IHA suite of an index that directly quantified the magnitude of high flow conditions. Olden and Poff (2003) recommended that two of the measures used in their study, and which represented these flow conditions, could be included in the IHA suite since they are relatively easy to calculate. One of these is the M_H1 index applied in this Study. M_H1 was therefore included in this Study to ascertain whether it was useful for explaining the major sources of variation across the streamflow conditions found in different hydrogeographical environments across South Africa. However, this Study found that M_H1 was only dominant for the Extreme Seasonal group and is probably not worthwhile pursuing for inclusion in a desktop suite of statistical indices of the hydrological variability of South Africa's river systems.

Despite the differences between them, both studies showed that multivariate analysis can reduce the number of hydrological indices available to an ecologically relevant subset of indices that are stream-type-specific and that certain indices are transferable across different streamflow types.

Sensitivity of the Hydrological Indices to Record Length

It is important to note that the analysis of the sensitivity of the hydrological indices to record length should not be confused with the screening of time series of the 35 intra-annual indices outlined in Section 3.4 of this Chapter and Figure 5.4, where the aim was to determine whether the streamflow records were free from linear trend, stationary and homogenous across the record span and, as such, were acceptable for statistical analysis. In addition, the two exercises were conducted at different spatial resolutions. The screening exercise was performed for the streamflow records representing each of the individual stream sites, whereas the analysis of length of record required for stable indices was performed for the *overall behaviour* of the indices at a broad spatial scale (across the combined set of All-83 Streams and the reduced set of All-43 Streams) and at the resolution of the six distinct streamflow types.

The analysis of the length of record required to ensure stable estimates of the hydrological indices applied in this Study provided additional information on the usefulness of the indices for the diversity of streamflow types found in South Africa. The majority of the indices examined fall within the IHA suite of indices. Therefore, to a certain extent, this part of the Study addresses the need to evaluate the sensitivity of the IHA indices to record length in different ecoregion locations (Richter *et al.*, 1997; 1998), at least for South Africa. The findings were based on a quantitative comparison of the values of CD in each instance, rather than any statistical test. While this may be regarded as a shortcoming of the Study, it is in keeping with previous studies (*e.g.* Clausen and Biggs, 2000). Moreover, statistical tests are primarily designed to inform the researcher of the following:

(a) whether the observed difference between the average or variance of two, or more, samples is significant, or whether the difference is due to sampling error, and

(b) if a difference between the average or variance of two, or more, samples is indeed significant, what is the extent of the difference (Fowler and Cohen, 1990).

Thus, statistical testing of any difference among the record lengths analysed in this Study would not necessarily provide any information regarding the preferred length of record for stable estimates of the hydrological indices. The main benefit of the analysis of the length of record required to ensure stable estimates of the hydrological indices characterising the temporal variability in the streamflow regime, performed in this Study, is the provision of information on the sensitivity of the indices to various records lengths, at different hydrogeographical (spatial) scales. This information provides a *guide to the preferred length of record required for analysis* of the indices of different streamflow characteristics. In addition, the *extent of the sensitivity of different indices* at different spatial scales can supplement the information on the indices of high information identified by the PCA, allowing a researcher to attach a level of confidence to the usefulness of an index for any particular ecohydrological study, which in turn depends on knowledge of the streamflow type or pattern and the length of streamflow record available for use in the study.

Overall Behaviour of the Indices: Broad Hydrogeographical Scale

The nature of the analysis of the behaviour of the indices of inter-annual variability, streamflow predictability and overall variability in this Study differed to that for the indices of central tendency in that it was not possible to fully confound among site variability of the values of the indices.

Nonetheless, the overall behaviour (measured by the extent of the CD value) of the hydrological indices representing the intra-annual variability for the entire set of "All-83 Streams, could provide a useful first indication, or low confidence evaluation, of the differences among the main streamflow characteristics, regarding the extent of record required for their reliable assessment. For example, it may be useful for stakeholders to be aware that, across this broad hydrogeographical scale, longer records are required for the assessments of average flows in any calendar month (M_A1 to M_A12) than are required for assessments of the baseflow component (as described by M_L3, the Alt-BFI) or of the high flow component (as described by M_H1, the HFI). In addition, longer records are required for reliable assessments of M_L1 (the IHA baseflow index) than are required for the M_L3 at this broad spatial scale.

Behaviour among the Distinct Groups: Finer Hydrogeographical Scale

The behaviour of the indices (of intra-annual variability) among the distinct streamflow types provides a better indication, or higher confidence evaluation, of the record length required for each of the main characteristics of the streamflow regime, since in most cases a finer hydrogeographical scale is appraised. Comparison of the behaviour of these indices among the distinct streamflow types confirms the general recognition that streams in arid areas generally require longer records than those in higher rainfall regions (*c.f.* Figures 5.12 to 5.16).

However, the behaviour of the indices examined in this Study has shown that, for several streamflow indices, this is not a straightforward feature across the transition of the environmental gradient from arid to moderate conditions. While, the generalisation holds for the indices of average monthly flows (*e.g.* M_A1 to M_A12) and of the minima flows (D_L1 to D_L5), several of the streamflow regimes in the perennial main group require records of similar length, and often longer, to those in the "mixed" main group. This is evident for many of the indices representing the frequency, timing and rate of change of flow conditions (*e.g.* F_L1 and F_H1, the numbers of low and high flow pulses; T_L1, the timing of the minimum flow; R_A3 the number of hydrograph reversals). However, it is particularly so for indices of the duration of high flow conditions (*e.g.* D_H1 to D_H6, the maximum and multi-day maxima of streamflow events). Thus the findings in this Study highlight the sensitivity of hydrological indices which describe relatively fine time steps, across different hydrogeographical regions.

The "similarly", high CD values of standardised annual values of the indices of the duration of high flow conditions for streams at both the arid and the more moderate ends of the environmental gradient (*c.f.* Figure 5.15) indicate that there is high inter-annual variability in the ecological structure of both these streamflow types (*i.e.* Extreme Seasonal and Sustained Baseflow). Strong, flood driven variability has been cited throughout the literature as being highly relevant for many ecological processes, including landscape organisation and nutrient cycling, with recognised impacts on habitat availability, community composition and distribution as well as whole river functioning (Clausen and Biggs, 2000; Naiman *et al.*, 2002; Bunn and Arthington, 2002). Awareness of the extent of variability required to maintain high flow disturbance in either of these streamflow types could be useful in environmental flow assessments.

The similarity of the CD values of the variability in the seasonal (90-day) maximum flow, D_H12 (*c.f.* Figure 5.32), the predictability of streamflows, T_A1 (*c.f.* Figure 5.33) and the constancy of streamflows, T_A2 (*c.f.* Figure 5.33) among the streams that are located in either high rainfall or arid conditions confirms the usefulness of these indices for assessments of these very different streamflow types.

However, the indices T_A1 and T_A2 (the predictability and constancy of streamflows) do not represent high variability *per se*, but rather the *temporal invariance* of the streamflow regime. Moreover, while other studies have found that it takes "more than 20 years" for T_A1 to stabilise (Gan *et al.*, 1991), T_A2 reaches its long-term value much sooner (Clausen and Biggs, 2000). Nonetheless, the relative consistency of the CDs of these indices across the diversity of the streamflow types examined in this Study highlights these indices as being of high value for any desktop assessments of the environmental flow requirements of South Africa's river ecosystems. Other CDs of indices which are consistent across the diversity of streamflow regimes, include M_L3, the Alt-BFI based on the Desktop Reserve model index of short-term variability, and T_L1 as well as T_H1, the IHA indices of the Julian dates of the minimum and maximum flows. Together, these findings augment the results of the PCA conducted to find those indices which explain the major sources of variation within the entire dataset of 74 hydrological indices examined in this Study, where they are well represented in Tables 5.15 and 5.16.

Preferred Record Length: Broad Hydrogeographical Scale

The analysis of the *preferred record length* for reliable assessments of the hydrological indices necessitated that the dataset of 83 DWAF gauging stations be reduced to 43 DWAF gauging stations, in order to accommodate a longer record length for analysis, albeit only a slightly longer record (*i.e.* record length increased from 36 years to 42 years). This procedure represented a marked difference in sample size, not only for the reduced set of All-43 Streams, but more pertinently for the reduced number of stream sites within each of the six distinct streamflow groups. This situation again highlights a major difficulty that researchers face, particularly in developing countries, when wishing to analyse an adequate number of good quality and sufficiently long records. This author acknowledges that some of the groups have very small samples (*e.g.* the Extreme Seasonal group was reduced to only four sites for each of the indices) and consequently, it could be argued that any assessment of the dispersion from the average is inconsequential, even although non-parametric statistics were applied in this Study. Nonetheless, since the results of the analysis were assessed quantitatively, rather than by statistical tests, the size of the samples was considered to be only of marginal concern, in light of the usefulness of the results.

At a broad hydrogeographical scale, there is little difference among the three record lengths analysed, for most of the hydrological indices. However, the indices of central tendency generally stabilise over shorter record lengths than the indices of variability and of predictability. Thus, where records are relatively short, greater confidence can be placed in the streamflows required for average or "maintenance" years than for years where hydrological conditions are required to "stress" ecosystems. This feature of "highly variable" river systems is to be expected, even at this broad spatial scale.

If only short records (*i.e.* records of, say, 20 years) are available for analysis, stakeholders could make confident assessments of D_H5, the seasonal (90-day) maximum flows for average years (*c.f.* Figure 5.18), but they would need longer records for assessments of the variability associated with this flow condition (D_H11; Figure 5.35). Relatively short records would also be adequate for assessments of D_H6, the number of high pulses (*i.e.* freshes) as well as timing of the annual flood pulse (T_H1). As described in other Chapters of this thesis, these streamflow characteristics form critical elements of environmental flow assessments of aquatic ecosystems, including the South African BBM and DRIFT methods. Moreover, these characteristics of high flow disturbance in streamflow patterns generate highly desirable ecosystem goods and services for the environment, society and people.

Longer records (*i.e.* 36 years) would result in greater confidence in assessments of M_A1 to M_A12 (average monthly flows), R_A1 and R_A2 (the rise and fall rates of the river levels) R_A4 and R_A6 the variability in the rise rate and number of hydrograph reversals), T_H3 (the seasonal predictability of flooding, and T_A1 as well as T_A2 (the predictability and constancy of the streamflows). However, at this broad spatial scale, many of the indices examined require more than 36 years of record for confidence in their assessments.

Preferred Record Length among the Distinct Groups: Finer Hydrogeographical Scale

The analysis of the sensitivity of the hydrological indices of high information, for each of the distinct streamflow groups, indicated that stakeholders would, again, be able to express greatest confidence in assessments of the indices of central tendency of the different streamflow characteristics. If only relatively short records were available, reliable assessment could be made for most of the indices of high information of streams in higher rainfall regions. For these streams, 36 years of record is adequate for at least one index representing each of four (magnitude, duration, timing, and rate of change) out of the five characteristics of the streamflow regime (which additionally includes frequency) described by Richter *et al.* (1996). Consequently, stakeholders could build a fairly objective representation of the *essential* environmental flow requirements of these streamflow types, using 36 years of record.

However, the sensitivity of the indices to record length performed for this Study indicates that many important indices require more than 36 years to stabilise. This could pose limitations to using hydrological records as a "long-term" ecological resource for rivers in South Africa, particularly for ephemeral or intermittent streams. The best way to overcome this would be to simulate longer records of "natural" streamflows using validated and verified hydrological models. However, modelling streamflow patterns for ephemeral or intermittent river systems, where the processes of hydrological partitioning and linkages among hill-slope, groundwater table and river channel are complex, requires sophisticated models with a large number of input parameters to address the many different process representations. There are also shortcomings in the spatial availability of equally long records of observed data for rainfall events (Lynch, 2004). These shortcomings are routinely overcome with rainfall record infilling, a process which can introduce additional shortcomings. In addition, this Study has highlighted the sensitivity of the indices which describe *relatively fine time steps* to record length. There is therefore a need to link the output of hydrological models to the temporal resolution of the indices of high information described in this Study.

General Observations

Notwithstanding the limitations of the length of record described above, to date, this Study represents the most comprehensive analysis of statistical indices of the long term records of relatively natural streamflows for the diversity of South Africa's rivers. This Study has emphasised the importance of maintaining inter-annual variability in streamflow regimes. For example, the PCA showed that, across the diversity of the combined set of streams, the relationship between the *indices of dispersion* (*i.e.* inter-annual variability) in the number of high flow pulses and their durations is stronger than the relationship between the indices of central tendency (*i.e.* inter-annual average) of the number and duration of these disturbances. This has implications for water management strategies, including reservoir release operating rules.

Poff (1996) concluded that from his study of ten hydrological indices of stream sites in the USA that the Coefficient of Variation may be an adequate single descriptor of flow variability in streams. Olden and Poff (2003) concluded that the IHA method adequately represented the majority of variation explained by the entire population of the 171 indices they examined, across the same streamflow sites in the USA. While the majority of the 74 indices examined in this Study comprised the IHA indices, the indices related to the Desktop Reserve model baseflow and overall variability indices were consistently dominant indices of hydrological variability, across the diversity of streamflow regimes types in South Africa. However, it could be beneficial to include some of the high information IHA indices in future desktop studies of environmental flow assessments. Across a broad spatial scale, indices describing the inter-annual variability in seasonal high flows, streamflow predictability and seasonal predictability would be useful. For each "streamflow regime type" scale, different indices could be selected from Table 5.15.

This Study also highlighted that some indices are probably not worth pursuing as contenders for inclusion in a "suite of desktop indices". For example, the seasonality of non-flooding (D_H13) did not explain a high proportion of the variation for any of the streamflow types (*i.e.* this index does not appear in Table 5.15). This is not to say that D_H13 is not a good descriptor of the seasonal predictability of the streamflow regimes found in South Africa, but rather that such an index represents "a portion of the total variation that is not represented by the other hydrological indices" (Olden and Poff, 2003). Moreover, this index is highly sensitive to record length at different spatial scales and may have limited use in desktop studies.

It is acknowledged that had longer records been suitable for use in this analysis, different values may have been derived for any of the 74 hydrological indices examined. Consequently, the PCA could have identified different loadings on each of the significant principal component axes, and different indices may have emerged as being of high information of the hydrological variability associated with the diversity of stream sites used. In addition, examining different stream sites could also have led to different results. However, an important feature of this Study is that the best available information was applied and that the results are repeatable.

6 RESULTS OF RE-ANALYSIS OF A REVISED AND REDUCED WORKING DATABASE

6.1 Introduction

Subsequent to the completion of the research and writing comprising Sections 1 to 5 of this Chapter, an important finding emerged regarding the database of the DWAF gauging stations assumed to represent "natural flows" and the authenticity of the naturalness of those flows (*c.f.* Section 3 of this Chapter). This concern, regarding the validity of a catalogue of sites which represent "natural flow regimes" across South Africa, came to light as a result of reviewing a Report of the Reserve Determination for the Thukela Catchment prepared for the DWAF by IWR Environmental (IWR Environmental, 2003). The various issues are detailed here for the sake of transparency.

6.2 Assumptions Regarding the Selection of the DWAF Gauging Stations Recording Natural Streamflows

The 1994-Study (Joubert and Hurly, 1994) reviewed and applied data from 352 gauging weirs "that according to DWAF regional technicians and subsequent checking of gauging station positions on maps supplied by the Hydrological Research Institute of DWAF, were situated upstream of all major impoundments or abstractions and had a minimum record span of 20 years". While the authors of the 1994-Study stated that "the process of selection was not exhaustive with regards to checking upstream alterations to flow patterns, [they considered that] subsequent non-homogeneity tests would indicate which gauges were recording flow that was changing with time" (*c.f.* Section 3.3 of this Chapter).

The testing for non-homogeneity of the daily streamflow data recorded at each of the 352 DWAF stations performed by the 1994-Study resulted in certain gauging stations being excluded from the 1994-Study. Exclusion of a station was based on the identification of breaks in double mass plots, created from pairs of data from rainfall and streamflow gauging stations, plotting cumulative monthly flow against cumulative monthly rainfall for each station. The exclusion of a station was operated on the premise that "where breaks in the plots occurred which were obvious by visual assessment, the flow gauging station concerned was either excluded from the analysis, or the data after the break were excluded, where this was possible" (Joubert and Hurly, 1994). This form of analysis was reviewed by this author in Section 3.3 of this Chapter and alternative methods were employed in the Study in this Chapter to ascertain the stationarity and non-homogeneity of streamflows recorded at the gauging stations comprising the "working database" (*c.f.* Section 3.5 of this Chapter). It may be that, following their tests applied for non-homogeneity, only a partial record was applied by the 1994-Study Report for some stations. However this is not clear from the information provided by the 1994-Study Report, since years with missing data were also removed prior to their tests for non-homogeneity. Consequently, it was assumed that the statement in the 1994-Study that the "stations remaining after this exercise [their test for non-homogeneity] were felt to be recording reasonably natural flow" was a valid starting point for the Study in this Chapter. Moreover, it was assumed for the purposes of the Study in this Chapter that the stations comprising the "best200" gauging weirs (*c.f.* Section 3.3 of this Chapter) obtained from A.R. Joubert in 1995 by the BEEH represented stations which were indeed "recording reasonably natural flow". It was on the basis of this understanding that the "best200" gauging weirs were further investigated (*c.f.* Section 3.3.1 of this Chapter) for suitability and updating for the research and writing comprising Sections 3, 4 and 5 of this Chapter.

6.3 Misgivings Concerning the Database of the "best200" Gauging Weirs

A number of anomalies regarding the "working database" of Section 3.5 of this Chapter became apparent when reviewing a Report of the Reserve Determination for the Thukela Catchment prepared for the DWAF by IWR Environmental (IWR Environmental, 2003) for information relating to the methods employed in an up-to-date, comprehensive EFA and for the writing of other Chapters in this thesis. The Thukela Catchment is a high profile designated Water Management Area in KwaZulu-Natal, since not only are both its headwater areas and its estuary recognised as World Heritage sites, but the catchment is an important provider of water for the economic heartland of the country via inter-basin transfer. Moreover, the Thukela Catchment has the potential to uplift the economic and social standing of the people who live within its boundaries. As a consequence, many ongoing projects have been instigated to ensure the sustainable management of the Thukela Catchment (*e.g.* the DWAF's TWP (Thukela Water Project, http://www.dwaf.gov.za) and the International Hydrology Programme's HELP (Hydrology for the Environment, Life and Policy) Project, http://www.unesco.org/water/ihp/help), including the assessment of an

Ecological Reserve and the Instream Flow Requirements of its river systems (IWR Environmental, 2003). Consequently, there is an abundance of literature available against which to check the relevant findings of the results of the analysis presented in earlier in this Chapter.

The first signs of irregularity regarding the "working database" developed from the "best200" gauging weirs arose from attempts to compare the "natural" flow regimes of the Thukela gauging stations in the "working database" with the assessments of natural and present-day hydrological conditions for different Resource Units (*c.f.* Chapter 4, Section 2.5) in the Thukela Catchment, performed by the hydrological specialist for the Thukela Water Project Decision Support Phase Reserve Determination Module (IWR Environmental, 2003). Based on the understanding that the Thukela gauging stations in the "working database" were recording natural streamflows, as described above and in Section 3 of this Chapter, it should be expected that their streamflow regimes should equate with the present-day assessments of hydrological conditions performed for the Thukela Water Project Decision Support Phase Reserve Determination Module (TWP Reserve Determination). This transpired to be erroneous for a number of the Thukela stations in the "working database". Table 5.20 indicates the extent of the discrepancy encountered. The terminology of the South African Reserve is described in detail in Chapter 4 and summarised in the Glossary of Terms of this thesis. Nonetheless, the terms *Resource Unit* and *Ecostatus* are redefined here in the context of Table 5.20.

A *Resource Unit* (RU) represents an area of relative homogeneity in physiographic, geographic, hydrological, and biological features. In this instance, the hydrological *ecostatus* refers to the present-day, hydrological state, or condition, of a Resource Unit. The method of assessment of the present-day hydrological condition, applied in the TWP Reserve Determination, was based on the comparison of natural and present-day 1-month annual duration curves (*i.e.* flow duration curves constructed from monthly volumes for all months of the year) as described by Hughes and O'Keeffe in the TWP Reserve Determination (IWR Environmental, 2003). In essence the method comprises:

(a) the calculation of the positive relative difference between the two duration curves at various percentage points,

(b) different weightings to account for the prevalence of low flows as well as for reversals, or major changes, to the seasonality of the baseflows and

(c) a final weighted score.

Table 5.20 Comparing the "natural" flow regimes of the Thukela Catchment gauging stations in the "working database" with assessments of reference, or natural, and present-day hydrological conditions for different Resource Units performed for the TWP Reserve Determination (*c.f.* IWR Environmental, 2003)

DWAF Station	**Resource Unit** (*indicates the closest RU)	**Hydrological Ecostatus: Present-day (1995) hydrological conditions** (**indicates monthly time series of present-day streamflows not available to the TWP)		**Confidence in Reference Hydrological Conditions**
V1H001	D	E**	No TWP comments	4
V1H009	N*	A	TWP Study reports that: • Present-day MAR is 97.3 % of natural MAR • Close to natural flows	5
V1H010	M	F	TWP Study reports that: • Present-day MAR is 88.9 % of natural MAR • Approximately 20% of the time when there was previously flow, the river will not now flow. This is heavily weighted in the calculation of the Present Ecological State.	4
V3H002	U*	B**	TWP Study reports for RU U that: • Present-day MAR is 86 % of natural MAR	4
V3H007	U*	B**	As for RU U	4
V3H009	U*	B**	As for RU U	4
V6H003	Q*	E**	TWP Study reports for RU Q: • Present-day MAR is 80.77 % of natural MAR • There is up to 25% zero flow in this reach at present-day modelled flow. However, the DWAF monthly data does not have any zeros in it.	4
V6H004	Q	E**	• As for RU Q	4
V7H012	N	A	TWP Study reports that: • Present-day MAR is 97.3 % of natural MAR • Close to natural flows.	5

Monthly time series of natural and present-day (1995 conditions) for the hydrological years 1925 to 1994 were applied in the assessment of TWP Reserve Determination. However, of the 22 RUs identified by the Reserve Determination Team for the Thukela Catchment, present-day flows were only available for 16 of the RUs. Thus, it is assumed, for the purposes of this Study, that for the remaining six RUs, the present-day hydrological condition reported in the TWP Reserve Determination was assessed by extrapolating the results representing the hydrological condition of the upstream RU. Linking the Thukela gauging stations in the working database with the TWP Reserve Determination RUs presented additional problems, since the RUs have been identified for the main tributaries of the Thukela river system. Consequently, since several of the Thukela gauging stations in the "working database" are located on first order tributaries and it was not possible to match each gauging station in Table 5.20 with an RU. For the purposes of this Section, the hydrological conditions for such sites were aligned with the closest RU. Where this occurred the RU is flagged with an asterisk in Table 5.20. Where monthly time series of present-day streamflows were not available to the TWP Reserve Determination for evaluation of the present-day hydrological conditions as described above, the hydrological ecostatus is annotated with two asterisks in Table 5.20.

In Table 5.20, the *confidence* associated with the hydrological assessment refers to the confidence evaluation of the hydrological specialist of the TWP Reserve Determination and indicates the degree of difficulty attached to defining reference, or natural, hydrological conditions, as a result of the limitations or the absence of observed streamflow data. A confidence score is assigned, ranging from 0 to 5, with 0 reflecting "no confidence" and 5 reflecting "high confidence" (IWR Environmental, 2003). As indicated in Table 5.20, the present-day hydrological conditions of the RUs closest to the Thukela gauges applied in the "working database" were assigned an hydrological ecostatus ranging from A to F by the TWP Reserve Determination. Although the confidence levels associated with the hydrological assessment were low, ranging from 4 to 5, it is clear that RUs with an hydrological ecostatus lower than B do not experience streamflows which are "reasonably natural", as was assumed for the streamflows recorded by the DWAF gauges comprising the "working database". It is important to note that the purpose of this Section is not to provide a critique of the methods or the results of the TWP Reserve Determination, but rather to highlight the reasons for reviewing the inclusion of some of the stations in the "working database" used in Sections 3 and 4 of this Chapter.

The information collated in Table 5.20 led to the investigation of the Thukela gauging stations, and subsequently the entire "working database", being reviewed for validation of the basic assumptions of their representation of natural flow regimes. This was performed by checking the information in Volumes of the Surface Water Resources of South Africa 1990, WR90, (Midgley *et al.*, 1994) for the relevant Quaternary Catchments upstream of and within which each of the gauging stations comprising the "working database" is situated. The following simple qualitative rules, shown in Table 5.21, were applied to the evaluation of the water use and land cover for the area upstream of each DWAF gauging station in the "working database". It was considered that where water useage was either negligible or minimal and land cover status was either negligible or largely unimpacted, in accordance with the qualitative rules in Table 5.21, the station could be included in a "***revised and reduced working database***" of stations recording "reasonably natural flows".

Table 5.21 Classification of the water useage and land cover status upstream of each DWAF gauging station in the "working database", assessed from information contained in the Volumes of the Surface Water Resources of South Africa 1990, WR90, collated by Midgley *et al.* (1994)

Water Useage		Land Status	
Negligible	• No water useage indicated on relevant WR90 Map • No more than 2 small dams	***Negligible***	• No artificial land cover indicated on relevant WR90 Map • Less than 10% afforestation • Less than 3% irrigation
Minimal	• Presence of a few small dams • No inter basin transfer • No major abstractions • No return flows	***Largely Unimpacted***	• 10% to 25% afforestation • 3% to 7% irrigation
Moderately Utilised	• Return flows • Presence of many small dams • Site of small dam • Downstream of sizeable dam	***Moderately Impacted***	• Downstream of urban area • 25% to 50% afforestation • More than 7% irrigation
Heavily Utilised	• Major run of river abstraction • Inter-basin transfer • Site of major dam • Downstream of major dam	***Heavily Impacted***	• More than 50% afforestation

6.4 Inclusion of Stations Recording Impacted Streamflow Regimes in the "working database"

The results of the investigation are provided in Table 5.22 which shows the extent of the problem, especially for stations situated in Quaternary Catchments V, U and X located in the higher rainfall regions of the country, where perennial streams are more heavily utilised by people and society. The results indicate that it could be contended that as many as 40% of the stations in the "working database" used in the Study in Sections 3 and 4 of this

Chapter were not recording "reasonably natural flows" as had been initially assumed. Consequently, the number of stations in the "working database" which could be considered to be recording "reasonably natural flows" was revised, and reduced, from 83 to 48 (hereafter referred to as the "best48" stations). The "best48" stations are shown in bold in Table 5.22.

Table 5.22 Upstream water useage and land cover status, based on examination of the information contained in Volumes of the Surface Water Resources of South Africa 1990 (Midgley *et al.*, 1994), for the relevant Quaternary Catchments upstream of each of the gauging stations comprising the "working database". Bold font denotes the "best48" stations.

DWAF Gauging Station	**Site of DWAF Station (Quaternary Catchment)**	**Upstream Water Useage**	**Upstream Land Cover Status**
A2H029	A23A	Negligible	Negligible
A2H032	A22C	Negligible	Negligible
A4H002	A42C	Minimal	Negligible
A4H005	A42F	Minimal	Negligible
A4H008	A42D	Negligible	Negligible
A5H004	A50B	Minimal	Negligible
A9H003	A91G	Negligible	Negligible
A9H004	A92A	Minimal	Largely Unimpacted
B1H002	B11H	Minimal	Negligible
B1H004	B11K	Moderately Utilised	Moderately Impacted
B4H005	B42F	Negligible	Negligible
B6H001	B60B	Negligible	Moderately Impacted
B6H003	B60C	Negligible	Largely Unimpacted
C5H007	C52F	Negligible	Negligible
C7H003	C70G	Negligible	Negligible
C8H003	C82B	Negligible	Negligible
D5H003	D52C	Minimal	Negligible
G1H008	G10E	Minimal	Largely Unimpacted
G1H009	G10E	Moderately Utilised	Largely Unimpacted
G1H010	G10E	Moderately Utilised	Largely Unimpacted
G1H011	G10E	Moderately Utilised	Largely Unimpacted
G2H012	G21C	Minimal	Largely Unimpacted
G4H006	G40K	Heavily Utilised	Negligible
G5H008	G50G	Negligible	Negligible
H1H007	H10E	Negligible	Negligible
H1H013	H10C	Moderately Utilised	Negligible
H3H005	H30C	Heavily Utilised	Negligible
H7H004	H70C	Minimal	Negligible
J4H003	J40C	Moderately Utilised	Largely Unimpacted
K3H002	K30B	Negligible	Largely Unimpacted
K3H004	K30B	Negligible	Largely Unimpacted

K4H002	K40C	Negligible	Largely Unimpacted
K4H003	K40A	Negligible	Heavily Impacted
K5H002	K50A	Negligible	Moderately Impacted
K6H001	K60A	Negligible	Negligible
K7H001	K70B	Negligible	Largely Unimpacted
K8H001	K80C	Negligible	Largely Unimpacted
K8H002	K80C	Negligible	Largely Unimpacted
L8H001	L82D	Moderately Utilised	Largely Unimpacted
R1H014	R10F	Negligible	Moderately Impacted
R2H001	R20A	Negligible	Largely Unimpacted
R2H006	R20C	Negligible	Negligible
R2H008	R20A	Heavily Utilised	Negligible
S3H006	S31E	Minimal	Negligible
S6H003	S60C	Negligible	Largely Unimpacted
T3H004	T32C	Minimal	Negligible
T3H008	T31G	Minimal	Negligible
T3H009	T35C	Negligible	Negligible
T4H001	T40C	Negligible	Largely Unimpacted
T5H003	T51D	Moderately Utilised	Negligible
T5H004	T51B	Minimal	Negligible
U1H005	U10E	Minimal	Negligible
U2H006	U20D	Moderately Utilised	Moderately Impacted
U2H007	U20B	Heavily Utilised	Largely Unimpacted
U2H011	U20H	Negligible	Negligible
U2H012	U20F	Minimal	Heavily Impacted
U2H013	U20C	Minimal	Largely Unimpacted
U4H002	U40A	Minimal	Moderately Impacted
U7H007	U70A	Minimal	Heavily Impacted
V1H001	V14A	Heavily Utilised	Moderately Impacted
V1H009	V14C	Negligible	Negligible
V1H010	V13C	Moderately Utilised	Largely Unimpacted
V3H002	V31C	Heavily Utilised	Negligible
V3H007	V31H	Negligible	Negligible
V3H009	V31F	Negligible	Negligible
V6H003	V60D	Minimal	Negligible
V6H004	V60B	Heavily Utilised	Largely Unimpacted
V7H012	V70D	Negligible	Negligible
W1H004	W13A	Negligible	Heavily Impacted
W4H004	W41D	Minimal	Largely Unimpacted
W5H006	W51D	Minimal	Largely Unimpacted
X2H005	X22F	Moderately Utilised	Heavily Impacted
X2H008	X23E	Negligible	Heavily Impacted
X2H010	X23A	Negligible	Heavily Impacted
X2H012	X21F	Negligible	Negligible
X2H013	X21E	Moderately Utilised	Largely Unimpacted
X2H014	X22A	Negligible	Heavily Impacted
X2H015	X21K	Moderately Utilised	Largely Unimpacted
X2H022	X23H	Moderately Utilised	Moderately Impacted
X2H024	X23C	Negligible	Heavily Impacted
X3H001	X31A	Negligible	Heavily Impacted
X3H002	X31A	Negligible	Largely Unimpacted
X3H003	X31C	Negligible	Heavily Impacted

The following conditions were found to have been masked by the basic assumption that the stations comprising the "best200" gauging weirs were indeed recording natural flows:

(a) While the "best200" database was believed to comprise stations situated upstream of all major impoundments or abstractions, this is erroneous for a number of weir locations. Several gauging stations are situated downstream from major impoundments which were in existence, in some instances for several decades before the 1994-Study was performed (*e.g.* the Thukela-Vaal Water Transfer Scheme comprising Woodstock Dam and the Driel Barrage on the Thukela River was completed in 1974 and is upstream of gauging station V1H001). It may be that, following the tests applied for non-homogeneity, only a partial record was applied by the 1994-Study for such stations. However, this is not clear from the information provided in the 1994-Study Report (*c.f.* Section 6.2 of this Chapter) and the reader is informed that the "stations remaining after this exercise were felt to be recording reasonably natural flow" (Joubert and Hurley, 1994). Notwithstanding the 1994-Study tests of non-homogeneity, stations situated downstream of major impoundments should not have been included in the initial selection of gauging weirs provided by the DWAF.

(b) The objective of the tests for non-homogeneity applied in the 1994-Study was to highlight "non-homogeneity in daily flow records, where changing upstream patterns of abstraction and land use change the runoff pattern recorded by the flow gauging station over the gauged time period". However, many of the gauging stations in the "best200" database are situated downstream of human altered landscapes which had been changed from their natural state before the 1994-Study was performed. Again, it may be that in such instances, following the tests applied for non-homogeneity, only a partial record was applied by the 1994-Study for such stations.

It could be argued that the tests for non-homogeneity devised for Section 3.4 of this Chapter should have highlighted the problems identified in points (a) and (b). To a certain extent this is reflected by the low scores of some, but not all, of the affected stations (*c.f.* Table 5.4). However, as a result of the erroneous assumptions regarding the "best200" gauging stations, the tests were performed for time periods that included human altered flow regimes. In several instances the entire time period may have compromised by such alterations. In some instances there may have been only small changes in the upstream patterns of abstraction and land use over the time period. These factors could have further disguised the anomalies regarding the performance of tests for non-homogeneity devised for the Study in Section 3.4 of this Chapter.

6.5 Extent of the Problem

On the basis of the findings presented in Table 5.22, a PCA was extracted from the 74 x 74 correlation matrix representing a "*revised and reduced working database*" (*i.e.* the 74 hydrological indices described in Section 3.3.2. of this Chapter for the "best48" stations shown in bold in Table 5.22).

A scatter plot of object scores (gauging stations) for the first two principal components from the 74 x 74 correlation matrix (Figure 5.44) was examined to re-assess the clustering of the 48 DWAF sites. The positions of the sites shown in Figure 5.44 were reviewed with the groups formed in Section 4.3.2 of this Chapter. As in Section 4.3.2, the letters shown in Figure 5.44 refer to the streamflow group used in the classification of Method One of the 1994-Study by Joubert and Hurly (eight groups, described as A to H). Numbers annexed to the letters distinguish stations within the "working database" of Section 3.5 of this Chapter (*c.f.* Table 5A1 (Appendix 5A) and Table 5.4 for details of each station). The six groups, I to VI described in Section 4.3.2 of this Chapter are distinguished by crosses of different colours; red, orange, green, blue, pink and purple crosses for Groups I to VI respectively.

The results of the analysis indicate that, even after the removal of the stations which were not recording reasonably natural flows, the general environmental gradient of "extreme seasonal", or "harsh arid", conditions represented by ephemeral river systems (Group I in Section 4.3.2 of this Chapter), through "mixed quasi-perennial-seasonal and perennial" regimes (Groups II, III and IV in Section 4.3.2 of this Chapter) to the more moderate regimes found in perennial systems (Group V and VI in Section 4.3.2 of this Chapter) is, by and large, still valid for the same groups of gauging stations (Figure 5.44) as was shown in Section 4.3.2 of this Chapter (*c.f.* Figure 5.9). However, this trend is least clear for the stream sites representing the "Sustained Baseflow" regimes of Group VI (purple crosses), particularly for stations C1 and B7. However, comparison with Figure 5.9 of this Chapter

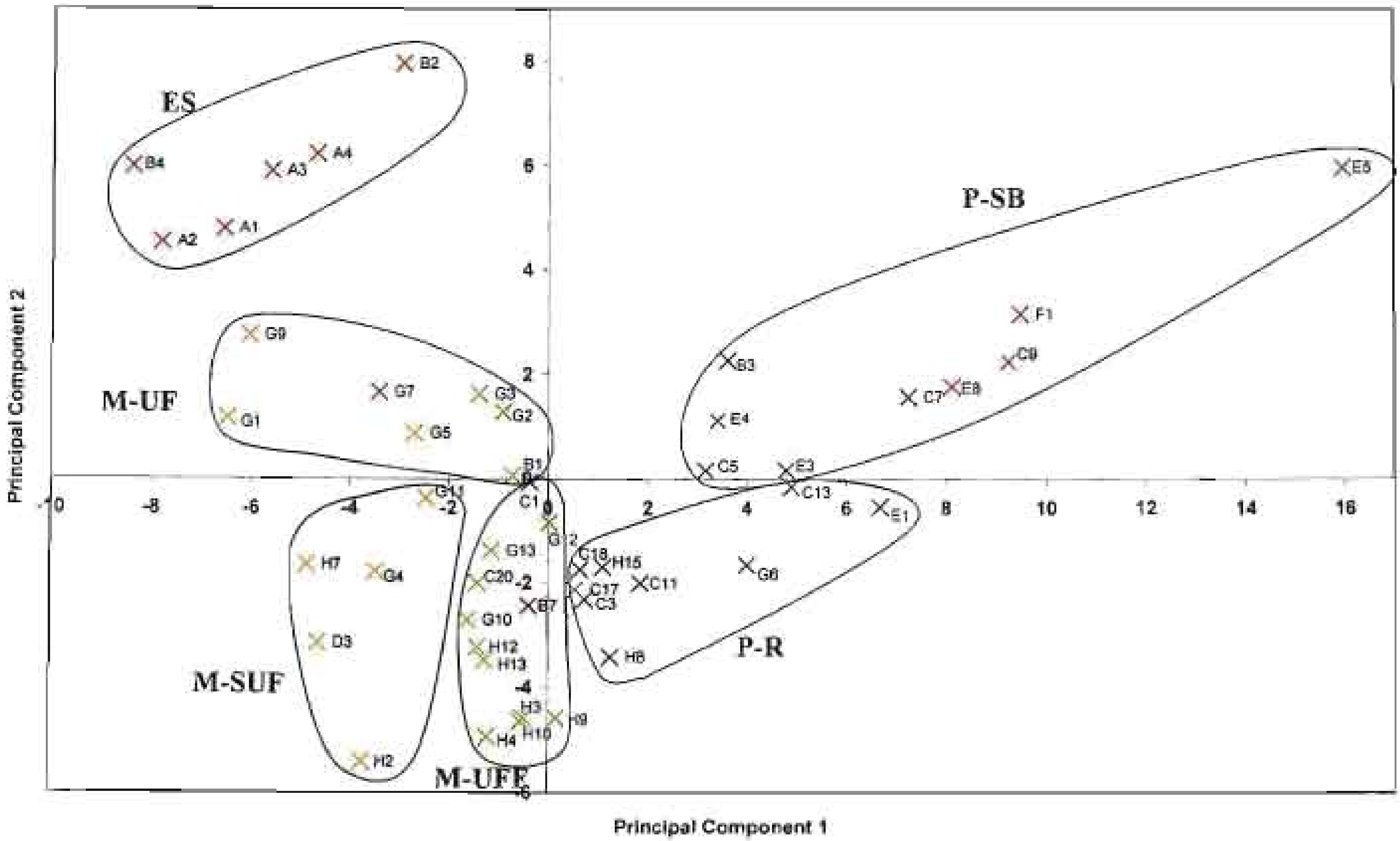

Figure 5.44 Bivariate plot of scores for the first two principal components from a PCA of 48 stream sites based on the correlation matrix of 74indices. Letters A to H refer to the classification of Method One of the 1994-Study by Joubert and Hurley (1994). Numbers annexed distinguish the sites within the 1994-Study grouping as well as the sites in the "working database" of Section 3.5 of Chapter 5 of this thesis. Coloured crosses denote the revised grouping used in Section 4.3.2 of Chapter 5; red, orange, green, blue, pink and purple denote Groups I, II, III, IV, V and VI respectively. Abbreviations ES, M-UF, M-SUF, M-UFF, P-R and P-SB (see text for details) denote the grouping of the "revised and reduced working database".

indicates that these sites were both "outliers" in the original Group VI and were included on the basis of their relatively high values of M_L3, the Alt-BFI (*c.f.* this Chapter, Section 4.3.1 for discussion). Moreover, Group VI experiences the greatest reduction (64%) in the number of stations as a result of the removal of the stations which were not recording reasonably natural flows (*i.e.* from 11 to four stations), whereas Groups IV (blue crosses) and V (pink crosses) experience more modest reductions (55% and 50% respectively), and Groups I (red crosses), II (orange crosses) and III (green crosses), experience the least reductions (30%, 33% and 36% reductions respectively).

However, the most serious implication of reducing the dataset of DWAF gauging stations which are considered to be recording relatively natural flow, from 83 to 48 sites, is that analysing the flow regime at only 48 sites is arguably not meaningful for the broad diversity of flow regimes found across South Africa. This is particularly pertinent in light of the reduced representation of natural conditions for the higher rainfall catchments in the region, as a result of the exclusion of many of the stations from the "working database". Figure 5.45 shows the distribution of the 48 sites.

It was highlighted in the discussion in Section 5 of this Chapter that *examining different stream sites could have led to different results, since the PCA could have identified different loadings on each of the significant principal components, and different indices may have emerged as being of high information of the hydrological variability associated with the diversity of stream sites used.* In addition, it can be expected that regrouping the reduced number of sites to form distinctions among the streamflow regimes would lead to further differences among the loadings of the hydrological indices on each of the significant principal components and, again, different indices could emerge as being of high information of the hydrological variability associated with any particular stream type. As discussed in Section 5 of this Chapter, this finding is an indication of the sensitivity of sample size when applying multivariate analysis of environmental datasets.

Notwithstanding these reservations, a preliminary investigation was initiated to assess the extent of any mismatch between the subset(s) of hydrological indices explaining a dominant proportion of statistical variation in the entire set of indices for different streamflow types when using the "revised and reduced working database" compared with the application of the "working database".

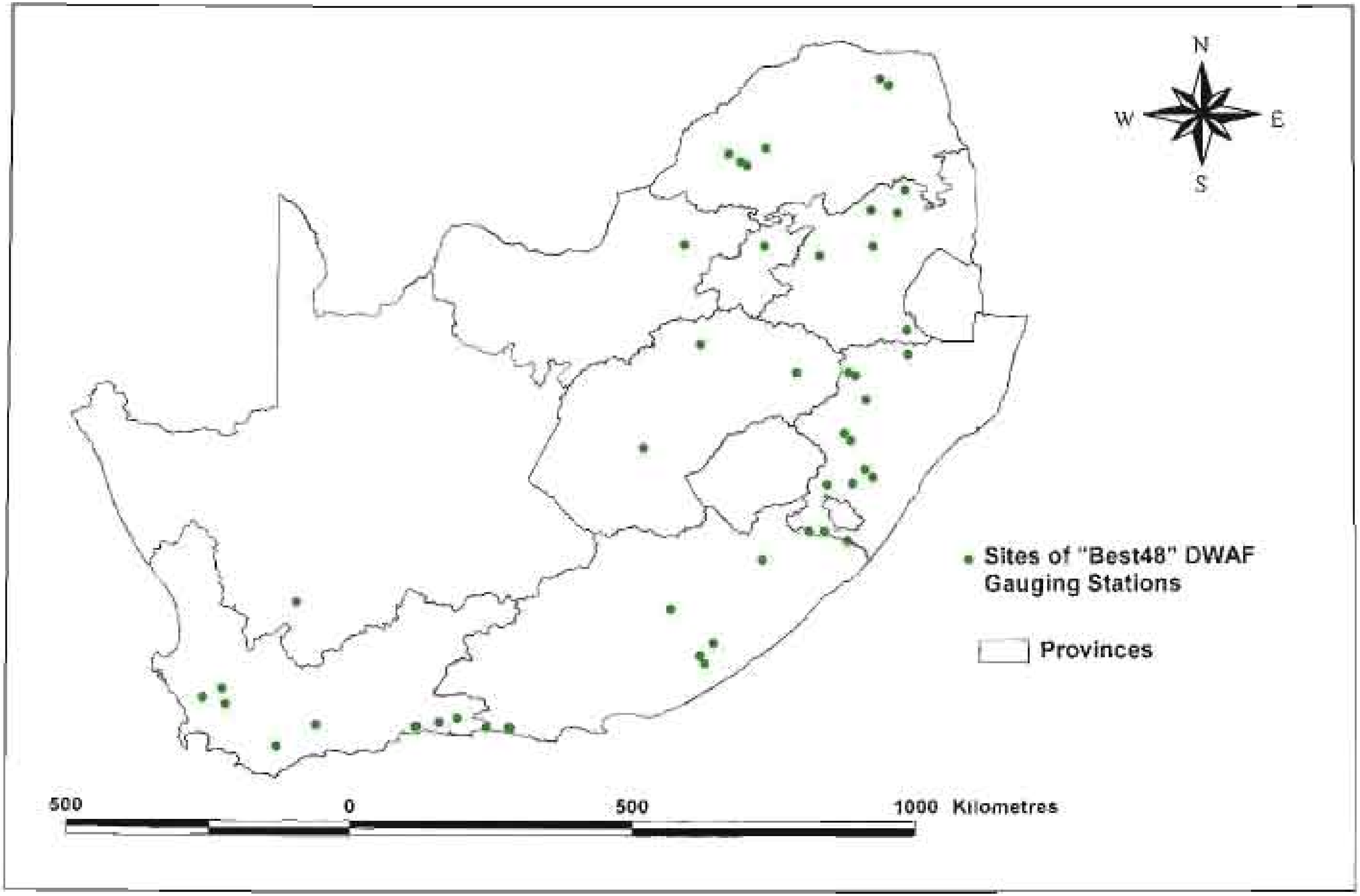

Figure 5.45 Distribution of the 48 stream sites across South Africa

Initial investigation of the Streamflow Groups formed by the "best48" stations

Following the methods described in Section 4.4 of this Chapter, the "best48" stations were re-grouped into six groups (shown with streamflow type abbreviations, e.g. ES for Extreme Seasonal; M-SUF for Short, Unseasonal Floods; M-UFF for Unpredictable Flow and Floods; M-UF for Unpredictable Flow; P-R for Runoff and P-SB for Sustained Baseflow) using the object (stream site) scores on the PCA, as shown in Figure 5.44. The revised classification of the streamflow types (or groups) based on the "best48" stations, numbers of sites within each group, group medians (large numerals) and CDs (smaller numerals) of selected indices (*c.f.* Table 5.9) of the updated regrouping of DWAF gauging stations are shown in Table 5.23. Shading around the medians indicates a distinguishing index of the Group. Figure 5.46 shows the distribution of the six revised groups of DWAF gauging stations, whereas Table 5.24 shows the station names and revised grouping.

Comparison between Table 5.23 and Table 5.10 shows that, as expected, the distinguishing characteristics of the regrouped stream sites of the "best48" stations do differ, to varying extents, to those of the groups formed for the stream sites in the "working database". While there is no real difference for the distinguishing characteristics of the sites forming the "Extreme Seasonal" group, there is no longer a "mixed" group which can be distinguished by "Seasonally Predictable Flow and Floods". In addition, the distinguishing characteristics used to classify the different streamflow regime types among the two perennial groups of the revised and reduced working database were not so readily separated (*c.f.* Table 5.23) as they were when analysing the 83 stations in the "working database" of Section 3.5 of this Chapter (*c.f.* Table 5.10).

Initial investigation of high information indices describing the stream sites at the "best48" stations

Following the methods described in Section 4.4.1 of this Chapter, the meaningful components of the PCA of the "*revised and reduced working database*", are shown in Table 5.25. The number of statistically significant principal component axes, using the proportions computed by the Broken-stick model (*c.f.* Section 4.4.1 of this Chapter), was similar to the findings for the PCA of the 83 stream sites and ranged from three (*e.g.* the Extreme Seasonal group) to six (*e.g.* the Runoff group). The first six principal components (PC1, PC2, PC3, PC4, PC5 and PC6) explained 32.31%, 14.56%, 11.19%, 7.54%, 6.73% and 5.11% respectively of the variation in the complete set of hydrological indices and

Table 5.23 Revised classification of streamflow types found in South Africa, based on revised and reduced working database of "best48" stations. Group medians (large numerals) and the Coefficients of Dispersion (smaller numerals) of selected hydrological indices of each of the six groups determined from the PCA are given. Shading around the medians indicates a distinguishing index of the Group.

	EXTREME SEASONAL	MIXED			PERENNIAL	
	Group: ES Number: 6	M-UF 7	M-SUF 5	M-UFF 12	P-R 9	P-SB 9
INDICATOR	**Extreme Seasonal**	**Unpredictable Flow**	**Short, Unseasonal Floods**	**Unpredictable Flow and Floods**	**Runoff**	**Sustained Baseflow**
ZERODAY	**281.00** 0.09	32.50 3.06	5.50 2.72	0.00 0.00	0.00 0.00	0.00 0.00
LOWDUR	0.00 0.00	4.47 1.65	7.42 0.64	8.84 0.41	10.38 0.75	**14.50** 0.25
HICOUNT	3.50 0.29	6.00 0.71	**12.00** 0.46	9.75 1.00	10.00 0.30	8.00 0.32
HIGHDUR	13.09 0.62	**12.58** 0.62	**6.56** 0.25	8.11 0.56	7.38 0.52	8.93 0.40
REVERSALS	16.25 0.48	58.5 0.30	74.00 0.13	81.25 0.14	**91.50** 0.19	86.00 0.09
BFI	0.15 0.24	0.23 0.19	0.28 0.38	0.38 0.34	0.38 0.35	**0.38** 0.20
CDB	61.01 0.85	24.75 0.41	15.78 0.09	8.28 0.85	7.31 0.59	5.70 0.88
HFI	4822.50 1.26	76.19 19.55	126 1.80	73.98 0.99	46.22 1.60	30.42 0.45
PRED	0.63 0.23	**0.25** 0.38	0.24 0.25	**0.28** 0.39	0.36 0.14	0.32 0.41
PROP	0.88 0.06	0.64 0.25	0.62 0.11	0.75 0.28	0.58 0.40	0.57 0.28
%FLOODS	0.72 0.83	0.36 0.13	0.28 0.36	**0.32** 0.39	0.34 0.18	0.36 0.17
FLOODFREE	2.5 2.80	6.00 4.25	**0.00** 0.00	4.50 3.33	8.00 2.38	**14.00** 1.65

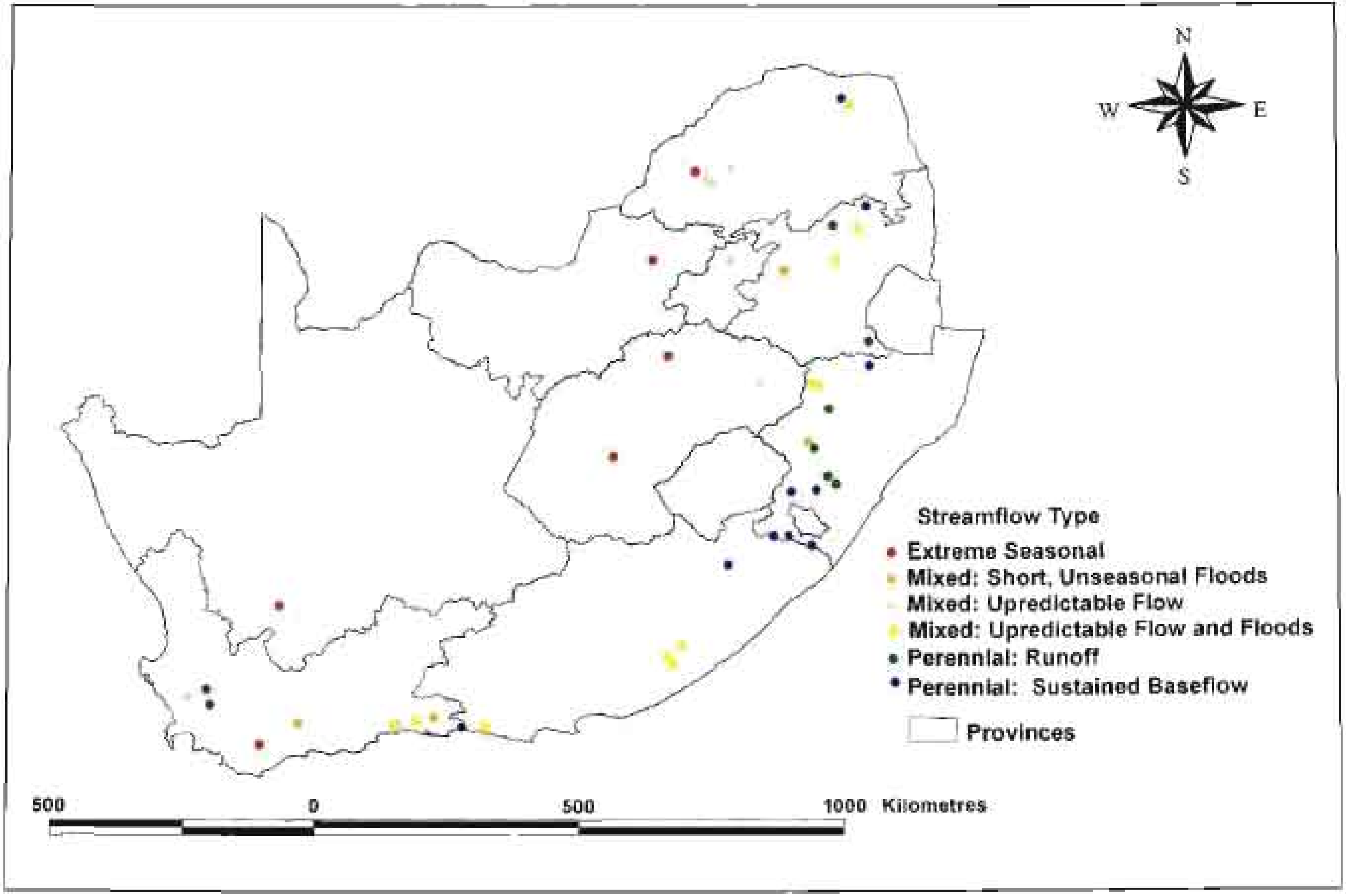

Figure 5.46 The distribution of the six revised groups of the "Best48" DWAF gauging stations

Table 5.24 Groups of the 48best DWAF gauging stations

Gauging Station	1994-Study Group	Working database Group	Revised and reduced Group
A2H029	G1	II	M-UF
A2H032	A1	I	ES
A4H002	B1	III	M-UF
A4H005	B2	I	ES
A4H008	G2	III	M-UF
A5H004	G3	III	M-UF
A9H003	C1	VI	M-UFF
A9H004	B3	VI	P-SB
B1H002	G4	II	S-UF
B4H005	C3	IV	P-R
B6H003	C5	VI	P-SB
C5H007	A2	I	ES
C7H003	A3	I	ES
C8H003	G5	II	M-UF
D5H003	A4	I	ES
G1H008	G6	IV	P-R
G2H012	G7	I	M-UF
G5H008	B4	I	ES
H1H007	E1	IV	P-R
H7H004	D3	II	S-UF
K3H002	H2	II	S-UF
K3H004	H3	III	M-UFF
K4H002	H4	III	M-UFF
K6H001	H7	II	S-UF
K7H001	H8	IV	P-R
K8H001	H9	III	M-UFF
K8H002	H10	III	M-UFF
R2H001	H12	III	M-UFF
R2H006	H13	III	M-UFF
S3H006	G9	II	M-UF
S6H003	G10	III	M-UFF
T3H004	E3	IV	P-SB
T3H008	C7	IV	P-SB
T3H009	E4	IV	P-SB
T4H001	F1	V	P-SB
T5H004	C9	V	P-SB
U1H005	E5	V	P-SB
U2H011	C11	IV	P-R
U2H013	C13	IV	P-R
V1H009	G11	II	S-UF
V3H007	G12	III	M-UFF
V3H009	G13	III	M-UFF
V6H003	H15	IV	P-R
V7H012	C17	III	P-R
W4H004	E8	V	P-SB
W5H006	C18	IV	P-R
X2H012	C20	III	M-UFF
X3H002	B7	VI	M-UFF

together explained 77.44% of the variation for the combined set of stream types, "All-48 Streams". There is, however, some divergence between the results of the two databases when investigating the meaningful components resulting from the PCA conducted for the "*revised and reduced working database*". By excluding a large number of the stations recording perennial flow from the analysis, an increased number of meaningful principal components is required to explain the variation in the indices for both the Runoff and Sustained Baseflow Types, whereas a reduced number of meaningful principal components is required to explain the variation in the indices for the Extreme Seasonal Type.

Table 5.25 Results from the principal component analysis based on the correlation matrix of 74 hydrological indices of 48 stream sites grouped into six streamflow types

Stream flow type	**Principal Component (% variation explained)**						**Total**
	1	2	3	4	5	6	
Extreme Seasonal	51.40	25.87	13.04	-	-	-	90.31
Mixed: Unpredictable Flow (M:UF)	40.31	21.27	15.45	13.71	5.65	-	96.39
Mixed: Short, Unseasonal Floods (M:SUF)	39.79	31.53	18.25	10.44	-	-	100
Mixed: Unpredictable Flow and Floods (M-UFF)	37.74	27.82	14.04	6.15	-	-	85.75
Perennial: Runoff (P: R)	32.00	24.19	16.07	8.69	7.56	5.24	93.75
Perennial: Sustained Baseflow (P:SB)	40.50	21.50	13.73	8.64	6.77	5.01	96.15
All-48 Streams	32.31	14.56	11.19	7.54	6.73	5.11	77.44

Similarly to the findings presented in Section 4.5 of this Chapter, the general patterns of inter-correlation among the indices indicate that many of the hydrological indices are highly inter-correlated. Figure 5.47 shows that, for the combined set of "All-48 streams", the highest correlation is generally, although not exclusively, among indices of central tendency describing certain streamflow characteristics, and again includes indices of the following streamflow conditions:

(a) The magnitudes of average flow conditions (*i.e.* the mean monthly streamflows);

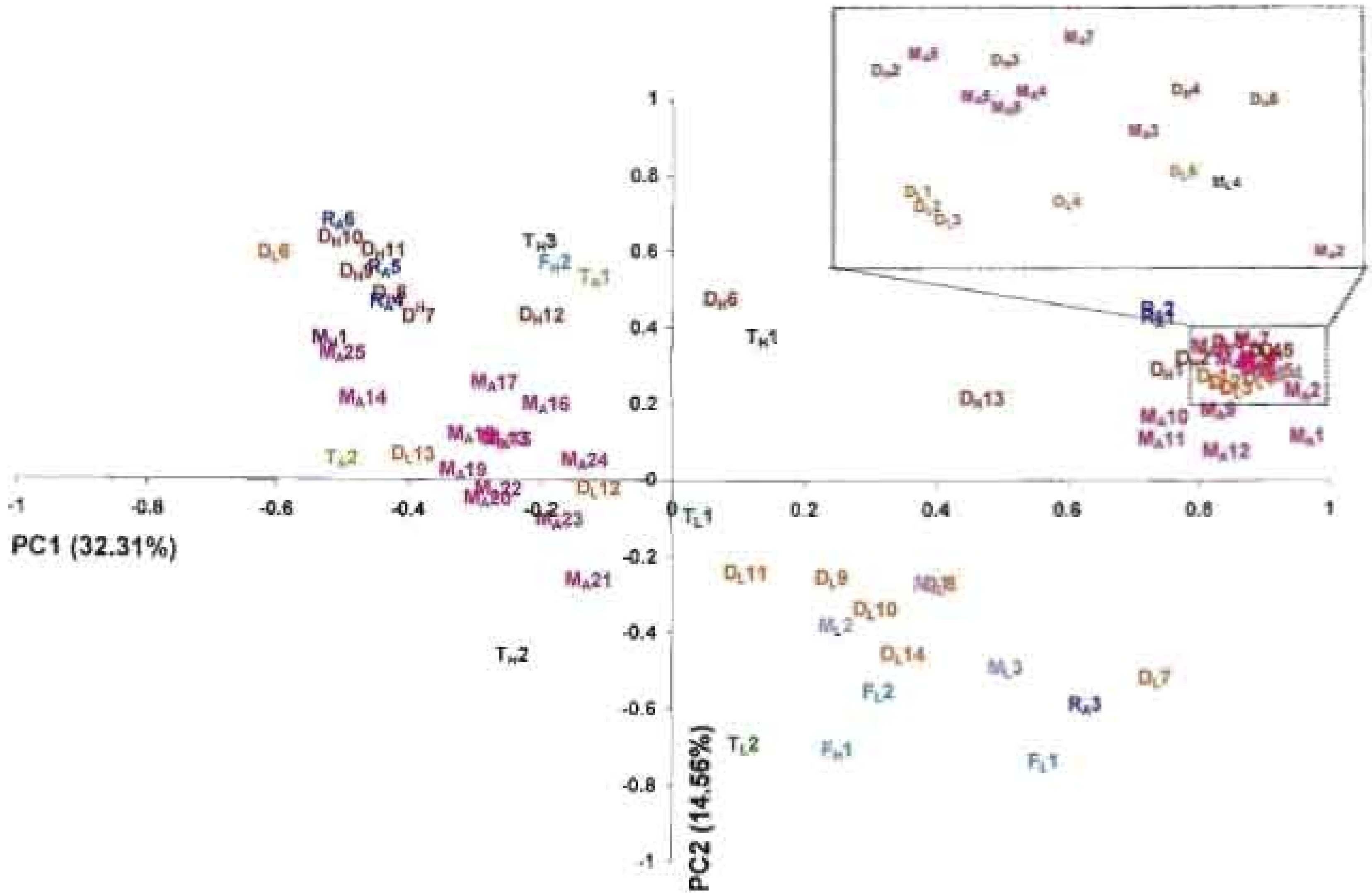

Figure 5.47 Ordinations of the 74 hydrological indices, from the PCA of 48 stream sites in South Africa, in the plane of the first two principal component axes. Correlations among the indices are interpreted as the cosine of the angle separating their index-axes. Each eigenvector was rescaled to the length $\sqrt{\lambda_4}$ to display the correlations among the indices. Some of the data have been shown in an enlargement box for clarity.

(b) A range of durations of high flow conditions (*i.e.* the 1-day, multiple-day and seasonal high flow disturbances);

(c) A range of durations of low flow conditions (*i.e.* the 1-day, multiple-day and seasonal low flow disturbances);

(d) The magnitude of low flow conditions (*i.e.* a flow threshold that is equaled or exceeded 75 per cent of the time, *i.e.* Q75); and

(e) The rates of change of average flow conditions (*i.e.* the rise and fall rates of river levels).

These highly correlated indices are shown in a cluster in the upper right quadrant of Figure 5.47.

Following the methods described in Section 4.4 of this Chapter, 12 hydrological indices (discussed in Section 4.4.3) with the largest absolute loadings on the statistically significant principal component axes for each streamflow type were selected. The results are shown in Table 5.26 which indicates the subsets of high information, relatively non-redundant indices (at least, among the subsets for each principal component axis) that describe the major gradients of variation in the indices for the different streamflow types. In addition, Table 5.26 includes the results of the analysis for the streamflow types resulting from the grouping of the 83 stations in the "working database" for comparison.

High information indices

Reducing the number of stream sites from 83 to 48 still results in all 7 main characteristics of the streamflow regime being represented in Table 5.26. However, different dominant indices are shown to be common across the different streamflow types. Rather than specific indices of the duration of high flow conditions being the most commonly shared, dominant indices among the different streamflow types, as in the case of the "working database", there are six occurrences of T_H1 (the date of the annual maximum flow) and three occurrences of T_H3 (seasonal predictability of flooding) as dominant indices across the diversity of streams represented by the "revised and reduced working database". Average flows for October (M_A1) still features as a common dominant index, but average flows for November (M_A2) and December (M_A3) and the variability of flows in March (M_A18) are also dominant indices, each having three occurrences among the different stream types.

Table 5.26 Comparison between the analyses conducted on the "working database" (WD) and the "*revised and reduced working database*" (RDD) regarding the hydrological indices with the largest absolute loadings on each statistically significant principal component. Streamflow types are based on a revised classification by Joubert and Hurley (1994). Indices annotated with "–" have a negative loading of the index on the principal component. Different colours represent the 7 main characteristics of the streamflow regime.

Principal Component	Streamflow Type													
	Extreme Seasonal		Mixed							Perennial				All Streams
	Extreme Seasonal		SUF	SUF	UUF	UF	SPFF	UFF	Perennial: Runoff		Perennial: Sustained Baseflow		All Streams	
	WD	RDD	WD	RDD	WD	RDD	WD	RDD	WD	RDD	WD	RDD	WD	RDD
1	D_L6 M_A1 D_H4 M_A6 D_H5-	M_A17- M_A11 T_L2 M_A1 D_L6 M_A24 M_A19	D_L12- M_A24- [illegible] D_L6-	D_H9 D_H12 D_H10 F_L2 M_A16	F_H1 M_A25- D_L12- D_H12- M_A24	T_H3 F_H1- D_L4 D_L6- M_A6	D_H4- M_A25- D_H5- D_H3-	M_L1 M_L3 D_L4 T_A1 D_L3	M_A1 D_H3 D_H4 [illegible]	M_A12 [illegible] T_H1- M_A11	M_L3 D_H2- D_H3- D_H4- M_A13- D_H1-	M_A18 M_L4- M_A1- M_A3- M_A2-	M_A1 M_A2 D_H5 M_A7 M_A6	M_A1 M_A2 D_H5 M_L4 D_H4
2	M_L1 [illegible] D_H8-	T_L1- T_L2 T_H1	M_A1 M_A9 M_A10	M_A4 M_A5 M_A8 M_A3-	M_A19 M_A11 M_A10	M_A11 M_A15 D_L12	M_A12 M_A11 T_H1	D_H6- F_H1 T_H1- [illegible]	D_L3 D_L1 D_L2	D_L1 D_L2 D_L3	D_L4 F_L2- D_L3	D_H2- D_H3-	T_H3- T_H1	F_L1- F_H1-
3	F_H2 T_L1- T_L2-	M_A25 [illegible]	M_A17 D_H7	M_L1- D_L1-	M_A2 D_L5-	[illegible] [illegible]	D_L1 D_L2	M_A2 M_A3	M_A21 M_A20	T_L2 T_H2	D_H8 T_H3	[illegible] D_H9	M_A28 M_L1-	M_A24- T_H1
4	M_A25		T_A2	M_A19-	D_L6-	D_H1	D_L8	M_A15	M_A15	[illegible]	D_L14	[illegible]	D_L10	M_A15
5			T_A2-		T_A1	T_L2	D_H8-		D_H8-	D_H10-		T_H1-	M_A22	T_H1-
6			M_A13				D_L7		F_H1	M_A18		T_H2	T_A2-	D_L8

High information indices at a broad spatial scale

Table 5.26 indicates that there is some similarity between the two datasets for high information indices, at least at a broad spatial scale (*i.e.* "All-Streams"), since there are four indices which are shared (*i.e.* M_A1, M_A2, the average monthly streamflow conditions of October and November; D_H5, the 90-day maximum flow and T_A1, Colwell's index of predictability). Reducing the number of sites in the analysis to 48 also results in six out of the seven main streamflow characteristics (*c.f.* Section 4.4.3 of this Chapter) being represented by high information indices. Again, no indices of the rate of change of streamflow conditions are identified as being of high information of the variation in the indices across the revised and reduced working database at this spatial scale. However, as a result of the exclusion of a large proportion of the DWAF gauging stations on perennial river systems from the revised and reduced working database, the index M_A25 (i.e. the index of overall variability calculated from the relationship between the inter-annual seasonality and the baseflow regime) does not feature in the list of high information indices (*c.f.* Table 5.26).

High information indices at a finer spatial scale

Extreme Seasonal Group

Reducing and regrouping the database results in only four, rather than seven, of the main streamflow characteristics being represented by the major sources of variation in the indices for the *Extreme Seasonal* regimes (*c.f.* Table 5.26). While there are six stations representing the Extreme Seasonal streamflow type in the "revised and reduced working database" rather than 10 corresponding stations in the "working database", there are five indices of high information which are shared between the two databases. As expected the number of days with zero flow (D_L6) and the timing of the minimum annual flow, and the variability thereof, (T_L1 and T_L2) are identified as dominant indices. However, it is encouraging to see that M_A1 (average flows for October) and M_A25 (overall variability), are again highlighted as indices of high information. Not only does this emphasise the importance of the first of season freshettes and overall variability to Extreme Seasonal streams, but the re-selection of these indices from the complete set of 74 in the PCA analysis highlights the usefulness of these indices for ecological studies of these river systems.

Mixed Main Group

The reduction of the database to 48 stations resulted in relatively more regrouping of the stations among the three streamflow types forming the "mixed" main group, than for the Extreme Seasonal Group. Consequently, there was some renaming of the groups to correspond with the distinguishing characteristics which separated the three "mixed" groups. For example, revising and reducing the "working database" to 48 stations failed to produce a streamflow group which could be described as "Seasonally Predictable Flow and Floods" as was the case with the grouping of the 83 stations of the "working database". This presented problems in comparisons of the streamflow types between the two databases (*c.f.* Table 5.26). While there was a fairly distinct Short, Unseasonal Floods group within each database, the following linking was necessary.

(a) The Unpredictable Flow group in the revised and reduced working database was compared to the Unpredictable Flow and Flood group within the 83 stations of the "working database", based on the similarity of the rankings of their distinguishing characteristics (*c.f.* Tables 5.10 and 5.23) and their positions towards the arid end of the environmental gradient as discussed in relation to both Figures 5.9 and 5.44.

(b) The Unpredictable Flow and Flood group in the revised and reduced working database was compared to the Seasonally, Predictable Flow and Floods group within the 83 stations of the "working database", based on the similarity of the rankings of some of their distinguishing characteristics (*c.f.* Tables 5.10 and 5.23) and their positions towards the moderate end of the environmental gradient as discussed in relation to both Figures 5.9 and 5.44.

There are no indices of high information shared between the "revised and reduced working database" and its parallel group in the "working database" for the Short, Unseasonal Floods regimes (*c.f.* Table 5.26). Similarly to the analysis of the "working database" there are five of the seven main streamflow characteristics represented by the major sources of variation in the indices for this streamflow type in the revised and reduced working database. One noteworthy feature is that the IHA baseflow index, M_L1 (*c.f.* Section 2.4.2 of this Chapter), appears to be important for describing this streamflow type. At this spatial scale, this index was criticised in the analysis in Section 4.5 of this Chapter for its unsuitability to characterising the streamflow conditions found in "highly variable" river systems.

Table 5.26 indicates that there are four indices of high information (D_L6, D_L12, F_H1 and M_A11) shared between the two datasets for the Unpredictable Flow streamflow type (equivalent to the Unpredictable Flow and Floods Group in the "working database"). These flow regimes are located towards the arid end of the environmental gradient within both datasets and, unsurprisingly, indices relating to low flow conditions (both D_L6 and D_L12) contain high information, even with the exclusion of some of the stations from the original "working database". Reducing and regrouping the database still results in six out of the seven main streamflow characteristics being represented by the major sources of variation in the indices of this streamflow type. However, M_A25 (the index of overall variability) is omitted from the list of high information, indices for the revised and reduced working database, whereas two indices relating to the rate of change of daily streamflow conditions (R_A4 and R_A5) are included for the revised and reduced working database.

As a result of the anomalies described above between the groupings of the two databases, there is only one high information index, T_H1 (the timing of the annual maximum flow), shared between the Seasonally Predictable Flow and Floods group within the 83 stations in the "working database" and its closest parallel of the Unpredictable Flow and Floods group in the "revised and reduced working database". Nonetheless, all seven of the main streamflow characteristics are represented by the high information indices for this streamflow type in the revised and reduced working database (*c.f.* Table 5.26). The IHA baseflow index, M_L1, appears to be important for describing streams in the Unpredictable Flow and Floods group in the "revised and reduced working database". However, this is also true for M_L3, the Alt-BFI based on the Desktop Reserve model index of short-term variability, representing "baseflow", the low amplitude and frequently occurring part of the hydrograph (*c.f.* Section 2.4.3 of this Chapter).

Perennial Group

As discussed above, many stations which were included in the perennial main group in the "working database" were found not to be recording reasonably natural flow. Excluding these stations from the revised and reduced working database resulted in the hydrological indices of streamflow regimes with a relatively large baseflow component being removed from the PCA. Consequently, there was less distinction between the Runoff and the Sustained Baseflow Streamflow Types in the revised and reduced working database than there was in the "working database". Nonetheless, as in the analysis of the "working database", five of the seven main streamflow characteristics are represented by the high information indices of the streamflow regimes in the Runoff Group. There are three high information indices (D_L1, D_L2, and D_L3, the shorter minimum flow events) shared between the two datasets for this streamflow type. Likewise, there are also five of the seven main streamflow characteristics represented by the high information indices of the streamflow regimes in the Sustained Baseflow Group formed from the revised and reduced working database, yet there are only two high information indices (D_H2, and D_H3, the shorter maximum flow events) shared between the two datasets for this streamflow type.

High information, non-redundant indices

Following the methods outlined in Section 4.4.3 of this Chapter, it is also possible to select high information, non-redundant indices with the condition that the indices represent each of the main facets of the streamflow regime. This was achieved by selecting the index with the highest absolute loading on each of the three to six significant principal component axes for each of the main streamflow characteristics as in Olden and Poff (2003). The results of this analysis, for the different streamflow types comprising the revised and reduced working database, are given in Table 5.27, which shows groups of indices which are relatively independent from each other since they are derived from different principle component axes (Olden and Poff, 2003). Some dominant indices are specific to particular streamflow types (*e.g.* M_A25, the CDB index of overall variability for Extreme Seasonal regimes), whilst others are shared among different streamflow types (*e.g.* D_H13, the seasonal predictability of non-flooding for groups at the arid end of the environmental gradient of streamflow regimes). Both M_L3 and M_L4 (the ALT BFI index, based on the Desktop Reserve model of short-term variability and Q75, respectively) are dominant indices across the range of different streamflow types.

Comparing the results from Table 5.27 with those in Table 5.16, it can be seen that there is some similarity between the revised and reduced working database and the working database, particularly with respect to the high information, non-redundant indices selected for both the Extreme Seasonal and for the All-Streams groups. In addition, the same high information, non-redundant indices representing the duration of high flow conditions (D_H2, D_H9, D_H8 and D_H12) were selected for the Sustained Baseflow group of the revised and reduced working database as were selected for the Sustained Baseflow group of the

Table 5.27 Hydrological indices with the highest absolute loadings on each of the three to six principal component axes for streamflow types in South Africa based on the revised and reduced working database of 48 streams. Indices are assigned to seven main streamflow characteristics in accordance with the largest loadings exhibited on each significant component. Superscripts denote the first to sixth principal components. Plain font denotes indices of central tendency (medians). Bold font denotes indices of dispersion (CD). Some indices are highlighted. Purple denotes indices used in the Desktop version of the South African Building Block Methodology (see text); blue denotes Colwell's indices of Predictability; green denotes the seasonal predictability of flooding; yellow denotes the seasonal predictability of non-flooding.

Streamflow Characteristic	Streamflow type						
	Extreme Seasonal	Mixed			Perennial		All Streams
	Extreme Seasonal	Short, Unseasonal Floods	Unpredictable Flow	Unpredictable Flow and Floods	Runoff	Sustained Baseflow	All-48 Streams
Magnitude of flow events							
Average flow conditions	M_A12^1 **M_A22^2** [illegible]^3	**M_A16^1** M_A4^2 **M_A22^3** **M_A19^4**	M_A6^1 M_A11^2 M_A1^3 M_A1^4 **M_A13^5**	**M_A13^1** M_A12^2 M_A2^3 **M_A18^4**	M_A12^1 M_A7^2 **M_A16^3** **M_A16^4** M_A5^5 **M_A18^6**	**M_A18^1** **M_A23^2** M_A10^3 **M_A17^4** **M_A15^5** **M_A14^6**	M_A1^1 M_A7^2 **M_A24^3** **M_A15^4** **M_A22^5** M_A11^6
Low or high flow conditions	M_H1^1 [illegible]^2 [illegible]^3	[illegible]^1 **M_L2^2** M_L1^3 [illegible]^4	[illegible]^1 M_H1^2 **M_L2^3** [illegible]^4 [illegible]^5	M_L1^1 **M_L2^2** [illegible]^3 **M_L2^4**	M_H1^1 [illegible]^2 **M_L2^3** [illegible]^4 [illegible]^5 M_H1^6	[illegible]^1 M_L1^2 M_L1^3 M_H1^4 **M_L2^5** [illegible]^6	[illegible]^1 [illegible]^2 **M_L2^3** M_L1^4 M_L1^5 **M_L2^6**
Duration of flow events							
Low flow conditions	D_L6^1 D_L5^2 **D_L13^3**	D_L7^1 **D_L11^2** D_L2^3 **D_L10^4**	D_L4^1 **D_L12^2** **D_L13^3** **D_L11^4** D_L5^5	D_L4^1 **D_L14^2** D_L5^3 **D_L10^4**	**D_L14^1** D_L1^2 **D_L7^3** **D_L12^4** **D_L14^5** D_L7^6	D_L4^1 **D_L8^2** D_L3^3 **D_L14^4** D_L7^5 **D_L14^6**	D_L5^1 D_L6^2 **D_L12^3** **D_L12^4** **D_L14^5** **D_L8^6**
High flow conditions	D_H5^1 D_H1^2 **D_H13^3**	**D_H9^1** **D_H13^2** D_H1^3 **D_H11^4**	D_H5^1 **D_H13^2** **D_H7^3** D_H1^4 **D_H11^5**	D_H2^1 D_H6^2 D_H5^3 **D_H7^4**	D_H2^1 **D_H9^2** D_H7^3 **D_H11^4** **D_H10^5** D_H6^6	**D_H12^1** D_H2^2 **D_H9^3** **D_H8^4** **D_H6^5** **D_H12^6**	D_H5^1 **D_H10^2** D_H6^3 **D_H7^4** **D_H8^5** D_H13^6
Timing of flow events	[illegible]^1 T_L1^2 [illegible]^3	[illegible]^1 [illegible]^2 T_H1^3 **T_L2^4**	[illegible]^1 T_L1^2 [illegible]^3 T_L2^4 T_L2^5	[illegible]^1 T_H1^2 T_L1^3 T_L1^4	T_H1^1 T_A1^2 **T_L2^3** [illegible]^4 [illegible]^5 [illegible]^6	[illegible]^1 [illegible]^2 [illegible]^3 [illegible]^4 T_H1^5 **T_H2^6**	[illegible]^1 **T_L2^2** [illegible]^3 [illegible]^4 T_H1^5 [illegible]^6
Frequency of flow events	F_H1^1 F_H1^2 **F_H2^3**	**F_L2^1** F_H1^2 **F_H2^3** F_L1^4	F_H1^1 F_H1^2 **F_H2^3** **F_H2^4** **F_L2^5**	**F_L2^1** F_H1^2 **F_L2^3** **F_H2^4**	**F_L2^1** F_L1^2 F_H1^3 **F_L2^4** **F_H2^5** F_H1^6	**F_L2^1** F_L1^2 **F_L2^3** **F_L2^4** F_L1^5 **F_H2^6**	F_L1^1 F_L1^2 **F_L2^3** **F_L2^4** **F_H2^5** F_H1^6
Rate of change in flow events	**R_A6^1** **R_A5^2** R_A2^3	**R_A4^1** R_A1^2 **R_A6^3** **R_A4^4**	**R_A6^1** **R_A6^2** **R_A5^3** R_A1^4 R_A3^5	R_A2^1 **R_A6^2** **R_A4^3** **R_A4^4**	R_A2^1 **R_A4^2** **R_A5^3** R_A3^4 **R_A6^5** **R_A6^6**	R_A2^1 R_A1^2 **R_A4^3** **R_A6^4** R_A3^5 **R_A6^6**	R_A1^1 **R_A6^2** R_A4^3 **R_A5^4** **R_A4^5** R_A1^6

working database. Moreover, similar dominant indices exist for the duration of low flow conditions (D_L4, D_L5 and D_L12) of the Unpredictable Flow and Flood group in the working database to those of the Unpredictable Flow group of the revised and reduced working database. Nonetheless, there are differences among the choice of indices representing the two databases as a result of revising and reducing the number of stream sites.

6.6 Summary and Conclusions

The "working database" developed in Section 3.5 of this Chapter, applied in Section 4 and discussed in Section 5 is inappropriate in terms of representing stations recording "reasonably natural streamflows" since it erroneously includes a number of DWAF gauging stations which record streamflows impacted by human alterations. A revised, yet substantially reduced, working database was derived for an initial investigation of the extent of the problem by assessing the water useage and land use upstream from each of the stations comprising the "working database".

There are clearly substantial differences between the two datasets regarding the dominant patterns of hydrological variability. This was to be expected since, as mentioned in Section 6.5 of this Chapter, "examining different stream sites could have lead to different results". Indeed, different indices did emerge as being of high information of the hydrological variability associated with the revised and reduced working database. Consequently, the choice of indices for characterising the diversity of streamflow regimes at the spatial scales applied in Section 4 of this Chapter is compromised by the inappropriate "working database".

However, application of the revised and reduced working database to meet the main aim of this Study, "to display correlations, or redundancies, among the hydrological indices and to identify a reduced subset (or subsets) of hydrological indices which explain a dominant proportion of statistical variation in the entire set of indices and which adequately represent the different facets of the streamflow regimes found in South Africa", also results in a compromised choice of indices, since it can be argued that the revised and reduced working database does not adequately represent the diversity of streamflow regimes found in South Africa. Consequently, after much deliberation by this author, and in consultation with the supervisors of this research, it was decided that there was limited usefulness in *further re-investigation* of the high information, non-redundant hydrological indices identified for the revised and reduced working database, at the spatial scales described.

7 GENERAL CONCLUSIONS

The use of multivariate statistics is increasing in ecological applications, particularly where large environmental datasets are involved. Despite any misgivings over the "working database", this Study has shown that multivariate analysis can be applied to the long term record of streamflows held by the DWAF to identify a reduced set of high information, non-redundant indices that explain the variation in a number of readily computable hydrological indices of the streamflow regimes in South Africa. The streamflow regime links many ecological processes in freshwater ecosystems. Consequently, long-term records are an important ecological resource for environmental flow assessment.

The results of the examination of the interrelationships between the 74 hydrological indices can be used to provide researchers and water users with guidance for the selection of high information, non-redundant indices relating to the major characteristics of the streamflow regimes at either a national to regional scale (based on the results from the combined set of All Streams) or at a regional to local scale (based on the results from the distinctly different streamflow types).

Although the analysis of the sensitivity of hydrological indices to record length was only conducted for the streamflows of the inappropriate "working database", the results of this component of the Study confirmed that by and large, longer records are necessary for indices which describe the variation associated with ephemeral and intermittent streams than are required for perennial streams. This Study indicates that "long-term" records should ideally be longer than 36 years for the derivation of ecologically relevant hydrological indices (based on the results from the combined set of All-43 Streams). However, the findings in this Study highlight the need for hydrological indices which describe relatively fine time steps, across different hydrogeographical regions.

The inclusion of the stations recording streamflows which are impacted by human alteration to natural systems in the Study conducted in Section 4 of this Chapter is regrettable. However, several lessons and conclusions can be drawn from this unfortunate situation, *viz.*:

(a) In order to fully meet the aim of the study, it is preferred to check the base data

The nature of the research in Section 4 of this Chapter was to identify a minimum subset of hydrological indices which adequately represent the different facets of the streamflow regimes found in South Africa, in their natural state. The reliance on the "working database" subsequently found to be inappropriate compromises the suitability of the indices highlighted in Section 4.5 of this Chapter as high information indices of natural, or reference, conditions for application in hydro-ecological studies. While the essence of science, and its development, is that new research builds on, or disputes, previous findings, the events that unfolded after the writing of much of this Chapter substantiate the merit of checking the base data, even if other well-respected researchers have used it in different studies.

(b) Data limitations are an inherent part of scientific research

One of the objectives of this Study was to identify high information indices for different streamflow types, *based on an analysis of indices derived from daily streamflow records which experienced the minimum of climatic variation.* Notwithstanding the flaws of the "working database", the events of the Study highlight some of the challenges that researchers face in data poor regions. While the study performed by Olden and Poff (2003) was able to utilise "good" or "better" records, from 420 sites across the USA, the number of stations recording reasonably natural flow, with the minimum of climatic variation, across the diversity of streamflow regimes found in South Africa is very much lower, at only 48. It could be argued that this sample size is too small for the type of analysis this Study set out to perform and from which to draw any meaningful conclusions.

(c) Revisiting the main aim of the Study

In essence the research presented in Sections 3, 4 and 5 of this Chapter represents a catalogue of high information indices of present-day conditions for the stream sites contained therein. Despite any misgivings, this information does have value since an important component of Reserve Determinations (*c.f.* Chapter 4) is the assessment of the trajectory of change in the ecological status of the river system from present-day ecological conditions. The tests for stationarity and non-homogeneity performed for the hydrological indices, as described in Section 3 of this Chapter, indicate that the majority of stations comprising the "working database" were found to be recording streamflows which showed little change over the time period tested (36 years from 1965 to 2000). Clearly, not all changes in ecological status, or indeed the present-day ecological status, are streamflow related (*e.g.* water quality changes as a result of soil erosion or pollution). Nonetheless, the present ecological status of a river system is inherently *hydrologically* related, since the hydrological cycle links aquatic ecosystems, terrestrial ecosystems and atmospheric processes. Thus, while the hydrological indices, described in Sections 3, 4 and 5 of this Chapter, are derived from records of streamflow discharge, the records do incorporate the *accumulated impacts* of human altered landscapes on the streamflow regime. Thus, knowledge of which indices describe the dominant patterns of hydrological variability for a particular stream type, in its present ecological state, *could* assist researchers in determining which streamflow components merit most focus when assessing any trajectory of change in ecological status.

A final word

Hydrological indices of natural streamflow variability are essential to environmental flow assessments and for setting water resource and ecological management targets. The findings of this Study could still be used as a frame of reference of the flow patterns of the different kinds of river systems in South Africa. It is anticipated that knowledge of the high information indices of hydrological variability for a particular streamflow type could assist water users in selecting the streamflow components required to sustain the beneficial ecosystem goods and services that water users need.

The absence of existing "reference" conditions, representing a natural river state, for many parts of South Africa is an undesirable situation that researchers face when using observed streamflow records as an ecological resource. In such instances the only alternative is to use the best estimate of a minimally impaired baseline state, using a hydrological model to simulate daily streamflow records.

* * * *

Understanding how the utilisation of water resources impacts on streamflow patterns and the consequent change in the generation of ecosystem goods and services remains a challenge. The alteration of the streamflow regime, as a result of human activity, is the focus of the Case Study which follows. Based on the findings of the Study in this Chapter, the Case Study presented in Chapter 6 utilises hydrological indices derived from long-term simulated streamflow records. The value of the techniques applied in Sections 4 and 6 of this Chapter, relating to the choice of indices for characterising a diversity of different streamflow patterns, is applied to assess the sustainability of different water resources management practices for the Mkomazi Catchment in one of the higher rainfall regions of South Africa, being, KwaZulu-Natal.

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APPENDIX 5A

Table 5A1 Gauging stations, the data from which comprised the "best200" daily streamflow database in the 1994-study, updated and assessed for further analysis in this Study

Gauging Station	Upstream Area (km^2)	Start Year*	End Year*	Longitude (degrees, decimal)	Latitude (degrees, decimal)	Weir Site	Selected for Analysis	Criteria for Rejection
		*hydrological yr						
A2H029*	129.0	1962	2001	28.383	-25.650	Edenvalespruit	Yes	
A2H032*	522.0	1963	2001	27.017	-25.633	Selonsrivier	Yes	
A2H039	3.6	1971	2001	27.183	-25.717	Waterkloof	No	Start Year post-1965
A2H049	371.0	1972	2001	27.833	-25.967	Bloubankspruit	No	Start Year post-1965
A2H050	148.0	1973	2000	27.833	-25.983	Krokodilrivier	No	Start Year post-1965
A3H001	1165.0	1906	1939	26.100	-25.533	Klein-Maricorivier	No	End Year pre-2000
A4H002*	1777.0	1948	2000	28.083	-24.267	Mokolorivier	Yes	
A4H005*	3786.0	1962	2000	27.767	-24.067	Mokolorivier	Yes	
A4H008*	504.0	1964	2000	27.967	-24.200	Sterkstroom	Yes	
A5H004*	629.0	1956	2000	28.400	-23.967	Palalarivier	Yes	
A6H011	73.0	1966	2000	28.333	-24.750	Groot-Nylrivier	No	Start Year post-1965
A6H012	120.0	1966	2000	28.467	-24.650	Olifantspruit	No	Start Year post-1965
A6H018	12.0	1973	2000	28.450	-24.650	Raslooprivier	No	Start Year post-1965
A6H019	16.0	1973	2000	28.550	-24.667	Hessie-se-Water	No	Start Year post-1965
A6H020	43.0	1973	2000	28.561	-24.671	Middelfonteinspruit	No	Start Year post-1965
A9H002	96.0	1931	1998	30.517	-22.900	Mutshindudirivier	No	End Year pre-2000
A9H003*	62.0	1931	2000	30.517	-22.883	Tshinanerivier	Yes	
A9H004*	320.0	1932	2000	30.539	-22.767	Mutalerivier	Yes	
B1H001	3989.0	1904	1950	29.317	-25.800	Olifantsrivier	No	End Year pre-2000
B1H002*	252.0	1956	2000	29.333	-25.817	Spookspruit	Yes	
B1H004*	376.0	1959	2000	29.167	-25.667	Klipspruit	Yes	
B2H001	1594.0	1905	1950	28.750	-25.783	Bronkhorstspruit	No	End Year pre-2000
B4H005*	188.0	1960	2000	30.217	-25.033	Watervalrivier	Yes	
B5H002	31416.0	1948	1978	29.800	-24.267	Olifantsrivier	No	Catchment > 10000 km^2
B6H001*	518.0	1910	2000	30.800	-24.667	Blyderivier	Yes	

B6H002	97.0	1910	1937	30.800	-24.667	Treurrivier	No	End Year pre-2000
B6H003*	92.0	1959	2000	30.800	-24.683	Treurrivier	Yes	
B6H006	43.0	1968	2000	30.546	-24.928	Kranskloofspruit	No	Start Year post-1965
B7H004	136.0	1951	1998	31.017	-24.550	Klaserivier	No	End Year pre-2000
B7H008	832.0	1956	1997	30.667	-24.000	Selatirivier	No	End Year pre-2000
B7H010*	318.0	1960	2000	30.433	-24.033	Ngwabaitsrivier	No	Several years missing
B9H001*	648.0	1960	2000	31.233	-22.833	Shisharivier	No	Several years missing
C1H007	4686.0	1972	2000	29.717	-26.833	Vaalrivier	No	Start Year post-1965
C2H026	26.0	1957	1995	27.667	-26.217	Middelvleispruit	No	End Year pre-2000
C2H027	4.0	1957	1991	27.650	-26.233	Kocksoortdspruit	No	End Year pre-2000
C2H028	31.0	1957	1991	27.583	-26.233	Rietfonteinspruit	No	End Year pre-2000
C2H065	860.0	1971	2000	26.350	-27.367	Leeudoringspruit	No	Start Year post-1965
C2H067	1895.0	1971	2000	26.233	-27.550	Sandspruit	No	Start Year post-1965
C3H003	10990.0	1927	2000	24.733	-27.567	Hartsrivier	No	Catchment > 10000 km^2
C5H007*	348.0	1923	2000	26.317	-29.133	Renosterspruit	Yes	
C5H008	593.0	1931	1984	26.200	-29.800	Rietrivier	No	End Year pre-2000
C5H012*	2372.0	1936	2000	25.967	-29.650	Rietrivier	No	Several years missing
C6H003	7765.0	1967	2000	26.617	-27.400	Valsrivier	No	Start Year post-1965
C7H003*	914.0	1947	2000	27.283	-27.350	Heuningspruit	Yes	
C8H003*	806.0	1954	2000	28.933	-27.833	Corneliusrivier	Yes	
C8H012	386.0	1971	2000	28.867	-28.067	Vaalbankspruit	No	Start Year post-1965
D1H003	37075.0	1914	2000	26.700	-30.667	Oranjerivier	No	Catchment > 10000 km^2
D1H009	24550.0	1960	2000	27.350	-30.333	Oranjerivier	No	Catchment > 10000 km^2
D2H001	13421.0	1926	1977	26.983	-29.700	Caledonriver	No	Catchment > 10000 km^2
D3H003	94765.0	1913	1950	24.200	-29.633	Oranjerivier	No	Catchment > 10000 km^2
D5H003*	1509.0	1927	2000	20.350	-31.800	Visrivier	Yes	
E1H006	160.0	1971	2000	18.933	-32.200	Jan-Disselsrivier	No	Start Year post-1965
E2H002*	6903.0	1923	2000	19.533	-32.533	Doringrivier	No	Several years missing
G1H003	46.0	1949	2000	19.067	-33.883	Franschhoekrivier	No	Not Available
G1H007	713.0	1951	1975	18.983	-33.800	Bergrivier	No	End Year pre-2000

G1H008*	395.0	1954	2000	19.067	-33.300	Klein-Bergrivier	Yes	
G1H009*	5.7	1964	2000	19.167	-33.383	Brakkloofspruit	Yes	
G1H010*	10.0	1964	2000	19.150	-33.383	Knolvleispruit	Yes	
G1H011*	27.0	1964	2000	19.100	-33.367	Watervalsrivier	Yes	
G1H012*	36.0	1964	1994	19.100	-33.350	Watervalsrivier	No	End Year pre-2000
G1H014	2.8	1964	1991	19.033	-33.817	Zachariahshoekrivier	No	End Year pre-2000
G1H015	1.9	1964	1986	19.050	-33.800	Kasteelkloofspruit-Bo	No	End Year pre-2000
G1H016	3.3	1964	1991	19.050	-33.817	Kasteelkloofspruit-Onder	No	End Year pre-2000
G1H017	1.7	1964	1986	19.017	-33.817	Zachariahshoekspruit	No	End Year pre-2000
G1H018	3.4	1964	1991	19.033	-33.817	Bakkerskloofspruit	No	End Year pre-2000
G2H008	20.0	1947	1993	18.950	-33.983	Jonkershoekrivier	No	End Year pre-2000
G2H012	244.0	1965	2000	18.733	-33.450	Dieprivier	Yes	
G3H001	647.0	1970	2000	19.750	-32.600	Kruisrivier	No	Start Year post-1965
G4H006*	600.0	1963	2000	19.600	-34.400	Kleinrivier	Yes	
G4H008	1.5	1964	1990	19.133	-34.150	Klein-Jakkalsrivier	No	End Year pre-2000
G4H009	2.0	1964	1990	19.133	-34.150	Jakkalsrivier	No	End Year pre-2000
G4H010	6.7	1964	1990	19.117	-34.167	Jakkalsrivier	No	End Year pre-2000
G4H012	0.7	1965	1990	19.133	-34.133	Klein-Jakkalsrivier	No	End Year pre-2000
G4H013	2.1	1965	1990	19.133	-34.133	Klein-Jakkalsrivier	No	End Year pre-2000
G4H014	252.0	1967	2000	19.233	-34.200	Botrivier	No	Start Year post-1965
G5H008*	382.0	1964	2000	20.017	-34.283	Soutrivier	Yes	
H1H007*	84.0	1935	2000	19.133	-33.567	Witrivier	Yes	
H1H013*	53.0	1965	2000	19.283	-33.350	Koekedourivier	Yes	
H1H017	61.0	1969	1990	19.100	-33.733	Elandsrivier	No	Start Year post-1965
H1H018	113.0	1969	2000	19.167	-33.717	Molenaarsrivier	No	Start Year post-1965
H2H001	697.0	1927	1987	19.500	-33.550	Hexrivier	No	End Year pre-2000
H2H003	718.0	1950	1984	19.500	-33.600	Hexrivier	No	End Year pre-2000
H2H005	15.0	1969	2000	19.533	-33.450	Rooi-Elskloofrivier	No	Start Year post-1965
H3H001	593.0	1925	1946	20.117	-33.783	Kingnarivier	No	End Year pre-2000
H3H004	14.0	1965	1990	19.917	-33.683	Keisierivier	No	End Year pre-2000
H3H005*	76.0	1965	2000	20.050	-33.700	Keisierivier	Yes	

H4H005	24.0	1950	1980	19.850	-33.750	Willem-Nelsrivier	No	End Year pre-2000
H4H012	14.0	1969	1990	19.583	-33.950	Waterkloofspruit	No	Start Year post-1965
H6H008	38.0	1964	1990	19.067	-34.050	Riviersonderend	No	End Year pre-2000
H6H010	15.0	1969	2000	19.317	-33.983	Waterkloofrivier	No	Start Year post-1965
H7H001	9829.0	1912	1939	20.383	-34.067	Breerivier	No	End Year pre-2000
H7H003	450.0	1949	1991	20.650	-34.000	Buffelsjagrivier	No	End Year pre-2000
H7H004*	28.0	1951	2000	20.700	-33.912	Huisrivier	Yes	
H9H004	50.0	1969	2000	21.283	-34.000	Kruisrivier	No	Start Year post-1965
H9H005	228.0	1969	2000	21.283	-34.083	Kafferkuilsrivier	No	Start Year post-1965
J1H015	8.8	1974	2000	19.717	-33.350	Bokrivier	No	Start Year post-1965
J2H005*	253.0	1955	2000	21.467	-33.483	Huisrivier	No	Several years missing
J2H006*	225.0	1955	2000	21.483	-33.483	Boplaasrivier	No	Several years missing
J2H007*	25.0	1955	2000	21.500	-33.483	Joubertrivier	No	Several years missing
J3H013	29.0	1966	2000	22.167	-33.367	Perdepoortrivier	No	Start Year post-1965
J3H017	348.0	1969	2000	22.133	-33.667	Kandelaarsrivier	No	Start Year post-1965
J4H003*	95.0	1965	2000	21.583	-34.017	Weyersrivier	Yes	
J4H004	99.0	1967	1995	21.767	-33.983	Langtourivier	No	Start Year post-1965
K3H002*	1.0	1961	2000	22.450	-33.933	Rooirivier	Yes	
K3H004*	34.0	1961	2000	22.417	-33.950	Malgasrivier	Yes	
K3H005	78.0	1969	2000	22.600	-33.933	Touwsrivier	No	Start Year post-1965
K4H001	111.0	1960	1991	22.800	-33.967	Hoekraalrivier	No	End Year pre-2000
K4H002*	22.0	1961	2000	22.833	-33.867	Karatararivier	Yes	
K4H003*	72.0	1961	2000	22.700	-33.900	Dieprivier	Yes	
K5H002*	133.0	1961	2000	23.017	-33.883	Knysnarivier	Yes	
K6H001*	165.0	1961	2000	23.133	-33.800	Keurboomsrivier	Yes	
K7H001*	57.0	1961	2000	23.633	-33.950	Bloukransrivier	Yes	
K8H001*	35.0	1961	2000	24.017	-33.967	Kruisrivier	Yes	
K8H002*	35.0	1961	2000	24.050	-33.967	Elandsrivier	Yes	
L1H001	3938.0	1917	1975	23.050	-32.233	Soutrivier	No	End Year pre-2000
L6H001*	1290.0	1926	2000	24.233	-33.200	Heuningkliprivier	No	Several years missing
L8H001*	21.0	1965	2000	23.833	-33.850	Waboomsrivier	Yes	

L8H002	52.0	1970	2000	23.300	-33.733	Haarlemspruit	No	Start Year post-1965
P4H001	576.0	1969	2000	26.733	-33.500	Kowierivier	No	Start Year post-1965
Q1H001	9091.0	1918	1991	25.467	-31.900	Groo-Visrivier	No	End Year pre-2000
Q3H004	872.0	1975	2000	25.517	-32.033	Paulsrivier	No	Start Year post-1965
Q9H002*	1245.0	1928	2000	25.297	-32.633	Koonaprivier	No	Several years missing
Q9H019	76.0	1972	2000	26.667	-32.550	Balfourrivier	No	Start Year post-1965
R1H001	238.0	1928	1979	26.850	-32.750	Tyumerivier	No	End Year pre-2000
R1H005	482.0	1949	1993	27.083	-32.750	Keiskammarivier	No	End Year pre-2001
R1H006	100.0	1946	1975	27.083	-32.750	Rabularivier	No	End Year pre-2002
R1H007	33.0	1949	1975	27.183	-32.633	Mtwakarivier	No	End Year pre-2000
R1H014*	70.0	1953	2001	26.933	-32.633	Tyumerivier	Yes	
R2H001*	29.0	1946	2001	27.283	-32.717	Buffelsrivier	Yes	
R2H005*	411.0	1947	2000	27.367	-32.867	Buffelsrivier	No	Many years missing
R2H006*	119.0	1948	2001	27.367	-32.850	Mgqakweberivier	Yes	
R2H008*	61.0	1947	2001	27.367	-32.767	Quencwerivier	Yes	
R2H012	15.0	1960	1996	27.250	-32.783	Mgqakweberivier	No	End Year pre-2000
S3H006*	2170.0	1964	2000	26.783	-31.917	Klaas-Smitrivier	Yes	
S6H003*	215.0	1964	2001	27.517	-32.500	Toiserivier	Yes	
T1H004*	4908.0	1956	2000	28.433	-31.917	Basheerivier	No	Many years missing
T2H002*	1199.0	1947	2001	28.783	-31.583	Mtatarivier	No	Poor Score (see Appendix 5A5)
T3H004*	1029.0	1947	2001	29.417	-30.567	Mzintlavarivier	Yes	
T3H008*	2471.0	1962	2000	29.150	-30.567	Mzimvuburivier	Yes	
T3H009*	307.0	1964	2000	28.350	-31.067	Mooirivier	Yes	
T4H001*	715.0	1951	2000	29.817	-30.733	Mtamvunarivier	Yes	
T5H002*	867.0	1934	2000	29.900	-30.400	Bisirivier	No	Many years missing
T5H003*	140.0	1949	2000	29.533	-29.733	Polelarivier	Yes	
T5H004*	545.0	1949	2000	29.467	-29.767	Mzimkulurivier	Yes	
U1H005**	1741.0	1960	2000	29.906	-29.744	Mkomazirivier	Yes	
U1H006*	4349.0	1962	2000	30.683	-30.167	Mkomaziriver	No	Several years missing
U2H001	937.0	1949	1991	30.233	-29.483	Mgenirivier	No	End Year pre-2000
U2H006*	339.0	1954	2000	30.267	-29.367	Karkloofrivier	Yes	

U2H007*	358.0	1954	2000	30.150	-29.433	Lionsrivier	Yes	
U2H011*	176.0	1958	2000	30.250	-29.633	Msunduzerivier	Yes	
U2H012*	438.0	1960	2000	30.483	-29.433	Sterkrivier	Yes	
U2H013*	299.0	1960	2000	30.117	-29.500	Mgenirivier	Yes	
U3H002	356.0	1950	1975	31.017	-29.600	Mdlotirivier	No	End Year pre-2000
U4H002*	316.0	1949	2000	30.617	-29.150	Mvotirivier	Yes	
U7H007*	114.0	1964	2000	30.233	-29.850	Lovurivier	Yes	
V1H001*	4176.0	1925	2000	29.817	-28.733	Tugelarivier	Yes	
V1H002	1689.0	1932	1968	29.350	-28.733	Tugelarivier	No	End Year pre-2000
V1H009*	196.0	1955	2000	29.767	-28.883	Bloukransrivier	Yes	
V1H010*	782.0	1965	2000	29.533	-28.817	Klein Tugelarivier	Yes	
V1H031	162.0	1970	2000	29.350	-28.717	Sandspruit	No	Start Year post-1965
V1H038	1644.0	1971	2000	29.750	-28.550	Kliprivier	No	Start Year post-1965
V2H001	1976.0	1931	1974	30.350	-29.017	Mooirivier	No	End Year pre-2000
V2H005	260.0	1972	2000	29.867	-29.359	Mooirivier	No	Start Year post-1965
V3H002*	1518.0	1929	2000	29.933	-27.600	Buffelsrivier	Yes	
V3H003	850.0	1929	1960	29.950	-27.917	Ngaganerivier	No	End Year pre-2000
V3H005	676.0	1947	1991	29.967	-27.433	Slangrivier	No	End Year pre-2000
V3H007*	129.0	1948	2000	29.833	-27.833	Ncandurivier	Yes	
V3H009*	148.0	1958	2000	29.950	-27.883	Hornrivier	Yes	
V6H003*	312.0	1954	2000	30.133	-28.300	Wasbankrivier	Yes	
V6H004*	658.0	1954	2000	30.000	-28.400	Sondagsrivier	Yes	
V6H006	109.0	1968	2000	29.750	-28.233	Sondagsrivier	No	Start Year post-1965
V7H012*	196.0	1963	2000	29.867	-29.000	Kleinboesmanrivier	Yes	
V7H016	121.0	1976	1990	29.633	-29.183	Ncibidwanerivier	No	Start Year post-1965
V7H017	276.0	1972	2000	29.633	-29.183	Boesmansrivier	No	Start Year post-1965
W1H004*	20.0	1948	2000	31.450	-28.867	Mlalazirivier	Yes	
W3H014	48.0	1969	2000	32.350	-28.317	Mpaterivier	No	Start Year post-1965
W4H004*	948.0	1950	2000	30.850	27.517	Bivanerivier	Yes	
W5H001	15.0	1910	1990	30.550	-26.250	Jessievalespruit	No	End Year pre-2000
W5H004	460.0	1950	1989	30.467	-26.750	Ngwenpisirivier	No	End Year pre-2000

W5H006*	180.0	1950	2000	30.833	-27.100	Swartwaterrivier	Yes	
W5H008*	701.0	1951	2000	30.633	-26.467	Bonniebrook	No	Poor Score (see Appendix 5A5)
X2H005*	642.0	1950	2000	30.967	-25.417	Nelsrivier	Yes	
X2H008*	180.0	1948	2000	30.917	-25.783	Queensrivier	Yes	
X2H010*	126.0	1948	2000	30.867	-25.600	Noordkaaprivier	Yes	
X2H011	402.0	1956	1998	30.267	-25.633	Elandsrivier	No	End Year pre-2000
X2H012*	91.0	1956	2000	30.250	-25.650	Dawsoni'sspruit	Yes	
X2H013*	1518.0	1959	2000	30.700	-25.433	Krokodilrivier	Yes	
X2H014*	250.0	1959	2000	30.700	-25.367	Houtbosloop	Yes	
X2H015*	1554.0	1959	2000	30.683	-25.483	Elandsrivier	Yes	
X2H022*	1639.0	1960	2000	31.317	-25.533	Kaaprivier	Yes	
X2H024*	80.0	1964	2000	30.817	-25.700	Suidkaaprivier	Yes	
X2H025	25.0	1966	1990	30.567	-25.283	Houtbosloop	No	Start Year post-1965
X2H026	14.0	1966	1990	30.567	-25.283	Beestekraalspruit	No	Start Year post-1965
X2H027	78.0	1966	1990	30.583	-25.283	Blystaanspruit	No	Start Year post-1965
X2H028	5.7	1966	1990	30.567	-25.283	Kantoorbosspruit	No	Start Year post-1965
X2H031	262.0	1966	2000	30.967	-25.717	Suidkaaprivier	No	Start Year post-1965
X3H001*	174.0	1948	2000	30.767	-25.088	Sabierivier	Yes	
X3H002*	55.0	1964	2000	30.667	-25.083	Klein-Sabierivier	Yes	
X3H003*	52.0	1948	2000	30.800	-24.983	Mac-Macrivier	Yes	
X3H006	766.0	1958	1999	31.117	-25.017	Sabieriver	No	End Year pre-2000
X3H007	46.0	1963	1989	31.000	-25.150	Whitewatersrivier	No	Not Available

* Denotes stations screened for predominantly good, continuous data between the years 1965-2000

** DWAF gauging station omitted from the 1994-Study as a result of different selection criteria

APPENDIX 5B SCREENING PROCEDURE FOR THE SELECTION OF THE SOUTH AFRICAN DWAF STREAMFLOW STATIONS

This Appendix describes the screening procedure, shown in Figure 5.4, for the selection of DWAF streamflow records used in the multivariable approach to identify high-information, non-redundant hydrological indices of the river systems found in South Africa.

5B1 Transforming the DWAF Data to the IHA Format for Extraction of Indices

The data comprising potentially useable records (*i.e.* a relatively long time span with predominantly good continuous data) were transformed from the DWAF Hydrological Information Service format to the IHA format with a macro written in Microsoft Visual Basic programming language at the BEEH. The macro was written to arrange the DWAF daily average flow rates in columns, representing hydrological years, in a Microsoft Excel Spreadsheet. Although the developers of the IHA state that the IHA program does not object to missing years, long periods of missing data present problems to the statistical analysis of daily time series. The macro was programmed to flag data which were either temporarily or permanently "missing" in the DWAF records. Any IHA results from datasets with missing records need to be viewed with caution, since the IHA software automatically performs linear interpolation across gaps in the data which are appropriately flagged and interpolates across year boundaries if required. The incidence of "missing" streamflow values in the DWAF records was checked at this stage to ensure that the values interpolated by the IHA program would be realistic and within the natural range for the affected season. In particular, records were checked to ensure that "infilling" did not overlap from one season to another. Nevertheless, "missing" days featured in several of the DWAF records for rivers in periods which otherwise recorded zero flow or very low flow (those records highlighted in yellow in Table 5B1). Therefore it was considered that, even over long gaps (*i.e.* in excess of 100 days) infilling would provide reasonable estimates of flows. Where long gaps spanned seasons in the records of quasi-perennial-seasonal (*c.f.* Section 4.3.1. of this thesis) rivers (highlighted in blue in Table 5B1), or of perennial rivers (highlighted in green in Table 5B1) it was considered that, infilling would be justified by the benefits of having a predominantly "good or better" record available for further analysis. In addition, since the main analyses in this study (multivariate analysis and the sensitivity of high information hydrological indices to record length) focus essentially on statistical moments of the central tendency (median) and dispersion (CD) of the hydrological indices derived from daily flows, it was considered that some infilling would be acceptable.

Table 5B1 Information relating to the completeness of the DWAF records of average daily streamflow used in this study. Where the Julian dates of the annual 1-day maximum flow or the annual 1-day minimum flow are spread over different seasons throughout the time span analysed, the information is indicated as "dates of extreme flow widely scattered". Yellow highlight denotes stations with missing days in periods which otherwise recorded zero flow or very low flow. Blue denotes stations sited on quasi-perennial-perennial rivers with long periods of missing data. Green denotes stations sited on perennial rivers with long periods of missing data.

DWAF Gauging Station	1959 - 2000		1965 - 2000		1981 - 2000	
	Longest period of missing days	Dates of extreme flow widely scattered	Longest period of missing days	Dates of extreme flow widely scattered	Longest period of missing days	Dates of extreme flow widely scattered
A2H029	*	*	35	yes	**	**
A2H032	*	*	58	no	**	**
A4H002	113	no	113	no	36	no
A4H005	*	*	214	yes	**	**
A4H008	*	*	22	no	**	**
A5H004	32	no	32	no	32	no
A9H003	374	no	374	no	374	no
A9H004	144	no	144	no	144	no
B1H002	313	yes	313	yes	228	no
B1H004	514	no	514	no	78	yes
B4H005	*	*	57	no	**	**
B6H001	209	no	209	no	209	no
B6H003	14	no	14	no	8	no
C5H007	60	no	60	no	60	no
C7H003	152	no	152	no	57	no
C8H003	624	yes	624	yes	495	yes
D5H003	43	no	43	no	43	no
G1H008	322	no	322	no	22	no
G1H009	*	*	169	no	**	**
G1H010	*	*	80	no	**	**
G1H011	*	*	41	no	**	**
G2H012			29	no		
G4H006	*	*	50	no	**	**

G5H008	*	*	22	no	**	**
H1H007	377	no	85	no	85	no
[illegible]			217	no		
H3H005			215	no		
H7H004	*1960 - 1964 missing*		15	yes	**	**
[illegible]			128	yes		
K3H002	*	*	343	yes	**	**
K3H004	*	*	66	yes	**	**
K4H002	*	*	50	yes	**	**
[illegible]	*	*	161	yes	**	**
[illegible]	*	*	145	yes	**	**
K6H001	*	*	36	yes	**	**
K7H001	*	*	64	yes	**	**
K8H001	*	*	16	yes	**	**
[illegible]	*	*	227	yes	**	**
[illegible]			242	no		
R1H014	35	yes	17	yes	17	no
R2H001	23	yes	15	no	0	no
R2H006	134	no	134	no	0	no
R2H008	351	no	351	no	8	no
S3H006	*	*	253	yes	**	**
S6H003	*	*	354	yes	**	**
[illegible]	259	yes	259	yes	64	yes
T1H004	212	no	212	no	22	no
T3H008	*	*	15	yes	**	**
T3H009	*	*	28	no	**	**
T4H001	52	no	52	no	43	no
T5H003	*1960 - 1964 missing*		100	no	**	**
T5H004	84	no	84	no	84	no
U1H005	*	*	85	no	**	**
U2H006	61	no	46	no	29	no
U2H007	100	no	100	no	57	no
[illegible]	576	yes	576	yes	476	
U2H012	*	*	60	no	**	**
U2H013	*	*	0	no	**	**
[illegible]	106	no	106	no	22	no
[illegible]	*	*	184	yes	**	**
[illegible]	124	no	124	no	124	no
V1H009	32	no	22	no	22	no
V1H010			19	no		
V3H002	221	no	221	no	221	no
V3H007	181	no	181	no	64	no
V3H009	**1959 - 1961 missing**		23	no	**	**
V6H003	99	no	99	no	99	no
V6H004	134	no	134	no	8	no
[illegible]	*	*	123	yes	**	**
W1H004	730	no	730	no	730	no
W1H005	581	no	581	no	581	**
W5H006	537	no	537	no	57	no
W5H008	31	no	31	no	20	no
X2H005	16	no	16	no	2	no
X2H008	69	no	31	no	31	no
X2H011	217	no	217	no	0	no

X2H012	71	no	62	yes	34	yes
X2H013	55	no	15	no	0	yes
X2H014	429	no	429	no	29	no
X2H015	62	no	62	no	62	no
X2H022	*	*	117	no	**	**
X2H024	*	*	15	no	**	**
X3H001	249	no	27	no	8	no
X3H002	*	*	29	no	**	**
X3H003	6	no	8	no	8	no

During the study, a problem of applying the IHA software to data recorded by the DWAF gauging network was identified for the streamflow regimes of some rivers. In the IHA software, computations are performed in floating point arithmetic to single-precision machine accuracy (about 4 significant figures on a Pentium PC), yet the output files are limited to two significant figures for some indices. While this is sufficient for data relating to rivers in more temperate regions, the daily average flow rates recorded for South African rivers are quite dissimilar to those of temperate regions. Daily flow rates of 1 x $10^{-3}m^3.s^{-1}$ are recorded by the DWAF for some rivers. Consequently, and on the advice of the software developer, the scale factor in the "Set-up" dialogue box in the IHA application was used to work in units of $10^{-3}m^3.s^{-1}$ so that the smallest datum was in the order of 1. Thereafter, the relevant values in the IHA output files were re-converted by multiplying by 10^{-3}.

5B2 Test for Absence of Linear Trend

The annual time series of each of the 35 intra-annual indices for each of the 85 DWAF sites was tested for absence of linear trend with Spearman's Rank-Correlation method. The Spearman Rank-Correlation method is a simple, distribution free test which has the advantage of nearly uniform power for linear and non-linear trends (WMO, 1996) and is frequently used in ecological numerical applications. The following statistical description and information is sourced from Dahmen and Hall (1990). The basis of the method is the Spearman rank-correlation coefficient, R_{sp}, which is defined as:

$$R_{sp} = 1 - \frac{6 * \sum_{i=1}^{n} D_i * D_i}{n * (n * n - 1)}$$

where n is the total number of data (*e.g.* 36 for the time series 1965 to 2000), D is the difference, and i is the chronological order number. The difference between the two rankings is calculated from:

$$D_i = Kx_i - Ky_i$$

where K_{xi} is the rank of variable x (*e.g.* the chronological order of the years 1965 through 2000). The series of observations, y_i is transformed to its rank equivalent Kyi, by assigning the chronological order number in the ranked series, y (the yearly values of the intra-annual indices). The null hypothesis, $H_0:Rsp = 0$ (no trend), is tested against the alternative hypothesis, $H_1:Rsp, <> 0$ (trend exists), with the test statistic t_t:

$$t_t = R_{sp}\left[\frac{n-2}{1-R_{sp} * R_{sp}}\right]^{0.5}$$

where t_t has the Student's t-distribution with v = n - 2 degrees of freedom. The Genstat Version 6 statistical software was applied to test for the absence of trend with computations of the Spearman-Rank Correlation coefficient, R_{sp} and the Student's t-distribution, t_t (Dahmen and Hall, 1990). The null hypothesis (*i.e.* the time series has no trend) was accepted if:

$$t\{v,2.5\%\}, t_t, t\{v,97.5\%\}$$

where the critical values of t_t at the 5% level of significance (two-tailed) for 36 - 2 = 34 degrees of freedom (*i.e.* for each time series of 36 years) are:

$$t\{34,2.5\%\} = -2.02 \text{, and } t\{34,97.5\%\} = 2.02$$

The results of this test, performed for the annual time series of each of the 35 intra-annual indices for each of the 85 DWAF sites, are too unwieldy to present in this thesis. However, an example of the results for the 36 year time period from 1 October 1965 to 30 September 2001 is provided in Appendix 5C for DWAF site U1H005 on the Mkomazi River in KwaZulu-Natal (*c.f.* Table 5A1 and Table 5.1).

5B3 Test for Stability of Median

In order to test the time series of each of the 35 intra-annual indices for stability of the median, the computed annual values at each site, were split into two, non-overlapping time series. In each instance, the medians of both subsets were compared for differences between the groups using the Mann-Whitney *U*-test. Fowler and Cohen (1990) state that the Mann-Whitney *U*-test is a non-parametric method which converts the values of observations to their ranks, and because the test distribution free, it is suitable for counts of things (*e.g.* high and low pulses) or derived variables such as proportions (*e.g.* the Alt-BFI). The Mann-Whitney *U*-test is a two-sample test of location (central tendency) difference, the null hypothesis being that there is no significant difference between the medians of the two samples, implying that the median for the time series can be considered to be stable. The following statistical information is sourced from the Genstat Version 6 statistical package which cites Siegel (1956) as a reference. The test statistic U is formed by assigning ranks to the combined data set, and is taken to be the smaller of the test statistics U_1 and U_2, calculated from:

$$U_k = n_1 n_2 + \frac{n_k(n_k+1)}{2} - R_k$$

where n_k is the size of sample $k = 1, 2$ and R_k is the sum of ranks for sample k. The sample with the lowest score, (*i.e.* the smaller of the two U values) is compared with established critical values in order to reject H_0 (Fowler and Cohen, 1990). For example, for two subsets of a 36 year time series (*i.e.* two subsets, both of 18 years), the null hypothesis was rejected where the smaller of the two U values was less than the critical value (at $n_1 = 18$ and $n_2 = 18$; $P<0.05$) of 99 (*i.e.* there is a statistically significant difference between the medians where the smaller U value is less than 99). The Mann-Whitney *U*-test of significance was applied using the Genstat Version 6 statistical package. While there is no requirement for the observations in the samples to be normally distributed, the Mann-Whitney *U*-test does assume that the two distributions are similar (Fowler and Cohen, 1990). Hence, "it is not permissible to compare the median of a positively skewed distribution with that of a negatively skewed distribution" (Fowler and Cohen, 1990). The distributions of the annual values of indices within each sample were assessed with Microsoft Excel (MS Office 2000) descriptive statistics analysis to ensure that any skewness was similar. Where the samples were found to be from different frequency distributions, the Mann-Whitney *U*-test is not valid and this additional provision was applied to the test for stability of the median.

Again, the results of this test, performed for the annual time series of each of the 35 intra-annual indices for each of the 85 DWAF sites, are too unwieldy to present in this thesis. However, an example of the results for the 36 year time period from 1 October 1965 to 30 September 2001 is provided in Appendix 5C for DWAF site U1H005 on the Mkomazi River in KwaZulu-Natal (*c.f.* Table 5A1 and Table 5.1).

5B4 Test for Stability of Dispersion

In addition to testing the time series of each of the 35 intra-annual indices for absence of trend, stability of the median and direction of skewness, the time series were also tested to determine whether the two non-overlapping samples of data described in Appendix 5B3 were different in respect of *dispersion from the median*. This was achieved by applying the Kolmogorov-Smirnoff two-sample test, which tests the null hypothesis that the two samples (in this instance two subsets of time series of the indices) originate from the same distribution. The Kolmogorov-Smirnoff two-sample test does not distinguish whether the samples differ in respect of location *or* skewness *or* dispersion (Siegel and Castellan, 1988). However, having already tested for differences in location and skewness by the procedure described in Appendix 5B3, any rejection of the null hypothesis that Kolmogorov-Smirnoff test guards against, would infer that the data comprising the two samples (*i.e.* the two subsets of time series of the indices) differ in respect of dispersion. The following statistical information is sourced from the Genstat Version 6 statistical package which cites Siegel (1956) as a reference. The Kolmogorov-Smirnoff test compares the two cumulative distribution functions, S_1 and S_2, in order to identify differences in shape of the underlying distributions. S_1 and S_2 are formed by:

$S_k(X)$ = (number of scores in sample $k \leq X$) / (size of sample k)

where S_k is the size of sample k = 1, 2; and X is an appropriate set of values taken from either sample k_1 or k_2. The maximum absolute difference:

$$MD = \max(abs\{S_1(X) - S_2(X)\})$$

is used as the basis for significance tests. The chi-square approximation (2 degrees of freedom) to this statistic is χ^2:

$$\chi^2 = 4 * MD * MD * (n_1 * n_2 / n_1 + n_2)$$

where n_1 and n_2 are the sizes of the samples. For each time series of 36 years, the null hypothesis was rejected where the test statistic χ^2 was greater than the critical value (on 2 degrees of freedom; $P<0.05$) of 5.99 (*i.e.* the two samples differ in respect of dispersion if the χ^2 value is greater than 5.99). The Kolmogorov-Smirnoff two-sample test was applied using the Genstat Version 6 statistical package.

Again, the results of this test, performed for the annual time series of each of the 35 intra-annual indices for each of the 85 DWAF sites, are too unwieldy to present in this thesis. However, an example of the results for the 36 year time period from 1 October 1965 to 30 September 2001 is provided in Appendix 5C for DWAF site U1H005 on the Mkomazi River in KwaZulu-Natal (*c.f.* Table 5A1 and Table 5.1).

5B5 Scoring System to Determine the Usefulness of the DWAF Records for Deriving Hydrological Indices

Scores were assigned to the results of the tests comprising Appendices 5B2, 5B3 and 5B4. The scoring system applied to determine those DWAF gauging stations which were recording daily streamflows that could be used to derive stationary time series of each the 35 intra-annual indices was as follows:

(a) If the null hypothesis of the Spearman Rank-Correlation method (Appendix 5B2) is accepted, indicating that the time series has no trend, one point is allocated. If the null hypothesis is rejected, no points are allocated for the existence of either a positive or negative trend in the time series.

(b) If the null hypothesis of the Mann-Whitney *U*-test (Appendix 5B3) is accepted, indicating that two sequential subsets of the time series have similar medians and, *in addition*, the skewness of the two distributions is similar, two points are allocated. If the null hypothesis is rejected, no points are allocated since the median of the time series cannot be considered to be stable and the additional skewness test need not be applied.

(c) If the null hypothesis of the Kolmogorov-Smirnoff test (Appendix 5B4) is accepted, indicating that the two subsets of sequential time series arise from the same distribution, one point is allocated. If the null hypothesis is rejected, no points are allocated for this component of the scoring system.

Thus a time series of any of the 35 intra-annual indices could attract a score of:

(a) Four points if the data passed all the tests;

(b) Three points if the data did not pass *either* the test for trend *or* the test of stability of the dispersion;

(c) Two points if the data did not pass the test for absence of linear trend and the test of stability of the dispersion, but did pass the test for the stability of the median;

(d) Two points if the data did not pass the test for the stability of the median, but did pass the test for absence of linear trend as well as the test of stability of the dispersion;

(e) One point if the data passed only the test for trend *or* the test for stability of the dispersion;

(f) No points if the data did not pass any of the tests performed.

Thus it was possible for a DWAF gauging station record to achieve a total score of 140 points if all 35 intra-annual indices passed all three tests. The quality classes, and defining criteria, allocated to total scores of the combined set of the 35 intra-annual indices derived from each of the 85 DWAF records of average daily flows are shown in Table 5B2.

Table 5B2 Quality classes and defining criteria allocated to total scores, as a result of tests for stationarity, consistency and homogeneity, of the combined set of 35 intra-annual indices derived from each of the 85 DWAF records

Quality class	**Score**
Excellent	140
Very Good	130 – 139
Good	100 – 129
Fair	70 – 99
Poor	Less than 70

5B6 REFERENCES

Dahmen, E.R. and Hall, M.J. (1990). Screening of hydrological data: Test for stationarity and relative consistency. International Institute for Land Reclamation and Improvement, Wageninen, The Netherlands. Publication 49.

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APPENDIX 5C

Table C1 Summary of results and scores for the tests for stationarity, consistency and homogeneity of the combined set of 35 intra-annual indices derived from the DWAF record at site U1H005

INDICATOR	TEST	RESULT	SCORE	INDICATOR	TEST	RESULT	SCORE	INDICATOR	TEST	RESULT	SCORE
Mean flow in October	Trend	no	1	1-day minimum flow	Trend	no	1	Date of minimum flow	Trend	no	1
(M_A1)	Median	√	2	(D_L1)	Median	√	2	(T_L1)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	*	0		Dispersion	√	1
Mean flow in November	Trend	no	1	3-day minimum flow	Trend	no	1	Date of maximum flow	Trend	no	1
(M_A2)	Median	√	2	(D_L2)	Median	√	2	(T_H1)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	*	0		Dispersion	√	1
Mean flow in December	Trend	no	1	7-day minimum flow	Trend	no	1	Number of low pulses	Trend	no	1
(M_A3)	Median	√	2	(D_L3)	Median	√	2	(F_L1)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in January	Trend	no	1	30-day minimum flow	Trend	no	1	Low pulse duration	Trend	no	1
(M_A4)	Median	√	2	(D_L4)	Median	√	2	(D_L7)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in February	Trend	no	1	90-day minimum flow	Trend	no	1	Number of high pulses	Trend	no	1
(M_A5)	Median	√	2	(D_L5)	Median	√	2	(F_H1)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in March	Trend	no	1	1-day maximum flow	Trend	no	1	High pulse duration	Trend	no	1
(M_A6)	Median	√	2	(D_H1)	Median	√	2	(D_H6)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in April	Trend	no	1	3-day maximum flow	Trend	no	1	Rise rate	Trend	no	1
(M_A7)	Median	√	2	(D_H2)	Median	√	2	(R_A1)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in May	Trend	no	1	7-day maximum flow	Trend	no	1	Fall rate	Trend	no	1
(M_A8)	Median	√	2	(D_H3)	Median	√	2	(R_A2)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in June	Trend	no	1	30-day maximum flow	Trend	no	1	Reversals	Trend	no	1
(M_A9)	Median	√	2	(D_H4)	Median	√	2	(R_A3)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in July	Trend	no	1	90-day maximum flow	Trend	no	1	Alternative Baseflow	Trend	no	1
(M_A10)	Median	√	2	(D_H5)	Median	√	2	(Alt BFI)	Median	√	0
	Skewness	√	-		Skewness	√	-	(M_L3)	Skewness	*	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in August	Trend	no	1	Zerodays	Trend	no	1	High flow index (HFI)	Trend	no	1
(M_A11)	Median	√	2	(D_L6)	Median	√	2	(M_H1)	Median	√	2
	Skewness	√	-		Skewness	√	-		Skewness	√	-
	Dispersion	√	1		Dispersion	√	1		Dispersion	√	1
Mean flow in September	Trend	no	1	IHA Baseflow	Trend	no	1	Total Score			134
(M_A12)	Median	√	2	(M_L1)	Median	*	0				
	Skewness	√	-		Skewness	√	-	NB	√	passed test	
	Dispersion	√	1		Dispersion	√	1		*	failed test	

CHAPTER 6 MAXIMISING THE GENERATION OF ECOSYSTEM GOODS AND SERVICES OF THE MKOMAZI CATCHMENT, KWAZULU-NATAL, SOUTH AFRICA

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STRUCTURE OF CHAPTER 6 MAXIMISING THE GENERATION OF ECOSYSTEM GOODS AND SERVICES FOR FUTURE OPTIONS OF THE MKOMAZI CATCHMENT, KWAZULU-NATAL

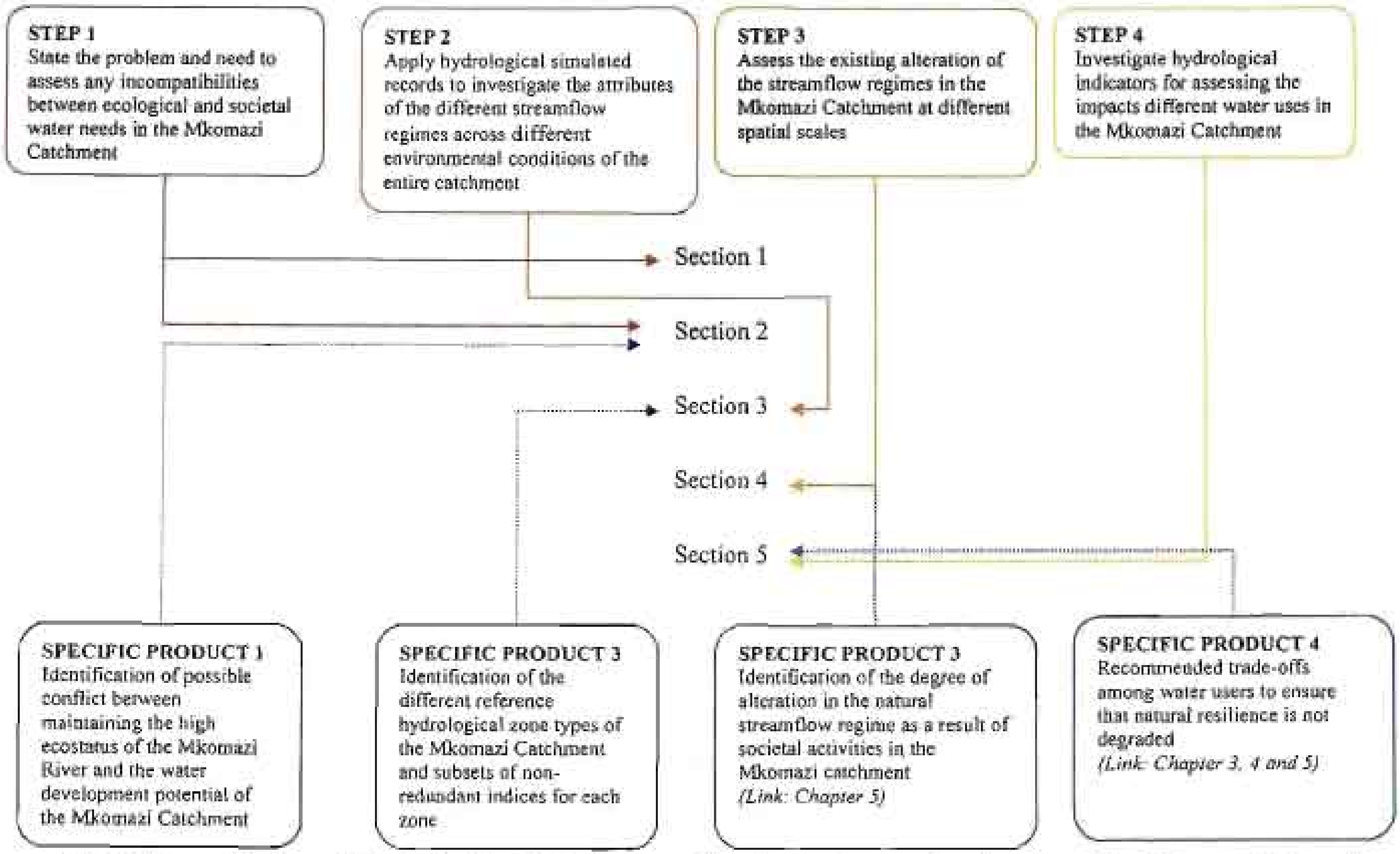


Figure 6.1 Structure of Chapter 6: Maximising the generation of ecosystem goods and services of the Mkomazi Catchment

CHAPTER 6 MAXIMISING THE GENERATION OF ECOSYSTEM GOODS AND SERVICES FOR FUTURE OPTIONS OF THE MKOMAZI CATCHMENT, KWAZULU-NATAL, SOUTH AFRICA

1 INTRODUCTION

There is a defined need to match the spatial and temporal scales of the ecological benefits provided by the hydrological cycle with basic human welfare and water resource development for societal prosperity. In both ecological and societal systems, different processes operate at different scales. Ideally, EFAs should be performed at scales which are meaningful to society, whether they relate to protecting ecological rights (Chapter 4, Section 3), servicing household water rights (Chapter 4, Section 4) or constructing water developments (Chapter 4, Section 5). While broad spatial scale studies are useful for providing first, or desktop, estimates of national or regional water resources requirements, assessments at finer spatial resolutions are necessary to ensure that the tenets of sustainability and equity envisaged by South African water resources management are addressed.

South African water law recognises the country's water resource as an integrated system, linking river system networks, wetlands, lakes and groundwater, with the hydrological cycle as its physical basis. The role of the Reserve in South African water resources management was examined in Chapter 4 of this thesis and its function in the vision for sustainable water use was highlighted. However, despite the far-sighted concept of the Reserve, there are many shortcomings to approaching the protection of water resources by focusing on aquatic and riparian systems alone (*c.f.* Chapter 4, Section 5). Many freshwater problems are not directly streamflow related (*e.g.* contaminants from land and the atmosphere), and in common with directly related streamflow problems (*e.g.* reduced flows and less predictability and seasonality as a result of societal activity) have their source far removed from the river channel. In addition, as discussed in Chapter 4, quantifying the instream flow requirements of the Ecological Reserve can be approached in two different ways, *viz.*, "how much water must be left in the ecosystem?" or "how much water can be taken out of the ecosystem?" (MacKay and Moloi, 2003; *c.f.* Chapter 4). Given the complexities of natural systems and of defining the water needs of different ecological components, the first option remains a real challenge, requiring much expertise and skill, even for its proponents. The second option, referred to by Silk *et al.* (2002) as "upside-down instream flow requirements", offers a simpler solution to the problem since it accepts that water developments will occur and yet water resource protection is still achievable, but more pertinently that societal water requirements are easier to evaluate than ecological water requirements. However, even when adopting the second option there remains one outstanding challenge: *the natural streamflow hydrograph should be known.*

While the intention of the setting aside an Ecological Reserve of water is the protection of ecological functioning for future water management options, this Chapter poses the following question: "*Is the 'Ecological Reserve' really the water resource base or is it rather a 'water store'?*" In addition, if the function of the Ecological Reserve is equated with "managing future options", then the ensuing question arises, "*Is precipitation not a better starting point as the aquatic ecosystem resource base*?" as exemplified by Falkenmark (2003) in recognition of precipitation as the basic water resource (*c.f.* Chapter 4).

Ultimately, the challenge of ecologically sustainable water resources management depends on understanding the human and societal use of the entire hydrological cycle as well as hydrological connectivity at different temporal and spatial scales (*c.f.* Chapter 4, Section 5). Proponents of integrated water resources management cite ecosystem goods and services as justification for the protection of catchment resources. However, there is a defined need for the benefits of integrated water resources management to freshwater resources to be clearly demonstrated. Consequently, there is a need for hydrological indicators which reflect hillslope processes, taking cognisance of the key hydrological processes of precipitation, infiltration, evapotranspiration, surface flow, sub-surface flow, baseflow as well as groundwater flow. These indicators are required at the scale where people interrupt the hydrological cycle for afforestation, rain-fed crops and rainwater harvesting, and where they partition hydrological functioning at the macro- and micro-environmental scales. In addition, there is a need in EFA to understand how natural, semi-natural and artificial systems respond to environmental change induced by hydrological alteration.

This Chapter introduces the Mkomazi Catchment in KwaZulu-Natal, a catchment where the mainstream river system is, to date, unregulated and water resource development is currently low. Two interlinked case studies of the Mkomazi to evaluate how the hydrological cycle delivers the ecosystem goods and services required by people and society are presented. The case studies represent applications of the concepts and findings of both Chapters 4 and 5 in this Thesis. Reference hydrological conditions at a societally-influenced, catchment-landscape hydrological response scale are prepared as a baseline for studies of how societal activities impact on the hydrological indices explaining the majority of variation in a large dataset of 74 hydrological indices of different streamflow regimes (*c.f.* Chapter 5) within the catchment, from the headwater water source zones to the accumulated mainstream recapture zones (Section 3 of this Chapter). This represents an application of Steps 1 and 2 of the proposed framework for ecologically sustainable water resources management outlined in Chapter 4, Section 5. The degree of alteration, from reference conditions, in each of the dominant hydrological indices for each of the Mkomazi Catchment reference hydrological zone types is assessed in Section 4 of this Chapter. In addition, knowledge of the degree of alteration is applied in Section 4 to assess any changes to the streamflow regimes (Steps 2 and 3 of the proposed framework). Finally, where practical, Steps 4 to 8 of the proposed framework are applied at the organisational scale of the hydronomic sub-zones described in Chapter 4, Section 5.3, in order to maximise the generation of ecosystem goods and services of the Mkomazi Catchment (Section 5 of this Chapter). However, first an overview of Mkomazi Catchment and its water development potential is required. Some of the background information relating to the Mkomazi Catchment case studies is taken from Taylor (2001).

2 BACKGROUND TO THE MKOMAZI CATCHMENT

2.1 Introduction

Meeting ecological and societal needs for freshwater presents major challenges to the ecologically sustainable water resources management of South Africa's catchment areas. Even in relatively high rainfall regions such as KwaZulu-Natal there is a need to develop ecologically sustainable water management programmes to reconcile ecological and societal water needs with the availability, patterns and timing of the water resource. The Mkomazi Catchment in KwaZulu-Natal is relatively undeveloped, with no major impoundments. Consequently, while present water use in the catchment is generally conservative and undemanding, the catchment has high potential for water development. The Mkomazi River has been identified in DWAF planning studies as the most feasible next option to augment the water supply in the neighbouring Mgeni Catchment to meet the needs of the Durban / Pietermaritzburg region (DWAF, 2004). The locations of the Mkomazi Catchment and the Mgeni Catchment, as well as the cities of Pietermaritzburg and Durban are shown in Figure 6.2.

2.2 General physiography

As narrated by Taylor *et al.* (2003) "The Mkomazi Catchment (Figure 6.3) comprises the 12 DWAF Quaternary Catchments (QCs) numbered U10A to U10M and covers an area of 4383 km^2. The catchment stretches 170 km from 3300m altitude in the northwest to sea level in the southeast and has a Mean Annual Precipitation (MAP) ranging from 1283 to 752mm (Table 6.1, Figure 6.4). The MAP is higher in the, higher altitude, upper reaches of Mkomazi Catchment (950 – 1283 mm) and consequently most of the catchment runoff is generated there (DWAF, 1998a)". Also narrated by Taylor *et al.* (2003) "The Mkomazi Catchment is characterised by steep gradients of altitude and rainfall, highly variable land uses as well as highly variable intra- and inter-annual streamflows. The annual water yield of the Mkomazi System under "present" land use conditions and consumption rates was estimated in 1998 to be 905 million m^3 (DWAF, 1998a)". Despite the variability of the streamflows, the mainstream Mkomazi River as well as most of its tributaries and its headwaters is perennial under present land use conditions.

2.3 Prior Studies and Mkomazi Catchment Modelling

In addition to the DWAF pre-feasibility study of 1997 /1998 (DWAF, 1998a) for proposed water resources developments in the Mkomazi Catchment for inter-basin transfer to the Mgeni System, an Installed Modelling System for the Mkomazi Catchment was developed at the School of Bioresources Engineering and Environmental Hydrology (BEEH), University of University of Natal (now KwaZulu-Natal) in Pietermaritzburg by this author (Taylor, 2001). In developing the Installed Modelling System for the Mkomazi Catchment, the streamflows were simulated with the *ACRU* agrohydrological modelling system (Schulze, 1995) under different land cover and development scenarios, including:

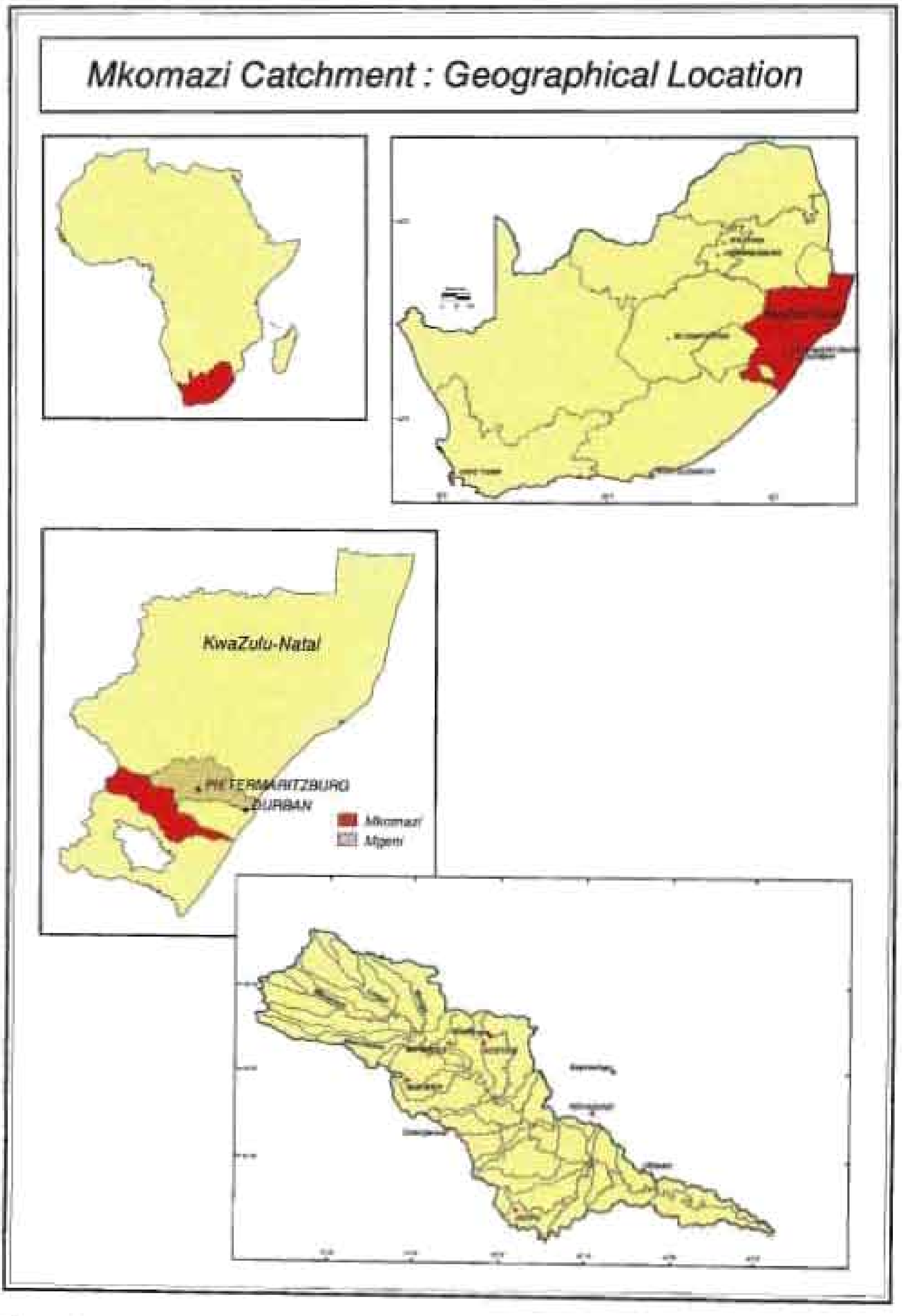


Figure 6.2 Mkomazi Catchment: Geographical location

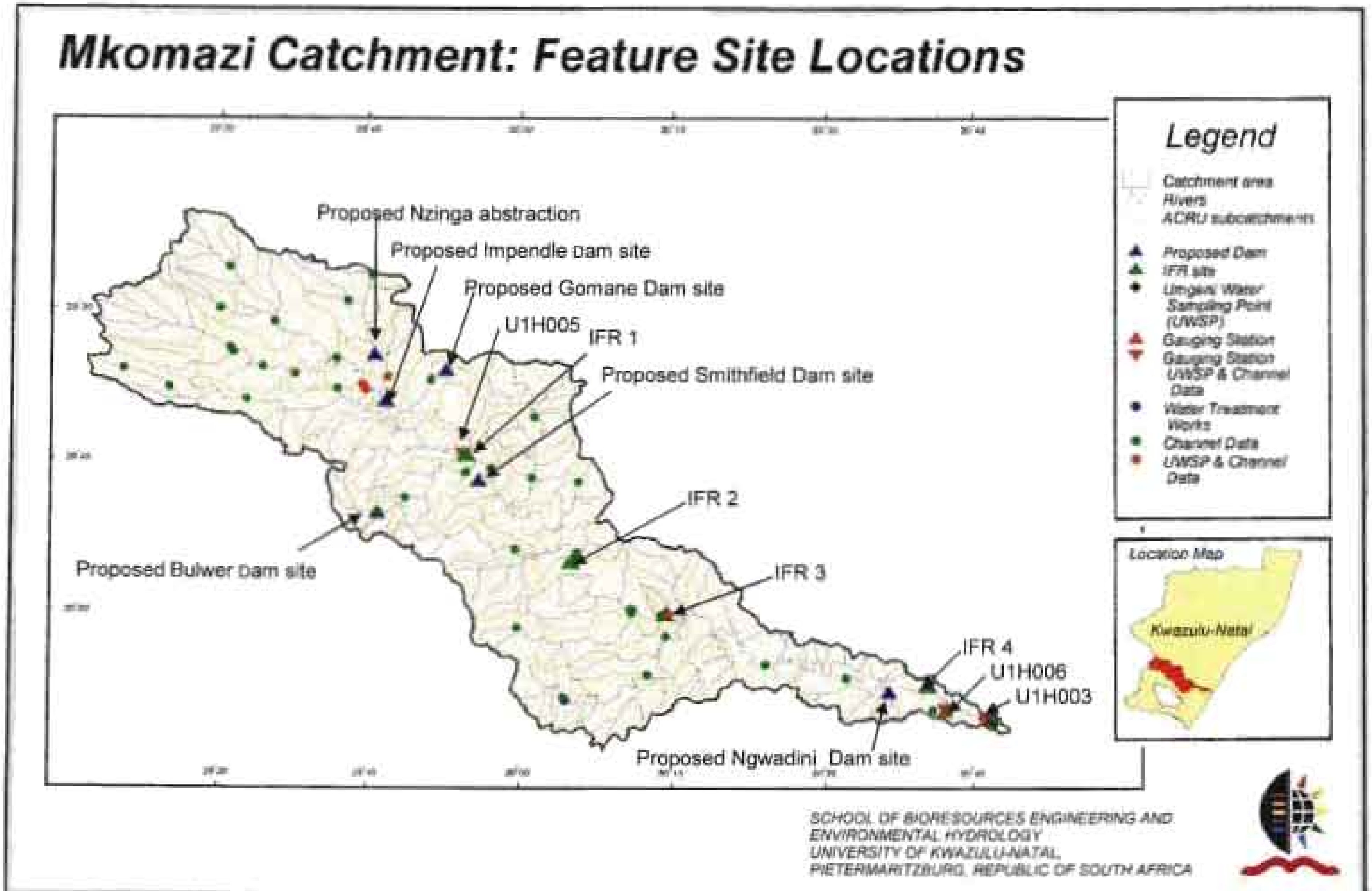


Figure 6.3 Mkomazi Catchment: Feature site locations (after Taylor *et al*., 2003)

Table 6.1 Mkomazi subcatchment information

SC or Zone No	QC No	Area (km²)	"Driver" Rainfall Station	"Driver" Rainfall Station Name	Mean Altitude (m)	Longitude (degree, decimal)	Latitude (degree, decimal)	MAP (mm)
1	U10B	162.91	0237606 W	Sani Pass (Pol)	2124	29.38	29.51	1107
2	U10B	63.32	0237606 W	Sani Pass (Pol)	1959	29.39	29.59	1095
3	U10B	141.69	0268359 W	Cyprus	1533	29.54	29.58	1044
4	U10B	29.22	0238132 W	Snowhill	1373	29.64	29.61	962
5	U10A	142.97	0268199 W	Highmoor (Bos)	2165	29.49	29.41	1283
6	U10A	57.76	0237731 A	Cobham, Himeville	2088	29.46	29.47	1179
7	U10A	208.01	0268359 W	Cyprus	1639	29.60	29.50	1095
8	U10D	47.09	0238341 W	Paulholme	1410	29.71	29.59	982
9	U10D	189.23	0268359 W	Cyprus	1851	29.71	29.49	1040
10	U10D	77.44	0238636 W	Impendle (Pol)	1643	29.79	29.56	946
11	U10C	93.12	0237606 W	Sani Pass (Pol)	2104	29.38	29.62	1068
12	U10C	32.87	0238132 W	Snowhill	1685	29.60	29.71	945
13	U10C	148.15	0238045 W	Himeville (Mag)	1568	29.60	29.69	951
14	U10D	29.97	0238341 W	Paulholme	1339	29.79	29.64	906
15	U10E	18.87	0238636 W	Impendle (Pol)	1680	29.90	29.60	935
16	U10E	70.94	0238636 W	Impendle (Pol)	1492	29.84	29.64	965
17	U10E	69.99	0268359 W	Cyprus	1678	29.71	29.72	1088
18	U10E	158.55	0238468 W	Bulwer (Tnk)	1310	29.81	29.72	1055
19	U10F	77.69	0238636 W	Impendle (Pol)	1435	29.91	29.68	997
20	U10F	55.13	0238468 W	Bulwer (Tnk)	1723	29.72	29.82	1073
21	U10F	24.12	0238293 W	Rockleigh	1554	29.77	29.85	988
22	U10F	9.06	0238468 W	Bulwer (Tnk)	1506	29.77	29.81	1089
23	U10F	145.82	0238806 W	Emerald Dale	1232	29.87	29.80	931
24	U10F	65.33	0238806 W	Emerald Dale	1155	29.92	29.81	897
25	U10G	136.00	0238636 W	Impendle (Pol)	1543	30.01	29.62	1003
26	U10G	106.48	0239133 W	Vaucluse	1351	30.02	29.70	947
27	U10G	116.18	0239133 W	Vaucluse	1172	30.02	29.78	970
28	U10H	137.54	0239133 W	Vaucluse	1090	30.01	29.82	980
29	U10H	117.43	0238806 W	Emerald Dale	1372	29.90	29.89	947
30	U10H	122.67	0238837 A	Emerald Dale	1260	29.95	29.96	860
31	U10H	76.55	0239472 W	Richmond (Tnk)	975	30.09	29.90	997
32	U10J	7.28	0239566 A	Little Harmony	780	30.10	29.91	876
33	U10J	76.07	0238837 W	Emerald Dale	1229	29.98	30.04	853
34	U10J	101.10	0210099 W	Ixopo (Pol)	855	30.11	30.05	844
35	U10J	211.44	0239138 W	Whitson	866	30.11	29.96	916
36	U10J	11.94	0239359 W	Naauwpoort	585	30.22	29.99	802
37	U10J	97.83	0239566 A	Little Harmony	761	30.23	29.96	938
38	U10L	23.74	0209795 W	Hancock Grange	840	30.28	30.12	778
39	U10L	56.04	0209825 A	Grange, Umzimkulu	725	30.13	30.10	758
40	U10K	48.36	0210136 A	Finchley, Ixopo	1161	30.02	30.13	888
41	U10K	29.74	0210099 W	Ixopo (Pol)	1091	30.06	30.15	810
42	U10K	30.07	0210099 W	Ixopo (Pol)	1026	30.09	30.16	819
43	U10K	143.62	0210099 W	Ixopo (Pol)	914	30.17	30.17	767
44	U10K	109.86	0239359 W	Naauwpoort	742	30.18	30.07	768
45	U10L	226.22	0239359 W	Naauwpoort	567	30.36	30.02	752
46	U10M	16.76	0211228 S	Esperanza	359	30.60	30.16	906
47	U10M	199.77	0210826 W	Sawoti	305	30.55	30.12	822
48	U10M	25.70	0211546 S	Illovo Mill	132	30.68	30.15	919
49	U10M	26.23	0211407 S	Renishaw	144	30.73	30.17	955
50	U10M	0.79	0211437 W	Scottburgh (Mun)	63	30.77	30.18	1023
51	U10M	2.30	0211546 S	Illovo Mill	95	30.77	30.18	1011
52	U10M	5.72	0211437 W	Scottburgh (Mun)	53	30.78	30.19	1053

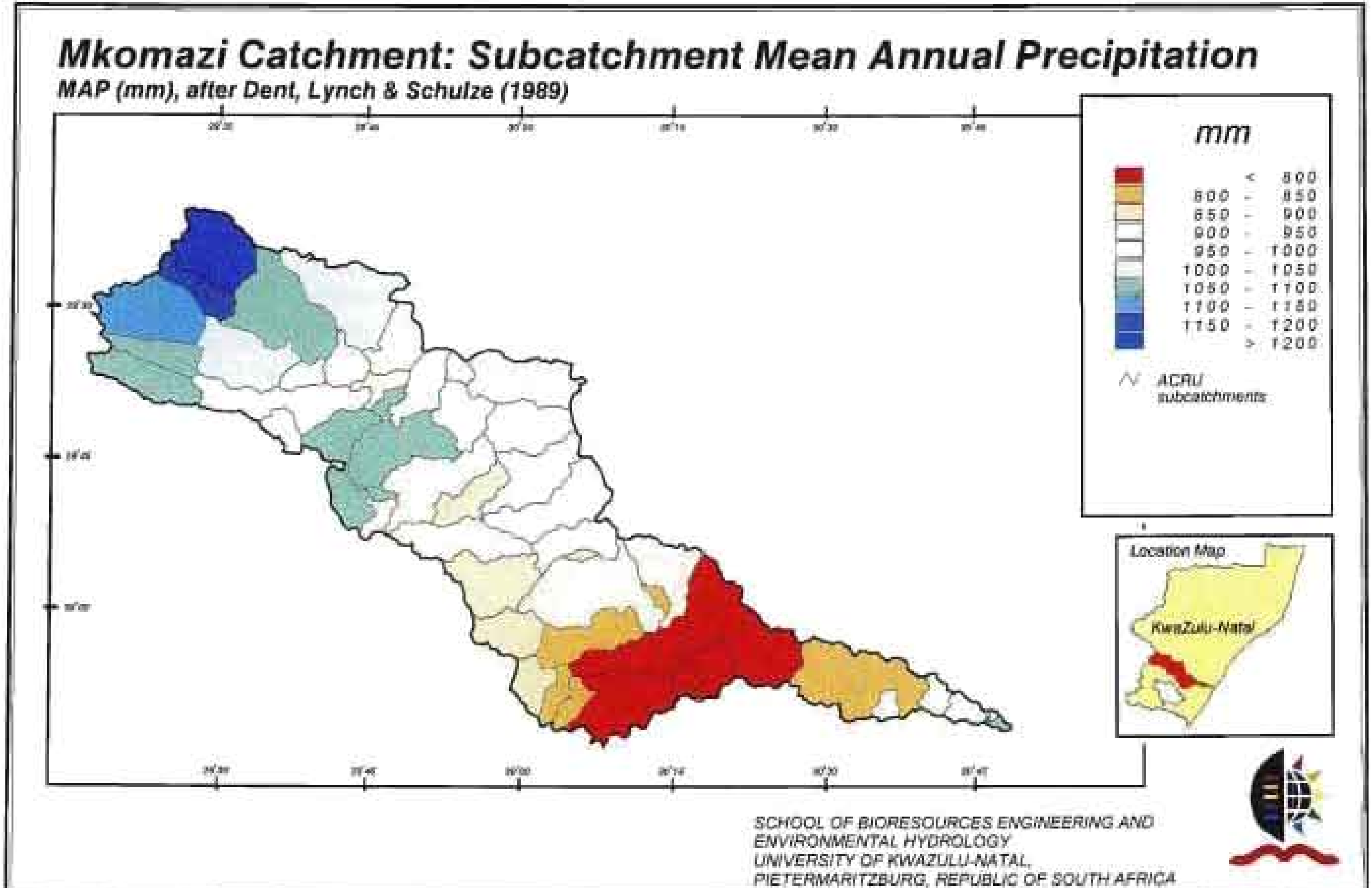


Figure 6.4 Mkomazi Catchment: Subcatchment mean annual precipitation (after Dent, Lynch & Schulze, 1989)

(a) "baseline" (or reference) land cover conditions, defined for the purposes of the study as Acocks' Veld Types (Acocks, 1988). The distribution of Acocks' Veld Types for the Mkomazi Catchment is shown in Figure 6.5,

(b) present land use conditions, defined in accordance with Thompson's (1996) land classification and the interpretation of the CSIR's 1996 LANDSAT TM image for South Africa. The distribution of present land cover and land use in the Mkomazi Catchment, as identified by the LANDSAT TM image, is shown in Figure 6.6. These land use conditions were ground truthed by the School of Bioresources Engineering and Environmental Hydrology (BEEH) for consistency in 1998. During numerous field trips to the Mkomazi Catchment, alien invasive tree species were detected throughout the catchment, particularly in riparian areas. Therefore, this land use was included in scenarios involving "present land use", by allocating a 30 metre buffer strip in which alien tree species grow along the main tributaries and river network.

(c) present land use, but including potential future impoundments, in accordance with DWAF's proposed Mgeni / Mkomazi Transfer Scheme.

ACRU is a daily time step, physical-conceptual model operating on a multi-layer soil water budget. Although it is a multi-purpose model, its major use is the output of times series of daily values of streamflow. Records of streamflows at a daily time step are essential for assessing the main characteristics of the streamflow regime (*c.f.* Chapter 5). The *ACRU* model is structured to be hydrologically sensitive to catchment land uses and changes thereof, and is consequently appropriate for defining reference conditions (*c.f.* Section 3 of this Chapter) of streamflow regimes and for the assessment of change thereof (*c.f.* Section 4 of this Chapter).

At the request of Umgeni Water, the bulk water supplier to the region and thus, the major beneficiary of the Installed Modelling System, the Mkomazi Catchment was configured to represent 52 major inter-linked subcatchments, based essentially on a division of the 12 DWAF Quaternary Catchments. The 52 subcatchments (*c.f.* Table 6.1) were delineated from 1:50 000 topographical map sheets for the Mkomazi Catchment, supplied by the Surveyor General (DSLI, 1997). The main objectives of the delineation were to represent the different land use and management practices as discrete hydrological units, as well as considering proposed development concerns within the catchment. Thus, the 52 units can

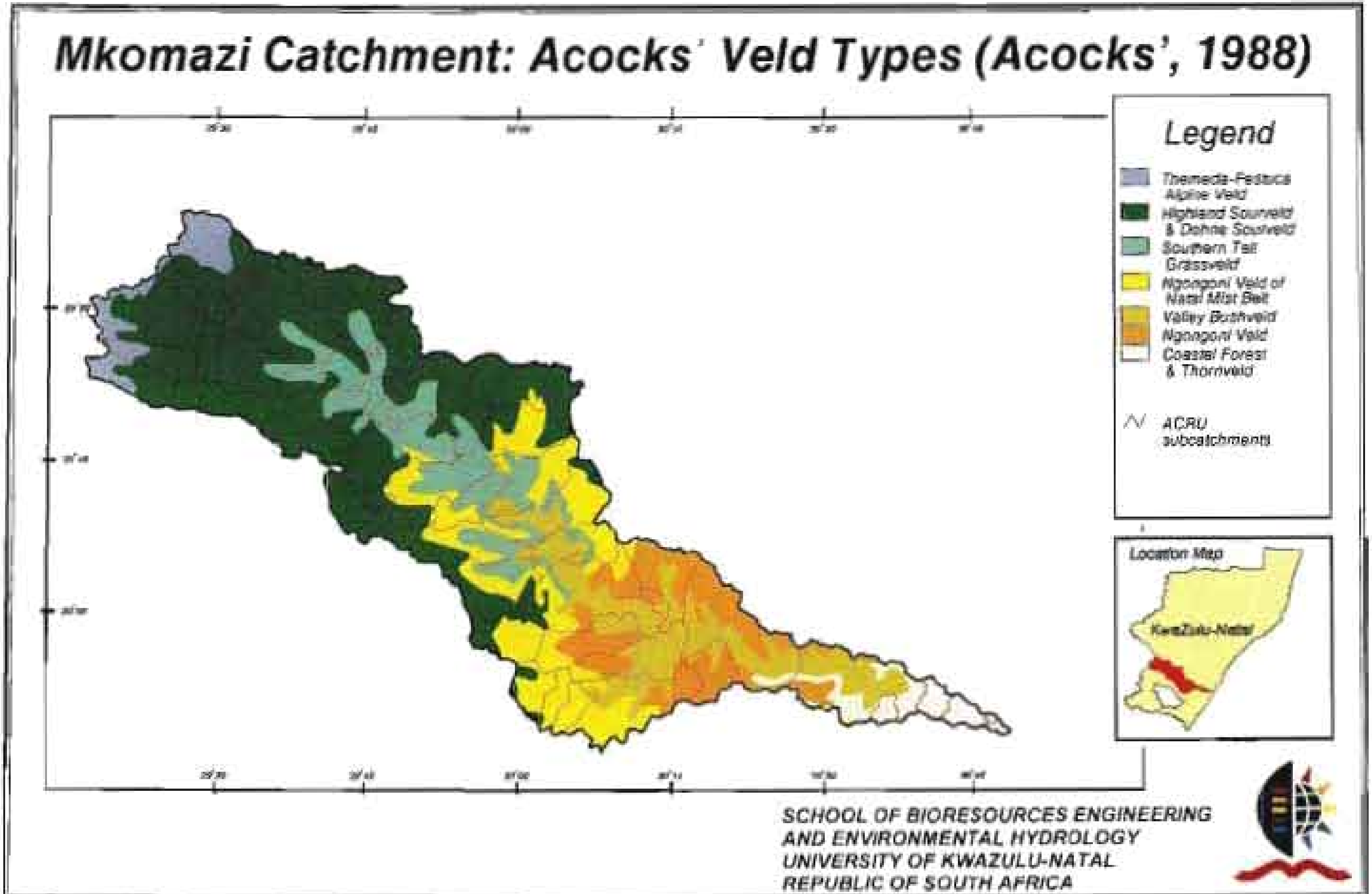


Figure 6.5 Mkomazi Catchment: Acocks' Veld Types (Acocks', 1988)

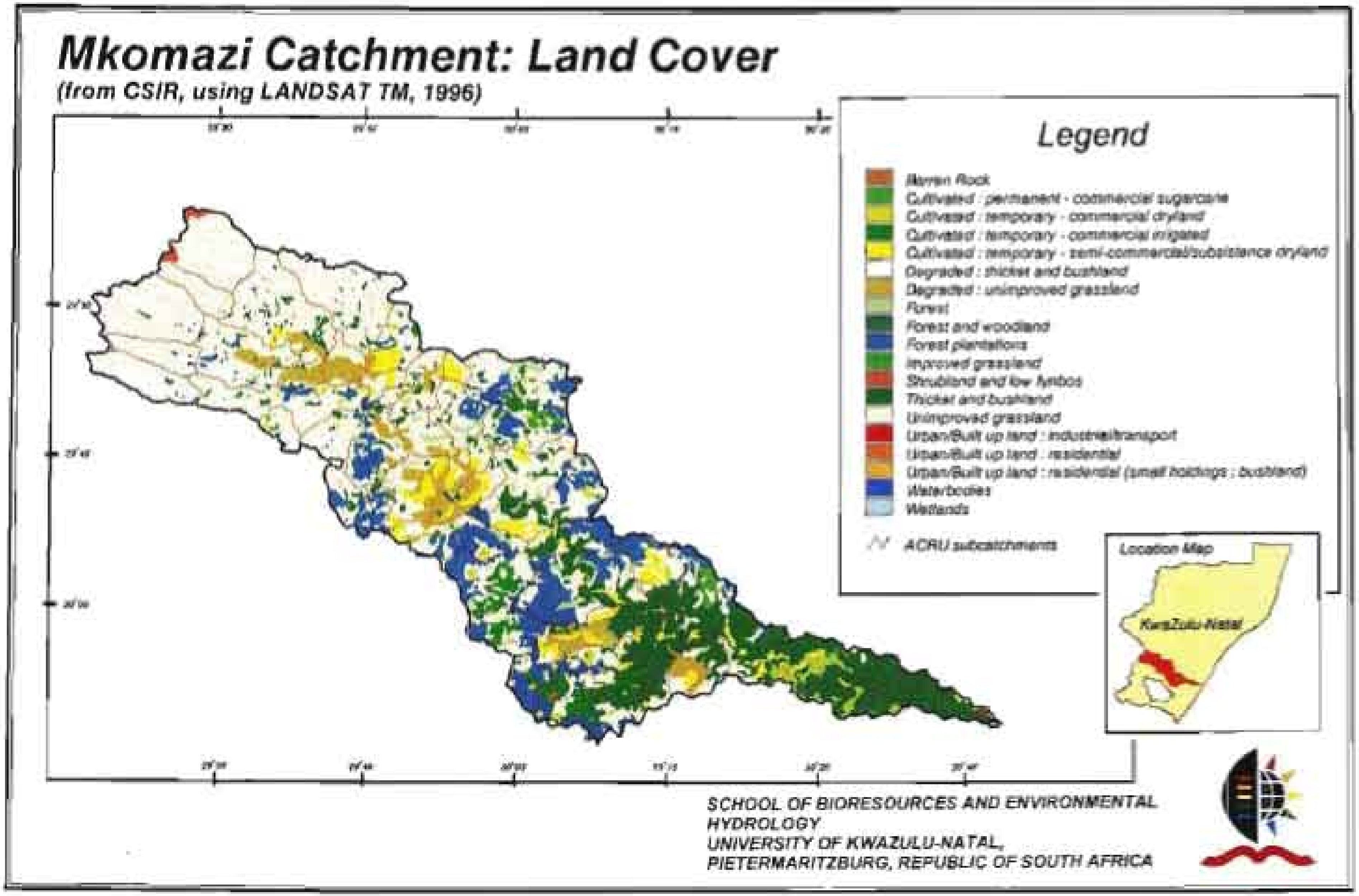


Figure 6.6 Mkomazi Catchment: Land cover (from CSIR, LANDSAT TM, 1996)

be considered to be *hydronomic zones* (*c.f.* Chapter 4, Section 5.3). The concept of hydronomic zones was first described by Molden *et al.* (2001), to illustrate different water management practices, depending on whether or not the water outflow from a zone is recoverable for downstream use (*c.f.* Chapter 2, Section 4.5 and Chapter 4, Section 5). Notwithstanding this classification by Molden *et al.* (2001), Figure 6.7 illustrates the criteria used, as requested by Umgeni Water, for discretising the individual subcatchments of the Mkomazi Catchment. The final configuration specifically includes:

(a) 2 major proposed dam sites on the Mkomazi river (Impendle and Smithfield)
(b) 4 proposed rural supply abstraction developments
(c) 4 Instream Flow Requirement (IFR) sites to assess the ecological water needs, all on the Mkomazi river
(d) Umgeni Water's sampling sites for water quality
(e) DWAF's streamflow gauging stations
(f) the waste water treatment works at Ixopo
(g) 3 sediment test sites which were used in collaborative research with the University of Florence in Italy, but not reported in this thesis
(h) the subdivision of the Drakensberg region on physiographic grounds because of the steep altitudinal, hence rainfall and consequent runoff gradients found there and,
(i) distinction of the different land uses which impact on hydrological responses.

Thus the main benefit of the Installed Modelling System for the Mkomazi is that the hydrological dynamics of the Mkomazi Catchment may be represented on a daily time step at a sub-Quaternary Catchment scale for different development scenarios.

2.4 Societal Needs for Freshwater

The current societal needs for freshwater in the Mkomazi Catchment are low for various physiographical and socio-economic reasons. This sub-section addresses the two main societal uses of water within the catchment, *viz.*, the Basic Human Needs Reserve and agri-business needs.

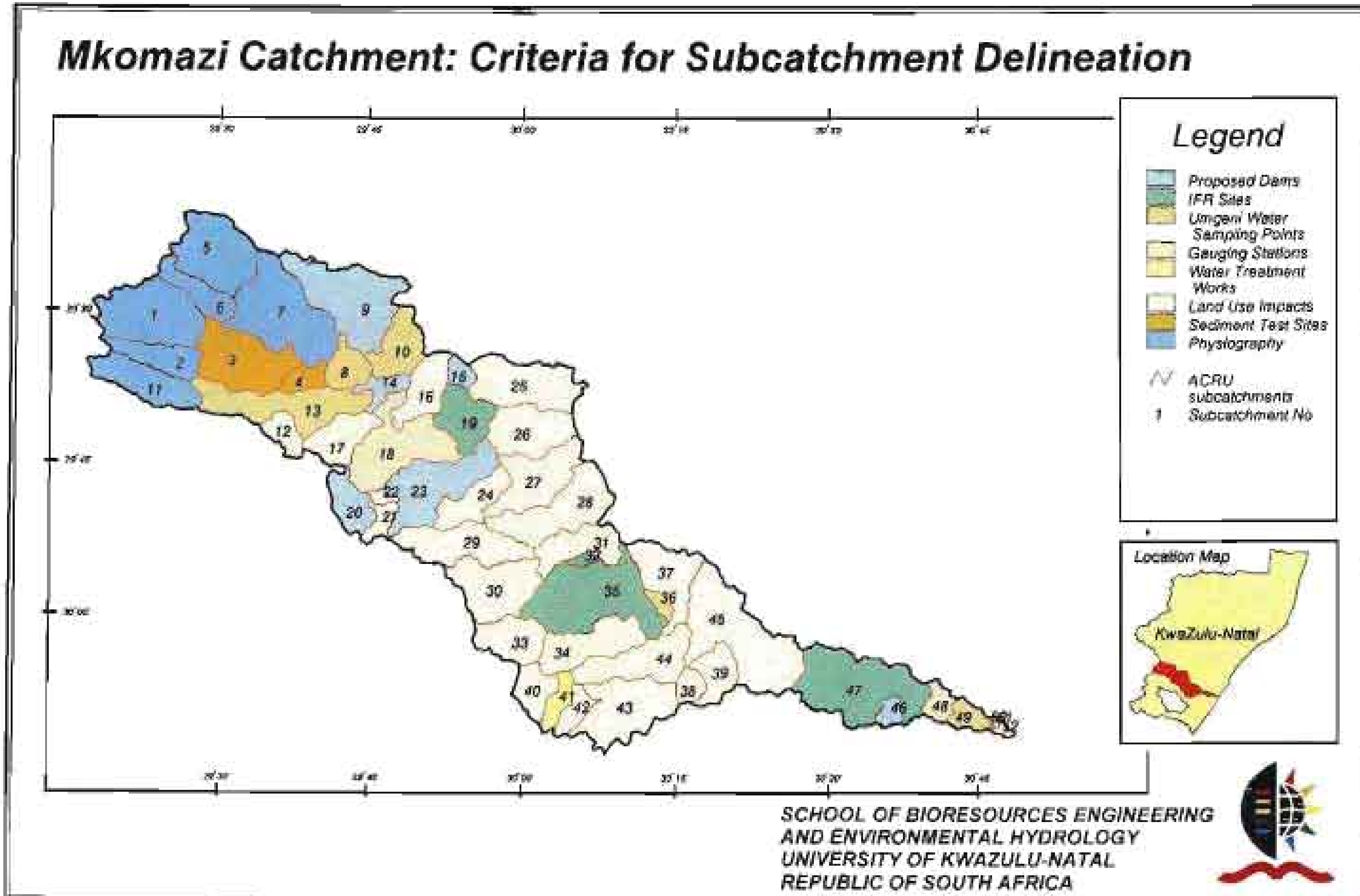


Figure 6.7 Mkomazi Catchment: Criteria for subcatchment or "hydronomic zone" delineation (Taylor, 2001)

2.4.1 The Basic Human Needs Reserve

The Mkomazi Catchment is sparsely populated, with no major urban areas except for the coastal town of Umkomaas. The work towards the Installed Modelling System for the Mkomazi by Taylor in 2001 (hereafter referred to as the Mkomazi 2001 Study) identified the current domestic demand (defined therein as (a) the BHNR, or "household right" (*c.f.* Chapter 4, Section 4), calculated from a consumption scenario of 30 - 150 litres per person per day for rural and urban populations respectively, and (b) animal husbandry and community / small-scale subsistence farming) was less than 1% of total catchment annual streamflows. As the vision for the BHNR is that the water allocation is to be provided within 200 metres of the dwelling (NWA, 1998), it is anticipated that a major management objective for a desired "household reserve category" (*c.f.* Chapter 4, Section 4) will result in the BHNR being sourced from stored water and piped to this proximity of settlement. Consequently, the BHNR will be sourced from high flows rather than baseflows and there may not be any substantial impacts on baseflows as a result of meeting the BHNR, or *vice versa*. Moreover, while the majority of the existing population currently has inadequate access to potable water and many rural communities rely on direct river abstractions for household, livestock and subsistence water use, four rural supply abstraction developments (shown as Nzinga, Bulwer, Gomane and Ngwadini on Figure 6.3) have been proposed to alleviate this need (*c.f.* Section 2.4.4 of this Chapter). Accordingly, meeting the BHNR of the Mkomazi Catchment is not anticipated to generate any incompatibilities with ecological water needs, even if high allocations to the "household reserve category" (*c.f.* Chapter 4, Section 4) are realised (Taylor, 2001).

2.4.2 Agri-business needs for freshwater

As narrated by Taylor *et al.* (2003), the Mkomazi Catchment "supports commercial forestry, extensive agriculture (principally livestock grazing and sugarcane), intensive agriculture (citrus and vegetables) and subsistence agriculture". The catchment is relatively undeveloped and neither agriculture nor forestry is a major land use throughout, although they are of local importance (DWAF, 2001; *c.f.* Figure 6.6). A large extent (70%) of the catchment is natural, comprising unimproved grassland, bush land and indigenous forest, based on Thompson's (1996) land cover classification and the interpretation of the 1996 LANDSAT TM image for South Africa (*c.f.* Figure 6.3).

The MAP of the upper catchment ranges from 1000 to > 1200 mm, (Figure 6.4). Consequently, farming practices in the upper catchment do not usually require supplementary water. However, the steep gradients, which exacerbate the difficulties of leading water from the river valley, have contributed to the major land uses in the upper catchment being those of ranching, especially in those areas formerly part of the KwaZulu homeland, and forestry. The main problems relating to ranching are those associated with overgrazing, which contributes significantly to disturbed soil conditions, soil transport and ultimately sedimentation of river channels and impoundments. Exotic, commercial tree plantations are the focus of a whole gamut of water related issues, ranging from streamflow reduction through interception and transpiration losses, particularly in times of low flows, to water quality problems which include excess acidity of headwaters and exacerbation of sedimentation at harvesting.

The economic and socio-political structure of the upper Mkomazi also excludes any crop husbandry other than low input commodities. As a result, yields per hectare are low. The area generally has poor infrastructure, with rural populations having severely limited access to amenities and agricultural technologies. The inequitable economic structure of the Mkomazi Catchment and problems of access to water become stronger downstream. Because of relatively high rainfall (950–1000 mm), commercial forestry is a significant land use in the middle Mkomazi, particularly around Richmond (*c.f.* Figures 6.2 and 6.6). The more intensive cultivation of both sugarcane and horticulture is also practised in this region, despite growing conditions being marginal as a result of the incidence of frost.

The lower Mkomazi Catchment is more physiologically amenable to intensive crop farming. The economic driving forces of the lower part of the catchment are much greater than in the upper and middle catchment and consequently there is greater concentration of farm dams, together with high input crops producing high yields per hectare. Paradoxically, these practices are most prevalent in that part of the Mkomazi Catchment that receives least rainfall (MAP less than 850 mm, Figure 6.4). Farmers in this region have compensated for this by storing excess streamflows in numerous farm dams and irrigating heavily from them. However, the consequence of the irrigation is that river flows in the affected catchments have all but ceased and there is no significant river flow in the tributary river channels downstream of the town of Ixopo.

With the exception of the Sappi Saicorr paper mill at the estuary, the coastal town of Umkomaas and Ixopo further inland, the catchment contains no major towns or industry. Consequently, water quality in the Mkomazi Catchment is generally high. However, there are some erosion problems in the upper Mkomazi, particularly in the Impendle subsistence agricultural area where, in addition, livestock and human populations impact on water quality through faecal contamination (DWAF, 2001). There are similar concerns about water quality in the lower tributaries and mainstream Mkomazi River, particularly where rural communities rely on untreated river water for household use (DWAF, 2001).

2.4.3 Hydrological stress for water users

Surface flows are generally available throughout the Mkomazi Catchment, with the result that minimal abstraction of groundwater is required for societal needs (DWAF, 2004). However, despite the currently low societal need for freshwater, there is hydrological stress for water users in low flow years (defined in the Mkomazi 2001 Study as "hydrological years in which the annual streamflow depth is not exceeded in four years out of five"), particularly in the low flow season (defined in the Mkomazi 2001 Study as "the period of low flow within a year representing the six consecutive months of lowest combined flow"). The low flow season of the Mkomazi Catchment was identified as June through November by Taylor (2001). In low flow years, the Mkomazi Catchment generally experiences at least a 20% reduction in annual streamflows from a year with median flows, under conditions of present land use (Taylor, 2001). The upper Mkomazi, mid Mkomazi Valley and the lower Mkomazi experience reductions in the range of 20% - 40%, with areas of the upper catchment being as impacted as those in the lower catchment. Where commercial forestry and irrigation are both practised there are reductions in streamflows exceeding 45% relative to those in a year of median flows (Taylor, 2001).

Water users repeatedly identify the low flow season as being the most critical period of the streamflow regime. The areas of the Mkomazi Catchment which are most stressed are those where both afforestation and irrigation are practised, with the Donnybrook / Ixopo region experiencing reductions of nearly 95% during the low flow season, even in a year of median flows, compared to the seasonal low flows under "reference" land cover conditions (Taylor, 2001). The impacts of present land use during the low flow season in this region are such that there is virtually no available water according to the *ACRU* model simulations except for seepage releases from farm dams (Taylor, 2001).

2.4.4 Proposed impoundment and inter-basin transfer

Water shortages for societal needs during low flow periods in the Mkomazi Catchment are considered by DWAF (DWAF, 2004) to be the result of insufficient storage on the mainstream Mkomazi River. Ironically, these water shortages are expected to be exacerbated by the implementation of the Reserve (*c.f.* Chapter 4, Section 3) to meet ecological needs for freshwater (DWAF, 2004). While there are presently no major impoundments on the Mkomazi River, the catchment has an historical mean yield of 905 million m^3 per annum (DWAF, 1998a) and there are plans to utilise this water resource. In September 1997, there were six dam sites identified for the Mkomazi Catchment. Three of these sites (Gomane, Nzinga and Bulwer, Figure 6.3) in the upper catchment have been identified as potential dam sites planned to alleviate rural water supply shortages and one site (Ngwadini, Figure 6.3) in the lower catchment has been identified for a proposed off-channel storage dam for the surrounding agricultural community. The remaining two sites (Smithfield and Impendle, Figure 6.3) are the focus of proposed inter-basin transfer, in a scheme known as the Mkomazi-Mgeni Transfer Scheme (MMTS). The upstream contributing areas of each of the six dam sites are provided in Table 6.2.

Table 6.2 Contributing areas of the proposed impoundments for the Mkomazi Catchment

Dam	Longitude (degrees, decimal)	Latitude (degrees, decimal)	Upstream Area (km^2)
Impendle	29.78	29.65	1423.74
Smithfield	29.93	29.77	2053.57
Gomane	29.88	29.60	18.87
Nzinga Abstraction	29.76	29.58	189.23
Ngwadini	30.61	30.14	16.76
Bulwer	29.76	29.84	55.13

The first phase of the MMTS will involve the construction of the proposed Smithfield Dam on the Mkomazi River. DWAF plan that initially 5.6 $m^3.s^{-1}$ will be a transferred (with a peak transfer capacity of 7.0 $m^3.s^{-1}$) from the proposed Smithfield Dam to the Mgeni Catchment via a pump-station-shaft-tunnel to an existing dam near Baynesfield (*cf.* Figure 6.2). It is anticipated that the abstraction of water from the Mkomazi Catchment to augment the Mgeni supply system will impact on those downstream abiotic characteristics of the Mkomazi River (hydrology, geomorphology, chemistry, temperature) as well as on the responses of the ecosystem components (fish, riparian vegetation and aquatic invertebrates).

2.5 Ecological Needs for Freshwater

A principal issue of concern relating to impacts of the MMTS on the catchment hydrological dynamics is that the provision of the ecological component of the Reserve will have a significant impact on the yield and operating rules of both proposed dams. A DWAF reconnaissance level basin study to determine the present and future water needs within the Mkomazi Catchment (DWAF, 1998a) identified that ecological water needs, in the form of instream flow requirements (IFRs) are a dominant water resource consideration, requiring approximately 30% of the Mean Annual Runoff (MAR). Thus, the determination, and fulfilment, of the Ecological Reserve for the Mkomazi Catchment is a major issue of concern.

Much preparatory work by the Institute for Water Research at the University of Rhodes in Grahamstown has already been conducted to determine the Instream Flow Requirements (IFRs) for the Mkomazi River (DWAF, 1998b). The output of this work is a set of Building Block Methodology (BBM) tables (*c.f.* Chapter 4, Section 3.2.2 in which the high flow and baseflow values required to meet ecological needs for each month of the year are specified for both maintenance and drought years. The four IFR sites selected by the BBM workshop process (*c.f.* Chapter 4, Section 3) are shown in Figure 6.3. The locations of all the IFR sites are downstream of either one, or both, of the proposed Smithfield and Impendle Dam sites (*c.f.* Figure 6.3). The respective upstream contributing areas for the four IFR sites are provided in Table 6.3. The present ecological state of IFR Sites 1, 2, 3 and 4 was assessed by the BBM workshop process as being C/B, C/B, D/C and C respectively (DWAF, 1998b). While the Reserve determination has still to be presented to the stakeholders for consultation and assessment of various flow-related scenarios, the BBM workshop process recommended a desired future Ecological Reserve Category

(ERC) of B (slight modification from natural state, or "good", as described in Table 4.2 of Chapter 4) as a reflection of its relatively undeveloped state. The general consensus at Mkomazi IFR workshops regarding the Mkomazi estuarine flow requirement (EFR) is that if the IFRs are satisfied, then the EFR will also be satisfied.

Table 6.3 Mkomazi Instream Flow Requirements sites: Contributing areas

Instream Flow Requirements Site	Locality	Longitude (degrees, decimal)	Latitude (degrees, decimal)	Upstream Area (km^2)
IFR1	Lundy's Hill	29.91	29.75	1819.43
IFR2	Hela Hela	30.09	29.92	2939.01
IFR3	Josephine's Bridge	30.23	30.02	3327.62
IFR4	Mfume	30.67	30.12	4321.56

2.6 Preliminary Catchment Management Plans for the Mkomazi Catchment

The Mkomazi Catchment is largely undeveloped and the potential to store the high flows of the river system, together with the proximity of the catchment to the denser distribution of population in the Mgeni system, provides the impetus for impoundment. Indeed, it has been proposed in the Department of Water Affairs and Forestry's National Water Resource Strategy that water development of the Mkomazi River should be reserved for that purpose (DWAF, 2004). This proposal could compromise the desired future ERC for the river set at Category B by the BBM workshop process described in Section 2.5. This is all the more pressing since the BBM biophysical assessments purposefully do not take cognisance of any subsequent water resource development.

There is an increasing need for shared appreciation among all interested parties in the assessment of how much water rivers need to sustain the integrity of aquatic and riparian ecosystems to meet both ecological and societal needs (van Wyk *et al*, 2006). A preliminary study of the degree of alteration of the mainstream Mkomazi River system was performed by this author in recognition of the potential incompatibilities between ecological and societal needs for freshwater, particularly after the construction of the MMTS. The study applied the Range of Variability Approach (RVA; Richter *et al.*, 1997), to statistically analyse the streamflow regime at two of the four IFR sites on the Mkomazi River described in Section 2.5 under pre-development and post-development conditions.

The RVA is an application of the Indicators of Hydrological Alteration (Richter *et al.*, 1996) representing those ecologically relevant hydrological indices described in Chapter 5. The major benefit of the RVA is that, in the absence of extensive biological data or ecological expertise, *preliminary management targets* designed to protect natural aquatic biodiversity and aquatic ecosystems can be set using either historical hydrological data or simulated hydrological information (Richter *et al.*, 1997).

The preparation, methods and results of the preliminary study of the degree of alteration of the mainstream Mkomazi River system have been published in the African Journal of Aquatic Science (Taylor *et al.*, 2003). A copy of the paper "Application of the Indicators of Hydrological Alteration method to the Mkomazi River, KwaZulu-Natal, South Africa" is reproduced in Appendix 6A of this Chapter.

2.7 Catchment Potential: The Case for New Approaches

The socio-economic development of the Mkomazi Catchment is heavily reliant on agriculture, particularly along its tributaries. Therefore, it makes economic, as well as hydrological, sense to utilise the entire catchment hydrological dynamics to utmost potential. Determination of the different impacts of different agricultural practices on the quality and availability of water resources is consequently of primary concern to catchment stakeholders. However, in common with many parts of South Africa, there are disparities within the socio-economic structure of the Mkomazi Catchment in terms of income, education and access to services (DWAF, 2001). In particular, rural populations in the Mkomazi Catchment experience "water poverty" as a result of inadequate supplies of good quality water, either from climatic variability and/or lack of access, as well as vulnerable "subsistence" agricultural practices (DWAF, 2001). The benefits of the hydrological regime, through the generation of ecosystem goods and services, should be available to fully meet the needs of the donor catchment before consideration is given to transferring water to other catchments.

2.8 Summary

Despite perennial flows, there are potentially conflicting water issues within the Mkomazi Catchment. Conflicts of water use and incompatibilities between ecological and societal freshwater needs are perceived to emanate from a variety of different agricultural practices, particularly in low flow years and in the low flow season. Several moderately-sized impoundments are planned to meet the need for increased quality of the BHNR, or the "household right", of the rural people of the Mkomazi. The major impoundment of the Mkomazi River (*i.e.* the proposed Smithfield Dam) and the MMTS are expected to alleviate the societal needs for freshwater of the neighbouring Mgeni Catchment. However, notwithstanding the likely alteration of the streamflow regime incurred by this impoundment and the proposed inter-basin transfer, there are concerns that maintaining a high Ecological Reserve Category (ERC) in the mid to lower Mkomazi Catchment will, in addition, have a significant impact on the yield and operating rules of the proposed Smithfield Dam.

*   *   *   *

The remainder of this Chapter focuses on maximising the generation of ecosystem goods and services for future options of the Mkomazi Catchment, KwaZulu-Natal. Thus the Installed Modelling System for the Mkomazi developed in 2001 by this author is revisited to investigate new approaches to match the spatial and temporal scales of the environmental benefits provided by the hydrological regime, with basic human welfare and water resource development for societal prosperity. Section 3 of this Chapter initiates this quest by re-assessing the natural streamflow hydrograph, and at a sub-quaternary scale. The Case Study which follows focuses on the tributaries of the main Mkomazi River, since this is the spatial scale at which there is greatest hydrological stress for water users. Sections 3, 4 and 5 as well as Appendix 6A of this Chapter, are structured to be stand-alone sections in the Case Study. Consequently, there is some repetition of the description of the Mkomazi Catchment, the Indicators of Hydrological Alteration and the Range of Variability Approach provided in the Paper "Application of the Indicators of Hydrological Alteration method to the Mkomazi River, KwaZulu-Natal, South Africa" referred to in Section 2.6 and the following Sections. These details have been included for clarity.

3 REFERENCE HYDROLOGICAL CONDITIONS FOR THE MKOMAZI CATCHMENT

3.1 Introduction

Stakeholders require information on ecosystem functioning under natural, or reference, conditions in order to ascertain the likely ecosystem response to alterations of the streamflow regime as a result of any societal water use and water development. The main aim of this Section is to investigate the reference hydrological conditions of the 52 subcatchments, or "hydronomic zones", of the Mkomazi catchment described in Section 2 of this Chapter, so that meaningful scenarios of water developments can be constructed. Thus, this Section addresses Steps 1 and 2, *viz.*, "Define reference hydrological zone and conditions" and "Assess ecological flow requirements", of an application of the proposed framework for ecologically sustainable water resources management outlined in Chapter 4 of this thesis (*c.f.* Figure 4.9) for the Mkomazi Catchment.

The study comprising Chapter 5 of this thesis indicated that hydrological indices of intra- and inter-annual variability as well as overall variability, predictability and seasonality can be used to describe the dominant patterns of variance for streams in varying climatic and geological conditions within South Africa. Even at a broad spatial scale, a subset of indices of high information which explain the dominant patterns of hydrological variability for different stream types (and see Olden and Poff, 2003) can be used as an ecological resource in environmental flow assessments. In this Section the methods described in the study in Chapter 5 are applied at the spatial scale of the 52 hydronomic zones of the Mkomazi, in order to select subsets of optimal indices for reference hydrological conditions, based on the hydro-climatic area in which each zone is located.

3.2 The Hydrological Record of Reference Conditions for the Mkomazi

There are two operational DWAF gauging stations (U1H005 and U1H006) on the Mkomazi River with relatively long records of observed daily averaged streamflows. Automated recording of streamflows commenced in 1960 and 1962 for U1H005 and U1H006 respectively. The locations of these stations are indicated on Figure 6.3, which shows that U1H005 records streamflows generated in the upper Mkomazi, whilst U1H006 records the accumulated streamflows of virtually the entire catchment. As stated in Section 2, there are no major impoundments on the Mkomazi River and the catchment is relatively unimpacted, particularly in the upper reaches. Notwithstanding the impacts of present land use in the low flow season in the lower part of the catchment (*c.f.* Section 2), both these stations have been considered to be "recording reasonably natural flow" and the streamflows recorded at these stations have been applied in broad scale and regional scale studies of reference hydrological conditions for South African river systems. Data from U1H005 were used in the study comprising Chapter 5 of this thesis whereas data from U1H006 were used in the study by Joubert and Hurley in 1994 (Joubert and Hurley, 1994) and by Hughes in 1995 (Hughes, 1997).

Together, the relatively long existence and locations of these stations should be advantageous to any study involving the information in the records. However, there are problems regarding the streamflow record at U1H006. These problems are well-known and documented; for long periods in the record the DWAF recording gauge at this site has a low discharge table limit and therefore produces unreliable high flow recordings (DWAF, 1998b). Thus information regarding the magnitude, duration, timing, frequency and rate of change (rising and falling river levels) associated with high streamflow events are not adequately represented in the DWAF record for U1H006. For this reason the DWAF gauging station U1H006 was omitted from the study in Chapter 5 of this thesis.

3.3 Supplementing the Hydrological Record

The unreliability of the streamflow record at U1H006 impedes any study to determine reference hydrological conditions in the lower Mkomazi Catchment. While the streamflow record at U1H005, located considerably upstream, could be extrapolated to estimate the downstream streamflow regime, this situation is tenuous for any studies, particularly ecological studies, which require reliable information reflecting the dynamic nature of the catchment hydrological processes at relatively fine temporal and spatial scales. Where the observed record of daily streamflows is inadequate, either as a result of insufficient length, or of poor quality, simulating daily streamflows using an appropriate hydrological model provides an invaluable substitute, particularly where the simulation can be used to generate a daily time series of streamflows which represents the natural streamflow range. In addition, hydrological simulation modelling has much to offer in the assessment of human influences on the streamflow regime and for formulating ecologically sustainable water management approaches. Computerised hydrological modelling has become an essential tool for understanding the nature, degree and location of human influences on natural flow regimes, since daily streamflow hydrographs resulting from a variety of human activities can be generated with relative speed and ease (Richter *et al.*, 2003). Comparing such daily streamflow hydrographs, either visually or statistically, can allow negotiators to assess potential conflicts or incompatibilities between the streamflows required for ecosystem protection and human altered streamflows (Richter *et al.*, 2003).

3.4 Methods Applied to Supplement the Hydrological Record of the Mkomazi Catchment

The time series of daily streamflows generated with the *ACRU* model for the Mkomazi 2001 Study, as described in Section 2.3, were revisited for the present study.

Verification of the *ACRU* simulated streamflows of present land use (*c.f.* Section 2.3 and Figure 6.6) with the corresponding DWAF record of streamflows at the site U1H005 was performed for the Mkomazi 2001 Study. The verification study focused on streamflows simulated under present land use, since it was to be expected that the DWAF record at U1H005 would also include any impacts of the human altered landscape, such as they are. Nonetheless, since "the generation of streamflows from present land use in the upper Mkomazi shows very little change from that of baseline conditions" (Taylor, 2001), it is highly likely that had the verification been performed with the simulated streamflows generated under baseline conditions using Acocks' Veld Types as reference land cover conditions (Acocks, 1988), *c.f.* Figure 6.5), the results would have been very similar. The reason for this is that unimproved grassland (Thompson, 1996), based on Acocks' (Acocks, 1998) term for unaltered indigenous grassland, represents more than 80% of the land cover upstream of U1H005 (Taylor *et al.*, 2003; *c.f.* Figure 6.6).

Verification of the *ACRU* simulations of streamflows, generated under conditions of present land use at the site of U1H005, from 1 October 1960 to 30 September 1996, with the corresponding time series extracted from the observed DWAF record, indicated that high confidence could be assigned to the monthly totals of daily streamflows simulated by the model ($r^2 = 0.89$; Taylor, 2001). Despite the problems associated with the DWAF record at U1H006, verification of the *ACRU* simulations of streamflows generated under conditions of present land use was also performed for the Mkomazi 2001 Study. Bearing in mind the anomalies resulting from the overtopping of the gauging plate at U1H006, the portion of the hydrograph representing the baseflow regime in the time series of monthly totals of daily flows compared well between the simulated and observed records (Taylor, 2001).

Nonetheless, the Mkomazi 2001 Study did not perform any verification of the *ACRU* simulated streamflows with the observed streamflow record at a finer time step than "monthly totals" of daily streamflows for either site. Moreover, the objectives of this Study require that a high level of confidence in the *ACRU* simulations of streamflows is established, at the time step appropriate to each of the 35 intra-annual indices described in Chapter 5, Section 3. Thus, it was necessary to verify the time series of each of the 35 intra-annual indices extracted from the *ACRU* record of simulated daily streamflows at the site of U1H005 with those extracted from the DWAF record of observed flows. For convenience, Table 5.2 of Chapter 5 is included as Appendix 6B to this Chapter, since the indices described therein (including the 35 intra-annual indices) are referred to extensively in this and the following Sections.

3.4.1 Verifying the intra-annual hydrological indices extracted from streamflows simulated with the *ACRU* model

The time span of the two records differed, with the *ACRU* time series spanning the hydrological years 1945 to 1995 and the DWAF time series spanning the hydrological years 1960 to 2000.

In the study in Chapter 5 the screening procedure and tests for absence of linear trend, for stability of the median and of the dispersion described in Appendix 5B were applied to the 35 intra-annual indices extracted for the period 1 October 1965 to 30 September 2001 (*i.e.* 36 years of record) from the DWAF record at gauging station U1H005. The results of the screening procedure and tests showed that the longest period missing from the DWAF record over the 36 year time span is only 85 days, whereas the "acceptability" (*c.f.* Appendix 5B; Table 5B2) of time series of the indices is very good with a combined score of 134 (out of a possible 140). The results of each of the tests for the times series from

1 October 1965 to 30 September 2001 (*i.e.* 36 years of record) from the DWAF record at gauging station U1H005 are referred to and provided in Appendix 5C.

In the study comprising this Chapter, the same screening procedure and tests were applied to the 35 intra-annual indices extracted from the *ACRU* record of simulated streamflows at U1H005 under present land use from 1 October 1960 to 30 September 1995 (also 36 years of record). The results of the tests indicated that while the "acceptability" of the *ACRU* time series is good with a combined score of 125 (*c.f.* Appendix 5B; Table 5B2), there is less stationarity and consistency in the *ACRU* record of streamflows than there is in the DWAF record at UIH005. It should be noted that since the *ACRU* model is "consistent" with respect to land use, the inconsistency is most likely as a result of the rainfall records utilised in the *ACRU* model. "Rainfall is the fundamental driving force and pulsar input behind most hydrological processes" (Schulze, Dent, Schäfer, Kienzle and Seed, 1995). The selection of driver rainfall stations formed an important component of the Mkomazi 2001 Study where the records from 26 stations were selected for input to the *ACRU* model. The names and details of these stations are provided in Table 6.1, whereas their distribution is shown in Figure 6.8.

The discrepancy of the time span between the two records for verifying the 35 intra-annual hydrological indices was addressed by comparing for the concurrent time period which spanned the hydrological years from 1960 to 1995 (*i.e.* 36 years). For each of the 35 intra-annual indices, the time series from both records were "combined" to represent a hypothetical 72-year time series, with the DWAF record spanning the first 36 years and the *ACRU* record spanning the second 36 years. The tests for absence of linear trend, stability of the median and of dispersion described in Appendix 5B, and which essentially hypothesise that the time series comprises two separate samples of measurements, were applied to each of the 35 72-year time series to ascertain if there were any significant statistical differences between the two records at the time step appropriate to each of the 35 intra-annual indices. The results of the tests are shown in Table 6.4.

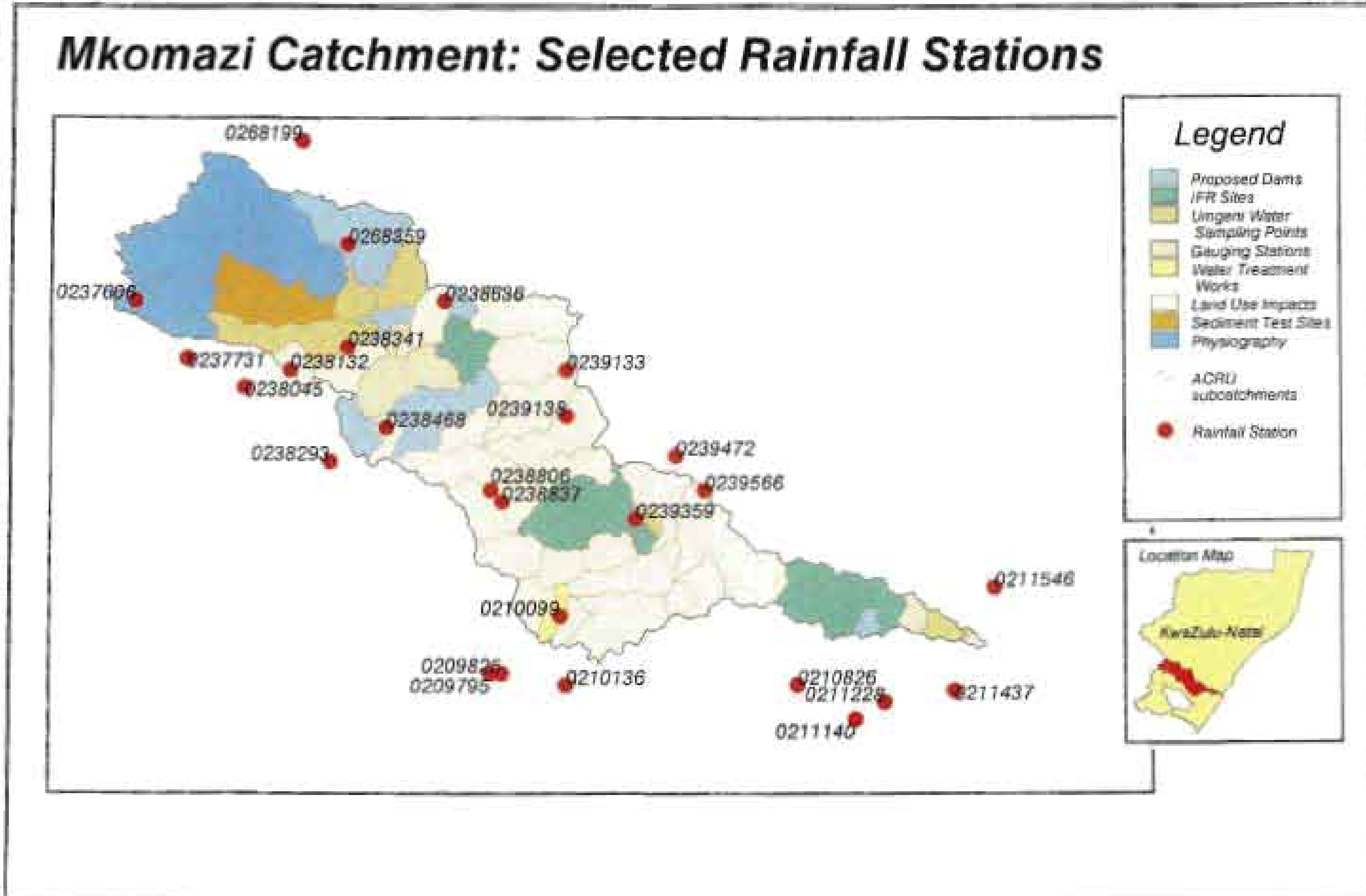


Figure 6.8 Mkomazi Catchment: Selected rainfall stations

Table 6.4 Summary of the tests applied to verify the *ACRU* simulated streamflows under present land use with the DWAF record at U1H005. Codes for the indices are described in Table 5.2 of Chapter 5. Indices of high information for the streamflow regime at U1H005, according to the Study in Chapter 5, are highlighted. Green depicts the central tendency of an index to have high information; blue depicts the variability associated with an index to have high information; yellow denotes that both central tendency and the variability associated with an index have high information. See text in Appendix 5B for details of scores and confidence levels. √ denotes test passed; * denotes test failed.

Hydrological Index	Statistical Test				
	Absence of linear trend	Stability of median	Skewness	Stability of dispersion	Score
M_A1	no	√	√	*	3
M_A2	no	√	√	√	4
M_A3	no	√	√	√	4
M_A4	no	√	√	√	4
M_A5	no	√	√	√	4
M_A6	no	√	√	√	4
M_A7	pos	*	√	*	0
M_A8	pos	*	√	*	0
M_A9	pos	*	√	*	0
M_A10	pos	*	√	*	0
M_A11	pos	*	√	*	0
M_A12	no	*	√	*	1
M_B1	pos	*	√	*	0
M_L1	pos	*	√	*	0
M_L3	pos	*	√	*	0
D_L1	pos	*	√	*	0
D_L2	pos	*	√	*	0
D_L3	pos	*	√	*	0
D_L4	pos	*	√	*	0

Hydrological Index	Statistical Test				
	Absence of linear trend	Stability of median	Skewness	Stability of dispersion	Score
D_L5	pos	*	√	*	0
D_L6	no	√	√	√	4
D_L7	no	*	√	*	1
D_H1	neg	*	√	*	0
D_H2	no	*	√	√	2
D_H3	no	√	√	√	4 (8)
D_H4	no	√	√	√	4
D_H5	no	√	√	√	4
D_H6	no	√	√	√	4
T_L1	no	√	√	√	4
T_H1	no	√	√	√	4 (8)
F_L1	no	*	√	*	1
F_H1	no	√	√	√	4
R_A1	neg	*	√	*	0
R_A2	no	√	√	√	4
R_A3	pos	*	√	*	0

Hydrological Indices		Indices of High Information	
Score (score as a %)	64 (46%)	Score (score as a %)	33 (75%)
Confidence level	3	Confidence level	2

3.4.2 Confidence in the intra-annual hydrological indices derived from *ACRU* simulated streamflows

Initial inspection of the results of the tests, as shown in Table 6.4, is disappointing, with the "acceptability" of the combined time series of the *ACRU* record and the DWAF record attracting a score of only 64 points (*c.f.* Table 6.4). This score is classified as "poor" using the convention adopted in Chapter 5, Section 3.4 (*c.f.* Appendix 5B, Table 5B2). If one accepts that the DWAF observations are an accurate reflection of the streamflow regime at U1H005, then it is clear from the tests that the *ACRU* model simulations are overestimating the baseflow component of the hydrograph. The low flow months of April through September, the 1-day and multi-day minimum flows as well as the baseflow indices all fail the tests, with the results indicating that the *ACRU* streamflows provide higher streamflow values than the DWAF record.

However, it has to be emphasised that verification of hydrological model simulations are seldom performed with such rigorous or powerful statistical tests, or at less than a monthly time step. Despite the discrepancies regarding the baseflow component of the *ACRU* record, there is considerable credibility regarding the high flow component, with the high flow months of October through to March, the multi-day maxima streamflows, high flow pulses and timing of maximum streamflow all performing well in the verification study. These indices are also well understood by society since they describe the characteristics of the streamflow regime which deliver the ecosystem goods and services most prized by stakeholders.

Therefore, it was considered appropriate to assign a less rigorous "scoring" system to the performance of the *ACRU* record. Typically in EFAs, a confidence level is assigned to the information representing the different ecological components of the resource. In this verification study a simple structure has been devised to represent the confidence that could be attributed to the hydrological indices represented by the *ACRU* record. The overall score (64) resulting from the screening of the record was expressed as a proportion of the total possible score (140). This proportion of 46% indicates a "fair confidence" level (level 3) in accordance with the level structure shown in Table 6.5.

Table 6.5 Confidence levels and associated percentage scores for the acceptability of the *ACRU* simulated streamflows

Percentage	**Confidence level**	
80 – 100	Very high confidence	1
61 - 79	High confidence	2
41 - 60	Fair confidence	3
21 – 40	Low confidence	4
1 – 20	Very low confidence	5
0	No confidence	6

In addition, the situation becomes more encouraging when one focuses on the hydrological indices of high information characterising the streamflow regime of the type recorded at DWAF gauging station U1H005. The DWAF record of daily flows recorded at U1H005 was identified in Chapter 5 as being of the type Perennial, Sustained Baseflow (*c.f.* Table 5.26). The hydrological indices which accounted for the majority of the variation provided by all the indices for this streamflow type are highlighted in Table 6.4. Six of the indices shown in Table 5.26 are indices of dispersion (M_A18, R_A4, D_H9, R_A4, T_H2 *i.e.* respectively the variability of flows in March, rate of rising river level, 7-day maximum flow, number of hydrograph reversals and the timing of the maximum flow) or general flow conditions, M_L4 (Q75), rather than indices of central tendency. For the assessment of the acceptability of the time series extracted from the *ACRU* record, indices of dispersion were assigned the same score as that attracted by the associated index of central tendency. For example, verification of the *ACRU* record with the DWAF record for index D_H3 (the 7-day maximum flow) attracted a score of four points in the screening tests applied (*c.f.* Figure 5.4, Appendix 5B, Table 5B2 and Table 6.4). The variability associated with this index (D_H9) was assumed to be equally reliable for analysis and was also assigned four points, giving a total of eight points as shown in brackets in Table 6.4. The index M_L4 (the Q75) was omitted from the scoring system since it could not be verified using the screening tests described above. The confidence level attached to the screening of the record for the indices of high information (score of 33) expressed as a proportion of the total possible score (44) at 75% indicates a high level of confidence (level 2) in accordance with the structure given in Table 6.5.

This level of confidence in the hydrological indices extracted from the *ACRU* record representing present land use conditions generated at the site of U1H005, was assessed to be sufficiently strong to undertake a parallel exercise to assess the same indices extracted from the *ACRU* record of simulated streamflows representing "baseline", or reference, conditions (*i.e.* simulated using Acocks (1988) Veld Types as land cover) spanning the same 36-year period from 1960 to 1995. Thus, for each of the 35 intra-annual indices, the time series from both the observed and the simulated "baseline" records were "combined" to represent a hypothetical 72-year time series, with the DWAF record spanning the first 36 years and the *ACRU* record spanning the second 36 years. As expected, the tests indicated that there was very little difference between the *ACRU* record of streamflows under present land use and the *ACRU* record of streamflows under reference conditions (*c.f.* Table 6.6). The results in Table 6.6 indicate an overall score of 65 for the combined set of 35 intra-annual indices and a score of 35 for the indices of high information. These scores translate into proportions of 46% and 80% respectively and also attract confidence levels of 3 and 1 respectively in accordance with Table 6.5. Thus, confidence was considered to be sufficient to utilise the *ACRU* daily streamflow record of reference conditions generated by the Mkomazi 2001 Study in the following study, at the temporal resolution of the hydrological indices for each of the 52 sub-catchments or "hydrological zones" (*c.f.* Section 2.3 of this Chapter and Figure 6.7).

Collective experience within the School of Bioresources Engineering and Environmental Hydrology (BEEH) at the University of KwaZulu-Natal in Pietermaritzburg indicates that when using the *ACRU* modelling system, the first few years of simulated streamflows are usually dissimilar to those of the subsequent years. This feature may not be material to any study applying the medians of index values. However, as an additional statistical precaution only the record period from 1 October 1952 to 1 September 1995 (of the available *ACRU* record period from 1945 to 1995) was used to extract the hydrological indices described in the following study. According to the study of record length in Chapter 5, Section 4.5.5, this record length (44 years) is sufficiently long to produce reliable site averages, or indices of central tendency (*i.e.* M_A1 to M_A12; M_L1 and M_L3; M_H1; D_L1 to D_L7; D_H1 to D_H6; T_L1 and T_H1; F_L1 and F_H1; R_A1 to R_A3), and of the dispersion (using the CD) of the indices (Chapter 5, Section 4.5.5 and Tables 5.18 and 19) for the Perennial Sustained Baseflow Group in Chapter 5, of which the DWAF record at U1H005 forms part.

Table 6.6 Summary of the tests applied to verify the *ACRU* simulated "reference" streamflows with the DWAF record at U1H005. Codes for the indices are described in Table 5.2 of Chapter 5. Indices of high information for the streamflow regime at U1H005, according to the Study in Chapter 5, are highlighted. Green depicts the central tendency of an index to have high information; blue depicts the variability associated with an index to have high information; yellow denotes both central tendency and the variability associated with an index have high information. See text in Appendix 5B for details of scores and confidence levels. √ denotes test passed; * denotes test failed.

Hydrological Index	Statistical Test					Hydrological Index	Statistical Test				
	Absence of linear trend	Stability of median	Skewness	Stability of dispersion	Score		Absence of linear trend	Stability of median	Skewness	Stability of dispersion	Score
M_A1	pos	*	√	*	0	D_L5	pos	*	√	*	0
M_A2	no	√	√	√	4	D_L6	no	√	√	√	4
M_A3	no	√	√	√	4	D_L7	no	√	√	*	3
M_A4	pos	√	√	√	3	D_H1	pos	*	√	*	0
M_A5	no	√	√	√	4	D_H2	no	√	√	√	4
M_A6	no	√	√	*	3	D_H3	no	√	√	√	4 (8)
M_A7	pos	*	√	*	0	D_H4	no	√	√	√	4
M_A8	pos	*	√	*	0	D_H5	no	√	√	√	4
M_A9	pos	*	√	*	0	D_H6	no	√	√	√	4
M_A10	pos	*	√	*	0	T_L1	no	√	√	√	4
M_A11	pos	*	√	*	0	T_H1	no	√	√	√	4 (8)
M_A12	no	*	√	*	1	F_L1	no	√	√	*	3
M_H1	pos	*	√	*	0	F_H1	no	√	√	√	4
M_L1	pos	*	√	*	0	R_A1	no	√	√	√	4
M_L3	pos	*	√	*	0	R_A2	pos	*	√	*	0
D_L1	pos	*	√	*	0	R_A3	pos	*	√	*	0
D_L2	pos	*	√	*	0	Hydrological Indices			Indices of High Information		
D_L3	pos	*	√	*	0	Score (score as a %)		65 (46%)	Score (score as a %)		35 (80%)
D_L4	pos	*	√	*	0	Confidence level		3	Confidence level		1

3.5 Statistical Analysis of Indices of the Reference Hydrological Conditions of the Mkomazi Catchment

The methods used to analyse the indices of reference hydrological conditions of the Mkomazi Catchment statistically, mirror those applied in the Principal Components Analysis (PCA) described in Chapter 5 of this Thesis. The concepts, strengths and weakness of PCA, as well as the ecological relevance of the indices, are described in Chapter 5 and, consequently, will only be reiterated where appropriate to this Mkomazi Study. The 74 ecologically relevant hydrological indices representing different components of the streamflow regime (*c.f.* Table 5.2) used in Chapter 5 were derived from the *ACRU* records of simulated streamflows of "reference hydrological conditions" from 1952 - 1995 at the outlet of each of the 52 zones of the Mkomazi Catchment.

PCA was extracted from the 74 by 74 correlation matrix (74 hydrological indices) using the Genstat Version 6 computer software. Following Olden and Poff (2003), PCA was conducted to investigate the inter-relationships among the hydrological indices and to ascertain subsets of indices explaining the major sources of variation, while minimising redundancy, for the different reference hydrological zones that could be identified for the Mkomazi Catchment. First, a PCA was performed at the catchment scale using the indices for the combined set of 52 zones. PCA provides information about the relationships among both objects and descriptors (Legendre and Legendre, 1998), since both the descriptor-axes and the objects can be plotted in reduced space. A scatter plot of PCA scores for the first two principal components (which together explained 77.29% of the total variance in the data set, *c.f.* Section 3.5.2) from the 74 x 74 correlation matrix (Figure 6.9) was examined to highlight the clustering of the 52 reference hydrological zones, with respect to the first two principal component axes.

As discussed in the study in Chapter 5, there is a degree of subjectivity involved in grouping objects based on their PCA scores (Ndlovu, 2004). Analysis of the environmental gradient of the 52 reference hydrological zones indicates that the zones could be grouped into five separate types. Those zones at the extremes of the plot were placed in three separate groups (Groups A, D, and E on Figure 6.9), whereas two additional groups were formed from the remaining zones, based on the similarity of their PCA scores along the first two principal components axes. In

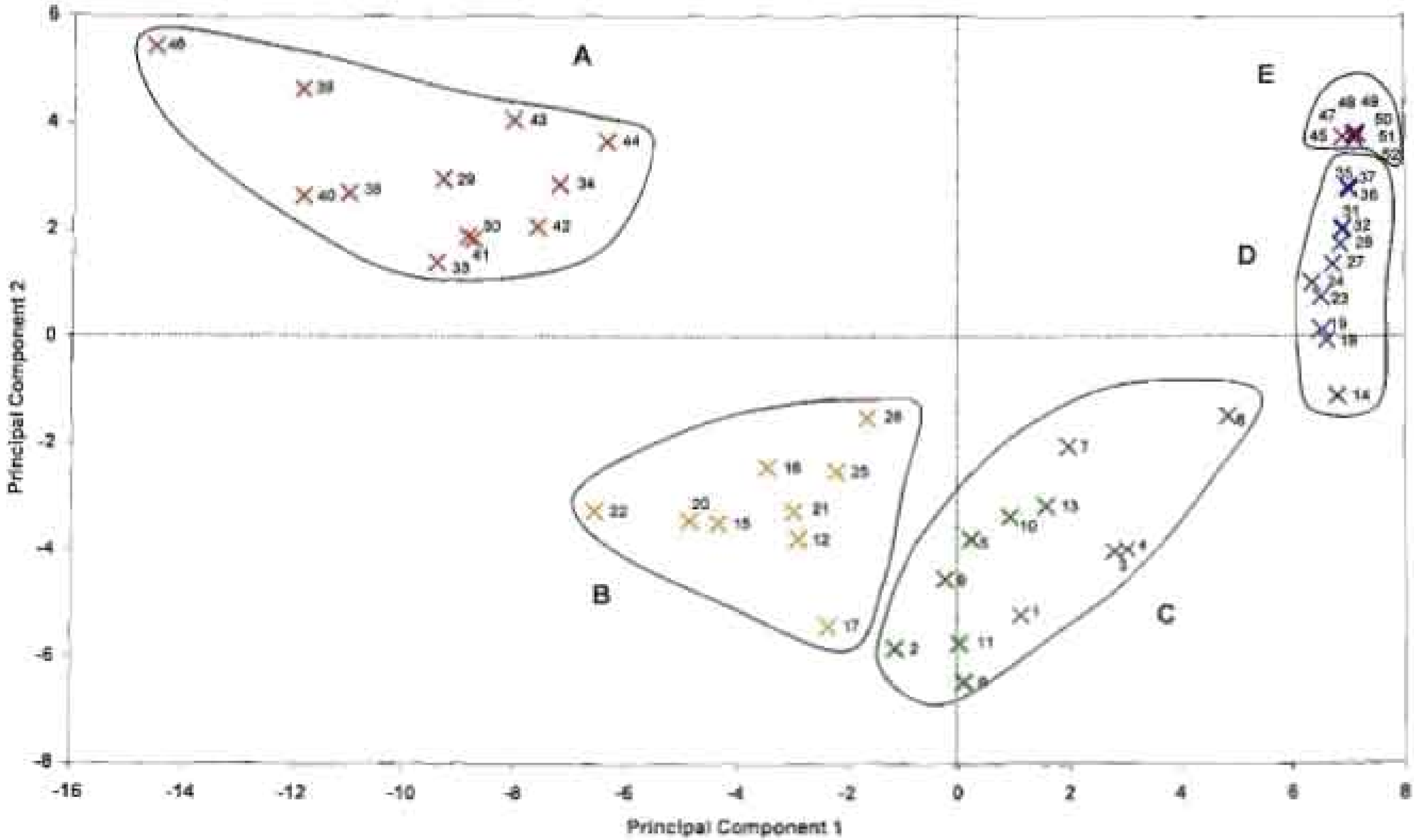


Figure 6.9 Bivariate plot of scores for the first two principal components from a PCA of the 52 Mkomazi reference hydrological zones (crosses 1 to 52) based on the correlation matrix of 74 hydrological indices. Letters A to E refer to groups of reference condition hydrological zones within the Mkomazi Catchment.

addition, some zones were grouped in a logical manner despite small differences in their scores, *viz.*:

(a) Zones 2 and 9 in Figure 6.9 have negative scores on the first principal component axis, yet it was more appropriate to include them in Group C rather than in Group B, given the similarities of their high values of HIGHDUR, *i.e.* the length of high flow pulses (12.1 for Zone 2; 13.1 for Zone 9) and FLOODFREE, *i.e.* the seasonality of non-flooding (36 for Zone 2; 22 for Zone 9) compared with the much lower values of these indices for zones in Group B (Group medians of 8.80 and 15.00 respectively).

(b) Zones 14 and 18 in Figure 6.9 have negative scores on the second principal component axis, yet it was more appropriate to include these zones in Group D rather than in Group C, given the similarities in the high values of REVERSALS, *i.e.* the numbers of hydrograph reversals (101 for Zone 14; 103 for Zone 18) and PRED, the predictability of the streamflow regime (0.49 for Zone 14; 0.48 for Zone 18) compared with the lower values for these indices for the zones in Group C (Group medians of 91 and 0.47 for reversals and predictability respectively).

In addition to the PCA performed at the whole catchment scale using the indices for the combined set of 52 zones, PCAs were conducted for the five different reference hydrological zone types identified in Figure 6.9. As in Chapter 5 and following Olden and Poff (2003), statistical significance of the Principal Component Analysis was evaluated using the Broken-stick model (Frontier, 1976), where observed eigenvalues computed by each PCA were compared with a decreasing list of the expected eigenvalues generated by the Broken-stick model (*c.f.* Chapter 5, Section 4.4.1). In the PCA, the Genstat Version 6 software computed the eigenvectors (loadings) of the hydrological indices on each of the principal components. Following Olden and Poff (2003), loadings of the hydrological indices on each significant principal component identified by the Broken-stick model were used to identify indices that explained the dominant patterns of variation for (a) the combined set of zones and (b) each of the five different reference hydrological zone types.

3.5.1 Reference condition hydrological zones

The five groups of reference condition hydrological zones were categorised within two main groups of streamflow type using the same set of streamflow characteristics as those applied in the study in Chapter 5 (*c.f.* Tables 5.9 and 5.10). The classification of the reference hydrological zones within the Mkomazi Catchment, determined using the object (*i.e.* zone) scores of the PCA (*c.f.* Figure 6.9) is shown in Table 6.7. The numbers of zones within each group, group medians (large numerals) and coefficients of dispersion (smaller numerals) are shown for selected characteristics. Shading around the medians indicates a distinguishing index of the Group. Figure 6.9 shows a schematic of the distribution of the five different reference hydrological zone types of the Mkomazi Catchment.

As described in Section 5.3.1 of Chapter 4, the hydrological cycle links the different aquatic ecosystems of rivers, wetlands and lakes and groundwater, with terrestrial ecosystems beyond the stream channel and floodplain through its energy, sediment and water flows. This connectivity is the focus of the zonation of the Mkomazi Catchment presented in this Study. At the catchment scale, precipitation is the basic water resource (Falkenmark, 2003) and the zonation of the Mkomazi Catchment response to climatic characteristics under reference conditions is based on the hydrological processes associated with precipitation. Zones where excess precipitation generates runoff or groundwater recharge for the downstream hydrological processes of infiltration, evaporation, transpiration and nutrient cycling as well as deposition, were grouped under the general umbrella of "water source zones" (see Molden *et al.*, 2001). Subsequently, these water source zones were formed into three subgroups (Groups A, B and C) each of which was based on distinct hydrological characteristics of streamflow response to climatic characteristics. Zones where the landscape drains surface and subsurface water into the main river channel were grouped under the general umbrella of "natural recapture zones" (see Molden *et al.*, 2001). Streamflows in natural recapture zones represent the portion of upstream runoff or groundwater which has not been utilised by upstream hydrological processes and additionally comprises the contribution of precipitation over the zone. The natural recapture zones were subsequently formed into two subgroups (Groups D and E), based on distinct streamflow characteristics.

The groupings are discussed below, with reference to group medians and coefficients of dispersion (Table 6.7) as well as the schematic of the distribution of the five different reference hydrological zone types of the Mkomazi Catchment (Figure 6.10). As in Chapter 5, and for the sake of consistency among different groupings, the style and terminology used mirror that applied in Method One of 1994-Study of Joubert and Hurly (1994).

Table 6.7 Classification of reference hydrological zones within the Mkomazi Catchment (discretised into 52 hydrological zones). Group medians (large numerals) and the coefficients of dispersion (smaller numerals) of selected hydrological indices of each of the five zone types determined from the PCA are provided. Shading of the medians indicates a distinguishing index of the zone type.

	Water Source Zone						Natural Recapture Zone			
	Low Precipitation Number: 12		Moderate Precipitation 9		High Precipitation 12		Mainstream 12		Accumulated 7	
INDICATOR	Short unseasonal floods		Unpredictable flow and floods		Seasonal predictable flow and floods		Predictable Runoff		Sustained Baseflow	
LOWDUR	5.50	0.16	7.90	0.37	14.25	0.27	12.90	0.15	11.00	0.00
HICOUNT	11.00	0.27	8.50	0.18	8.00	0.13	8.00	0.19	10.00	0.00
HIGHDUR	5.30	0.22	8.80	0.13	10.30	0.19	9.30	0.15	7.50	0.04
REVERSALS	95.00	0.09	87.00	0.13	91.50	0.11	105.00	0.03	113.00	0.00
BFI	0.43	0.04	0.46	0.08	0.44	0.04	0.46	0.01	0.47	0.00
CDB	7.26	0.21	4.53	0.29	3.14	0.14	3.18	0.12	2.95	0.09
HFI	45.92	0.38	17.70	0.59	12.58	0.11	11.04	0.10	12.37	0.00
PRED	0.23	0.13	0.33	0.21	0.47	0.09	0.47	0.06	0.44	0.00
PROP	0.65	0.06	0.57	0.11	0.57	0.04	0.59	0.03	0.62	0.00
%FLOODS	0.31	0.06	0.35	0.09	0.41	0.07	0.38	0.08	0.34	0.00
FLOODFREE	2.00	2.00	15.00	0.56	31.50	0.48	21.00	0.00	11.00	0.08

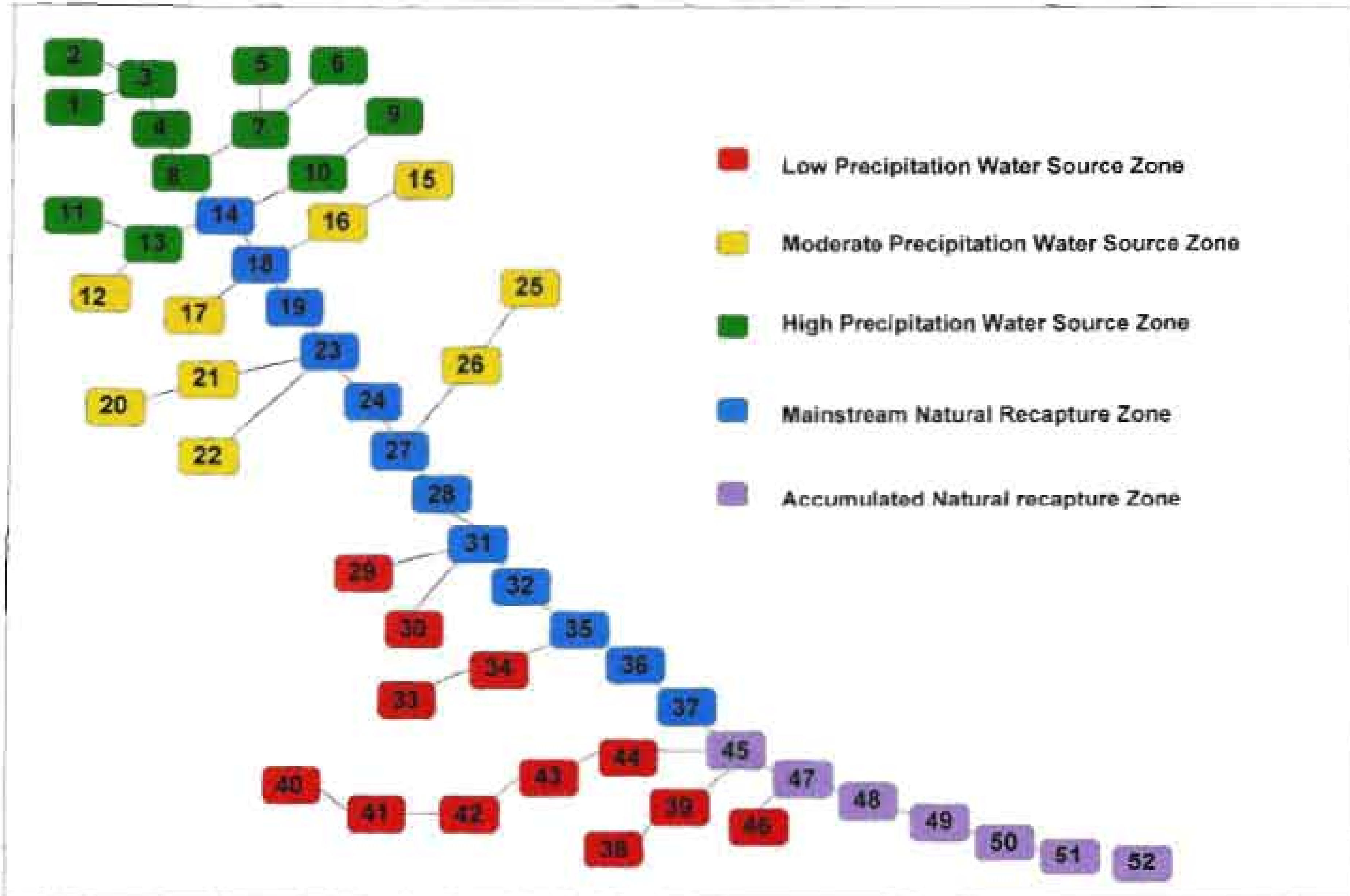


Figure 6.10 Schematic representation of the reference condition hydrological zones within the Mkomazi Catchment (discretised to 52 zones)

Water Source Zone Main Group

Low precipitation water source zone / Short Unseasonal Floods

Group A comprises 12 hydrological zones of "Low Precipitation Water Source" (LPWS), six of which generate quasi-perennial flow, with the remaining six generating perennial flow. However, all zones in this group have unpredictable flow regimes (group median PRED of 0.23), with an average of only 14% of PRED being attributable to constancy. The zones in this group also have very low numbers of days during a year for which no floods have ever occurred over the 36 year record (group median FLOODFREE of 2.00). Flood regimes are therefore highly unseasonal and unpredictable (group median %FLOODS of 0.31). The zones in this group also have, on average, the shortest flood durations (group median HIGHDUR of 5.30 days) and are relatively "flashy" with group median REVERSALS of 95.00. A relatively large proportion of the streamflows generated in the zones in this Group constitutes the "high flow component" (HFI). Figure 6.10 shows that most of these zones are located in the lower reaches of the Mkomazi Catchment where there is least rainfall (MAP generally less than 900 mm, Table 6.1 and Figure 6.4). All the zones are located on hillslopes to the west of the main river channel.

Moderate precipitation water source zone / Unpredictable Flow and Floods

Group B comprises 9 hydrological zones of "Moderate Precipitation Water Source" (MPWS), one of which (Zone 22) generates quasi-perennial flow whereas the remaining eight generate perennial flow. Similarly to the zones in Group A, zones in Group B also have relatively low predictability of flow (group median PRED of 0.33) and floods (median %FLOODS of 0.35). Group B zones have, on average, a lower number of floods each year than Group A zones (group median HICOUNT of 8.50 compared with 11.00) which on average, are longer than those of the Group A zones (group median of 8.80 days compared with 5.30 days). However, zones in Group B generally have a higher incidence of days on which no floods have occurred (higher FLOODFREE value, group median of 15.00 compared with median of 2.00) and a slightly lower PROP (median of 0.57 compared with median of 0.65) than those in Group A. Thus, Group B flood regimes are more seasonal than those of Group A zones. In addition, the zones in Group B generally have a higher baseflow component (group median Alt-BFI of 0.46 compared with 0.43) which, together with a much lower CDB (group median of 4.53 compared with median of 7.26), indicates that these streamflow regimes are also less variable than those in Group A. Group B zones are located throughout the middle to upper reaches of the Mkomazi

Catchment, both to the east and west of the main river channel (*c.f.* Table 6.1 and Figure 6.10) and where there is moderate rainfall (MAP of 900 - 1100 mm, Figure 6.4).

High precipitation water source zone / Seasonally, Predictable Flow and Floods

Group C comprises 12 hydrological zones of "High Precipitation Water Source" (HPWS), all of which generate perennial flow. Unlike the zones in Groups A and B, zones in this Group have streamflows of relatively high overall predictability (group median PRED of 0.47). However, on average, constancy was low for Group C zones, as is evidenced by the low group median value for PROP (0.57). Consequently, these zones generate streamflows with a degree of seasonal predictability. The zones in this Group also have the highest degree of seasonal predictability of non-flooding (group median FLOODFREE of 31.50). Figure 6.10 shows that these zones are mostly located in the upper reaches of the Mkomazi Catchment, and form the main water source regions and headwaters of the system, where there is high rainfall (greater than 950 mm, Table 6.1 and Figure 6.4).

Natural Recapture Zone Main Group

Mainstream natural recapture zone / Predicable Runoff

Group D comprises 12 hydrological zones of "Mainstream Natural Recapture" (MNR), all of which generate perennial flow. Similarly to the zones in Group C, zones in Group D also generate relatively predicable flow (group median PRED of 0.47), with long spells of low flow (group median LOWDUR of 12.90 days, *i.e.* where flows within a year are less than the 25th percentile of all flows across the record) and relatively short flood durations (group median HIGHDUR of 9.30). In addition, the higher average (group median of 105.00) of hydrograph REVERSALS (change in rising and falling river levels) indicates that the flow regimes in this group respond rapidly to either pulses of upstream flows or within-zone rainfall events and Group D zones could therefore be described as "flashy". Group D zones are all located in the upper to middle reaches of the Mkomazi River system. These zones recapture the upstream streamflows from the main water source regions and headwaters of the system as well as the lower altitude, more moderate rainfall water source zones and some of the low rainfall water source zones (*c.f.* Figure 6.10).

Accumulated natural recapture zone / Sustained Baseflow

The seven hydrological zones of "Accumulated Natural Recapture" (ANR) in Group E all generate perennial flow with a high baseflow regime (high Alt-BFI, with a group median of 0.47) and a relatively low "high flow component" (HFI, group median of 12.37). Zones in this group have the lowest overall variability (group median CDB of 2.95), yet have the highest change in rising and falling river levels (REVERSALS, group median of 113.00), indicating the greatest response among all the zone types to changes in pulses of flows from upstream sources. These zones are all located in the lower mainstream Mkomazi River system, accumulating all the upstream contributions of the river system. Consequently, these zones are characterised more by the upstream hydrological system than by rainfall as a primary source.

3.5.2 The meaningful components

The results from the PCA of the 52 reference hydrological zones, based on 74 hydrological indices, are shown in Table 6.8. The number of statistically significant principal component axes, using the values computed by the Broken-stick model, ranged from two (*e.g.* the ANR Group) to five (*e.g.* the HPWS Group) and together explained 77.29% of the variation for the combined set of "All-52 zones" and from 82.94% to 94.05% for the five reference hydrological zone types.

Table 6.8 Results from the principal component analysis on the correlation matrix of 74 hydrological indices based on 52 reference hydrological zones of the Mkomazi Catchment grouped into five zone types

Zone type	**Principal Component (% variation explained)**					**Total**
	1	2	3	4	5	
Low Precipitation Water Source	51.50	14.73	9.58	7.13	-	82.94
Moderate Precipitation Water Source	45.18	25.96	10.67	8.18	-	89.99
High Precipitation Water Source	44.79	15.78	11.46	10.08	6.09	88.2
Mainstream Natural Recapture	68.83	10.80	9.55	-	-	89.18
Accumulated Natural Recapture	80.24	13.81	-	-	-	94.05
All-52 zones	60.90	16.39				77.29

The first two principal components (PC1 and PC2) together explained 77.29% of the variation for the combined set of "All-52 zones" and from 60.57% (HPWS Group) to 94.05% (ANR Group) for the five zone types, indicating that PC1 and PC2 explain the majority of the variation in the indices. In each instance, these results indicate that the first two principal component axes account for the majority of the variability for each of the zone types. Consequently, the results are very useful for ecological studies of the Mkomazi Catchment.

3.5.3 General patterns of inter-correlation, or redundancy, among the indices

Figure 6.11 shows the ordination from the PCA of the 52 reference hydrological zones (All-52 Zones) based on 74 indices, plotted in the plane determined by the first two principal component axes. As discussed in Chapter 5, the co-ordinates of the indices are the apices of the eigenvectors, plotted as a function of $\sqrt{\lambda}$ (*i.e.* each eigenvector was rescaled to length $\sqrt{\lambda_k}$, the square root of the k-th eigenvalues as in Legendre and Legendre, 1998). The correlation among indices is given by the angle between the index-axes (Legendre and Legendre, 1998). Consequently, as explained by Olden and Poff (2003), indices separated by small angles (*e.g.* M_A4 and M_A5 with high loadings on PC 1, as shown in Figure 6.11), are highly positively correlated; indices separated by angles up to 180^O(*e.g.* T_A2 and T_H3 with opposite "high" loadings on PC 2, as shown in Figure 6.11), are "highly" negatively correlated; whereas indices separated by a right angle (*e.g.* indices T_L1 and T_H2, as shown in Figure 6.11) are not correlated.

The positions of the index apices shown in Figure 6.11 indicate that many of hydrological indices are highly inter-correlated. Figure 6.11 shows that for the combined set of zones (All-52 zones) the highest correlation is among indices of central tendency describing certain streamflow characteristics, and includes indices of the following streamflow conditions:

(a) the magnitudes of average flow conditions (*i.e.* the mean monthly streamflows);

(b) a range of durations of high flow conditions (*i.e.* the 1-day, multiple-day and seasonal high flow disturbances);

(c) a range of durations of low flow conditions (*i.e.* the 1-day, multiple-day and seasonal low flow disturbances);

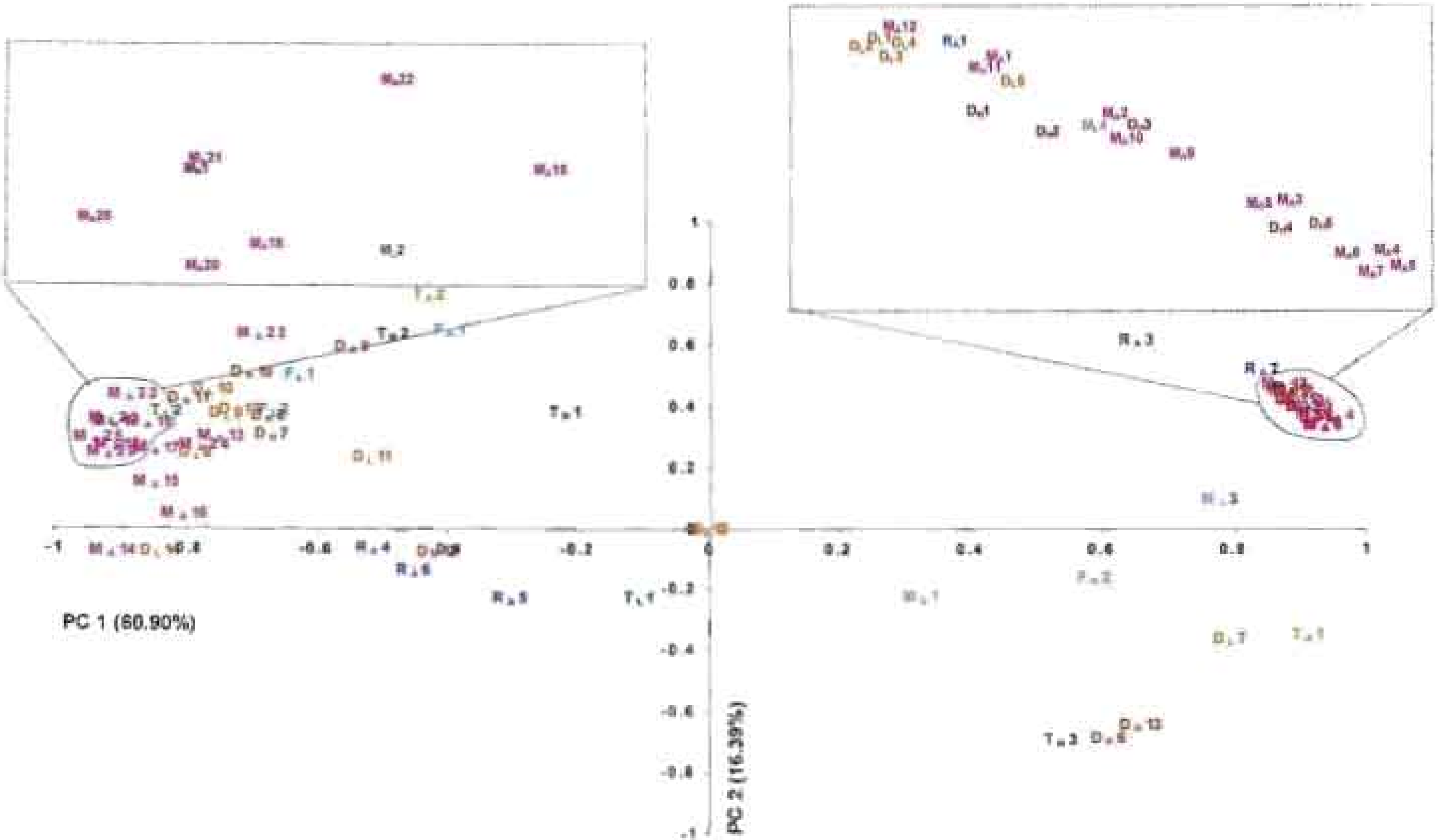


Figure 6.11 Ordinations of the 74 hydrological indices, from the PCA of 52 reference hydrological zones in the Mkomazi Catchment, in the plane of the first two principal component axes. Correlations among the indices are interpreted as the cosine of the angle separating their index-axes. Each eigenvector was rescaled to the length $\sqrt{\lambda_i}$ to display the correlations among the indices. Some of the data are shown in enlargement boxes for clarity.

(d) the magnitude of low flow conditions (*i.e.* a flow threshold that is equaled or exceeded 75 % of the time, Q75); and

(e) the rates of change of average flow conditions (*i.e.* the rise and fall rates of river levels).

These indices (M_A1-M_A12; D_H1-D_H5; D_L1-D_L5; M_L4 and R_A1-R_A2) are clustered (Cluster 1) in the upper right quadrant of Figure 6.11, with high loadings on the first principal component (PC1), and are highly correlated with each other. This finding mirrors that which was identified in the study in Chapter 5, where the same indices were identified as being highly correlated for the combined set of stream types found across South Africa (*c.f.* Figure 5.11 for All-83 Streams and Figure 5.47 for All-48 Streams). In addition, this finding confirms the transferability of these indices across varying "stream types" and different climatic conditions, since this subset of indices consistently explains the dominant patterns of variance at different spatial scales.

There are also several small clusters of inter-correlated indices shown in Figure 6.11. These include a small cluster of dispersion-based indices in the upper left quadrant with high loadings on PC 1 (Cluster 2). The indices in Cluster 2 represent the variability in the magnitude of average flow conditions for several calendar months; in particular, M_A17, M_A18, M_A19, M_A20, M_A21 and M_A22, the variability in flows for February, March, April, May, June and July respectively are correlated with M_L2 and M_A25, the IHA index of variability of "baseflow" and the Desktop Reserve Model index of overall variability) Hughes and Hannart, 2003) respectively as well as M_H1 (HFI, the high flow index). The indices in Cluster 1 are negatively correlated to the indices in Cluster 2. In addition, there is a second small cluster of correlated dispersion-based indices (D_H7 and D_H8; D_L9, D_L10 and D_L12; F_L2 and M_A13) with moderately high loadings on PC1 in the upper left quadrant (Cluster 3). The indices in Cluster 3 are also dispersion-based indices, representing the variability of the shorter high flow conditions and low flow conditions as well as the magnitude of average flows in October.

In addition, to the above there are a number of smaller groups of moderately correlated indices. These include the following:

(a) the constancy of flows (T_A2) as an indicator of the predictability of the seasonality of the flow regime, and the frequency of high pulses (F_H1);

(b) rates of change in streamflow conditions represented by the rise rate (R_A4) and the number of hydrograph reversals (R_A6) and

(c) the duration of high flow pulses (D_H6), the predictability of the flood regime (T_H3) and the seasonality of non-flooding (D_H13)

As in the study in Chapter 5, there are number of indices which are closer to the origin and are generally uncorrelated with the indices that have higher loadings on the first two principal components. These include the timing of both the minimum and maximum flows (T_L1 and T_H1) and the IHA "baseflow" index (M_L1), the variability of rate of falling flows (R_A5), the number of days with zero flow (D_L6) and the variability thereof (D_L13).

3.5.4 Selecting high information, non-redundant indices

The PCA identified groups (or clusters) of indices that exhibited the largest loadings on each significant principal-component axis for (a) the combined set of 52-zones and (b) the five distinct zone types. These indices contain the most information associated with the variation provided by all the indices in the dataset. As explained in Chapter 5, the significance of the index loadings cannot be tested using a routine statistical test for correlation coefficients because the principal components are linear combinations of the indices and, as such, are essentially correlated (Legendre and Legendre, 1998; Olden and Poff, 2003). In this Study, the 12 indices with the highest absolute loadings on the significant principal-component axes for each of the different reference hydrological types were selected to produce a subset of high information indices for researchers to use in environmental flow studies of the Mkomazi River and its tributaries. Following the procedure outlined by Olden and Poff (2003), this was achieved by setting the number of indices selected from each principal component equivalent to the proportion of variation explained by the component when compared to all significant axes.

This analysis identified groups of indices with the highest absolute loadings on each of the significant principal-component axes for the combined set of All-52 zones and for the five distinct zone types. Table 6.9 shows the 12 hydrological indices with the highest absolute loadings on each of the statistically significant principal components. While Table 6.9 indicates the direction of loading (positive or negative) for each index selected, this attribute is not important to this part of the analysis. Selecting indices on each of

Table 6.9 Hydrological indices with the largest absolute loadings on each statistically significant principal component. Zone types are based on the classification shown in Table 6.7. Indices annotated with "–" have a negative loading of the index on the principal component, although it is the magnitude of the loading rather than its direction that is relevant to this analysis. Different colours represent the seven main characteristics of the streamflow regime (see text for details).

Principal Component	Water Source Zone			Natural Recapture Zone		
	Low Precipitation	Moderate Precipitation	High Precipitation	Mainstream	Accumulated	All Zones
1	M_A3-	M_A10-	M_A5	D_L2-	D_L5-	M_A5
	M_A5-	M_A7-	M_A6	D_H4-	M_A10-	M_A4
	M_A4-	M_L4-	M_A4	D_L1-	M_A12-	M_A7
	D_H4-	M_A9-	D_H4	M_L4-	D_L11	M_A25-
	D_H5-	M_A11-	M_A8	D_H5-	D_H5-	M_A6
	D_H2-	M_A6-	M_A9	M_A11-	D_L2-	D_H5
	D_H3-			D_L4-	M_A6-	D_H4
				M_A12-	[illegible]	M_A3
				D_H1-	[illegible]	M_A8
					M_A11-	
2	T_H3-	M_A24-	M_A25-	M_A24	D_L9	T_A2
	[illegible]	D_L12-	M_A24-	M_L2-	F_L2-	T_H3
		M_A20-				D_H6-
		M_A19-				
3	M_A13	[illegible]	T_L1-	F_H2		
	D_H6-		D_L11-			
4	M_L2	T_H1-	[illegible]			
5			D_L9			

the statistically significant principal components ensures that the subsets of indices are independent from each other (Olden and Poff, 2003). Thus, Table 6.9 indicates the groups of indices that are representative of the major gradients of variation described by all the indices for the different zone types.

All seven main characteristics of the streamflow regime (*c.f.* Section 4.4.3 of Chapter 5) are represented in Table 6.9. The main streamflow characteristics and numbers of representative indices in the study comprising this Chapter are;

(a) the magnitude of average flow conditions ($n = 25$),
(b) the magnitude of either low or high flow conditions ($n = 5$),
(c) the duration of low flow conditions ($n = 14$),
(d) the duration of high flow conditions ($n = 13$),
(e) the timing of flow events ($n = 7$)
(f) the frequency of flow events ($n = 4$) and
(g) the rate of change of flow events ($n = 6$).

As high information indices, some indices are common to more than one zone type (Table 6.9). These are, most notably, D_H4 (30-day annual maximum flow), D_H5 (90-day annual maximum flow) and M_A6 (average March flows), each of which has four occurrences among the different zone types. As high information indices, M_A4, M_A11 and M_A12 (average flows in January, August and September), M_A24 (variability in September flows) and R_A4 (variability in the rate of rising flow) each have three occurrences among the different zone types.

In general, the magnitude of average flow conditions has the greatest representation of dominant indices among the streamflow characteristics across the different zone types. As explained in the study in Chapter 5, this is more than likely as a result of the higher incidence of these indices in the dataset compared with other indices. The duration of high flow conditions is also strongly represented across most of the different zone types, and in particular for zones in the LPWS group. Nonetheless, as explained in Chapter 5, where the indices are derived from the same component axes, as indeed these are (*i.e.* D_H2, D_H3, D_H4 and D_H5 all from PC1, *c.f.* Table 6.9), they are not independent from each other, and choices may need to be made. In light of this, the magnitude of low flow conditions is well represented across the different zone types.

Conversely, there are no indices of the timing of flow conditions shown as dominant indices across the zones in the main natural recapture group, and no indices of the frequency of flow conditions shown across the zones in the main water source group. Therefore, it can be deduced that indices relating to the timing of the annual maximum or minimum flow and the predictability or constancy of flow conditions and the seasonal predictability of flooding are less relevant for characterising the streamflow regimes of the mainstream Mkomazi River. On the other hand, indices relating to the frequency of both low and high flow pulses are less relevant for characterising the streamflow regimes of the Mkomazi headwaters or streamflow generating areas. That is not to say that indices of timing or frequency per se not important for these respective zone types, but rather that the dataset of 74 indices did not contain any indices of timing or frequency which adequately explained a high proportion of the variation in the indices of the respective zone types in the Mkomazi Catchment.

In addition, indices M_L1 and M_L3 (the IHA "baseflow" index and the Alt-BFI based on the Desktop Reserve Model index of baseflow) and M_H1 (the only index in the dataset comprising the magnitude of high flows) are not shown as being of high information for characterising any of the zone types in Table 6.9, including the combined set of All-52 zones. The calculations for M_L1 and M_L3 are very different (*c.f.* Chapter 5, Table 5.2 and Sections, 2.3.3 and 2.4.3), with the former representing low flow disturbance rather than the baseflow component of the streamflow regime. However, their omission from Table 6.9 indicates that indices of baseflow conditions may well be less relevant for explaining the variability associated with the different zone types in the Mkomazi Catchment. Likewise there is no incidence of the indices D_L6 and D_L13 (average number of days with zero flow and the variability thereof) shown in Table 6.9 for any of the zone types. Consequently, neither of these indices is useful for characterising the different zone types.

High information indices for the Mkomazi Catchment (All-52 Zones)

Table 6.9 indicates that for an initial (or pilot) study of the ecological flow requirements of the Mkomazi Catchment, researchers could choose from several high information indices representing four of the seven main characteristics of the streamflow regime. M_A3 to M_A8 (average monthly streamflow conditions from December through May), D_H4 (30-day maximum), D_H5 (90-day maximum flow) and M_A25 (the CDB, based on the Desktop Reserve Model index of overall variability) are all important indices of the combined set of

All-52 zones. However, because these indices are derived from the same component axis (*i.e.* all from PC1, *c.f.* Table 6.9), they are not independent from each other (Olden and Poff, 2003) and choices among those indices may need to be made. However, any of these indices could be paired with Colwell's predictability index (T_A2) as an important index of the timing of flow events, T_H3 (the predictability of flooding) or D_H6 (length of the high flow pulse) as useful measures of the streamflow conditions exhibited by the wide diversity hydrological zones in the combined set of All-52 zones.

Indices of central tendency are sufficient descriptors of the different zones types at the basin spatial scale. This feature was also prominent in the broad spatial scale study of the diversity of different flow regimes found across South Africa in Chapter 5. Indeed, there are similarities among the high information indices shown for the All-52 Zones and the All-48 Streams (and even the All-83 Streams despite its shortcomings in being representative of "reasonably natural flow", *c.f.* Table 5.26). However, there is no similarity between the indices of high information for characterising the Mkomazi Catchment (at the scale of the combined set of All-52 zones) and those identified as being of high information of the streamflow type describing the area upstream of DWAF gauging station U1H005 (*c.f.* indices for the Perennial, Sustained Baseflow Group in Table 5.26). However, the gauging station at U1H005 is just one site out of nine comprising the Perennial, Sustained Baseflow Group in Chapter 5). Moreover, Figure 5.44 shows that the object scores for the first two principal components of the PCA performed for the study of the "Best48" streamflow records placed U1H005 (E5) at one extreme of the environmental range of this Group. The discrepancies between this Study and that in Chapter 5 regarding the two sets of indices of high information indices reflect the differences between the streamflow regime representing the upper Mkomazi Catchment and that representing the entire catchment.

Indices for the Low Precipitation Water Source zone type

Table 6.9 indicates that the high information indices which could be useful to eco-hydrological studies of "short, unseasonal flood" regimes (*c.f.* Table 6.7) of the LPWS zone type represent five of the seven main characteristics of the streamflow regime. Indices regarding the duration of high flow events (D_H2 to D_H6) and the rise rate of streamflows (R_A4) are good measures of the variation in the indices for short, unseasonal flood streamflow regimes found in the LPWS zone type. Moreover, indices M_A3 to M_A5

(the magnitudes of flows from December through February and representing the beginning of the summer rainfall season) all contain high information, but are derived from the same component axis as D_H2 to D_H5 (PC 1 in Table 6.9). Consequently these indices are not independent and choices would need to be made depending on the aims of any water development study. On the other hand, the variability in the magnitude of streamflows in October (M_A13) is a high information and non-redundant index of average flow conditions. Using this method of selection, only M_L2 (variability in the IHA "baseflow index") could be chosen as an important measure of low flow conditions for the hydrological zones in this Group. In general, in the Mkomazi Catchment hydrological indices describing the central tendency in different streamflow components are more useful than indices of the dispersion (*i.e.* variability) for explaining the majority of variation in the indices for the flow regimes in the LPWS zone type.

Indices for the Moderate Precipitation Water Source zone type

Five out of seven of the main streamflow characteristics of the streamflow regime are represented in the list of indices which are useful for characterising the "unpredictable flow and floods" regimes (*c.f.* Table 6.7) of the MPWS zone type (Table 6.9). In contrast to the importance of indices of the start of summer rainfall season months for characterising the LPWS zone type, researchers should focus on indices of the low rainfall season when assessing the flow regimes of the MPWS zone type. Indices M_A6 and M_A7 (average flows in March and April); M_A9 to M_A11 (average flows from June to August); M_A19 and M_A20 (variability of flows in April and May) as well as M_A24 (variability of flows in September) are all important for characterising the flow regimes of the MPWS zone type. However, many of these indices are not independent from each other. Given their high information nature, the variability associated with the 90-day minimum flow event (D_L12), the timing of the annual maximum flow (T_H1) and M_L4 (Q75) are important indices of the dominant patterns of hydrological variability for the flow regimes of the MPWS zone type. In general, once again indices of dispersion are less relevant than indices of the central tendency of flow conditions for explaining the variability associated with the flow regimes of the MPWS zone type.

Indices for the High Precipitation Water Source zone type

Table 6.9 indicates that the indices which could be useful to eco-hydrological studies of "seasonally predictable flow and floods" regimes (*c.f.* Table 6.7) of the HPWS zone type represent five of the seven main characteristics of the streamflow regime. As expected, indices related to the streamflow response in the main rainfall season (M_A4 to M_A6, average flows in January, February and March) as well as the low rainfall season (M_A8 and M_A9, average flows in May and June) and M_A24 (variability of flows in September) are all important for assessing the variation associated with the flow regimes of the HPWS zone type.

In addition, the 30-day maximum flow (D_H 4), overall variability (M_A25) and the timing of the minimum annual flow event (T_L1) are high information, non-redundant indices for characterising the flow regimes of the HPWS zone type. Indices of dispersion are less relevant than indices of central tendency for explaining the hydrological variability of these flow regimes. However, in addition to M_A24 and R_A4, variability in the 3-day minimum flow (D_L9) and the 30-day minimum flow (D_L11) is important.

Indices for the Main Natural Recapture zone type

The high information indices for the "predictable runoff" flow regimes (*c.f.* Table 6.7) of the MNR zone type represent six out of the seven main streamflow characteristics (*c.f.* Table 6.9). Indices of average monthly flow are much less relevant for the flow regimes in the MNR zone type than the flow regimes generated by any of the water source zone types. Indices of the duration of both low flow events (D_L1, D_L2 and D_L4, the 1-day, 3-day and 30-day minimum flows) and high flow events (D_H1, D_H4, and D_H5, the 1-day, 30-day and 90-day maximum flows) as well as the frequency of high pulse counts (F_H2) all contain important information regarding the variability associated with the flow regimes of the MNR zone type.

Again indices of the end of the low rainfall season are important (M_A11, M_A12 and M_A24, average flow in August, September and the variability thereof) as is M_L4 (Q75). In addition, M_L2 (variability in the IHA "baseflow" index) is a high information, non-redundant index of the variability associated with these flow regimes. Clearly, the low flow component of the predictable runoff flow regimes in the MNR zone type needs to be given careful consideration in studies of ecological flow requirements.

Indices for the Accumulated Natural Recapture zone type

Five out of seven of the main streamflow characteristics of the streamflow regime are represented in the list of high information indices which are useful for characterising the "sustained baseflow" regimes" of the ANR zone type (Table 6.9). While it may be expected that M_L3 (the Alt-BFI based on the Desktop Reserve Model index of short-term variability, and representing baseflow) should feature as a dominant index for the flow regimes of the ANR zone type, its omission from Table 6.9 confirms that, as stated above, there is little variability of this index among the individual zones comprising this Group. Consequently, M_L3 is not a particularly useful descriptor of the variability among the flow regimes of the ANR zone type.

Again, indices of average monthly flows are less relevant than they are for the flow regimes generated in any of the water source zones, but researchers could chose from M_A6, M_A10, M_A11 or M_A12 (average flows in the high flow month of March, and the generally low flow months of July, August and September). Similarly to the flow regimes of the MNR zone type, indices of the duration of low flow events are important (D_L2, D_L5, D_L9 and D_L11, *i.e.* the 3-day and 90-day minimum flows as well as the variability of the 3-day and the 30-minimum flows). The importance of D_H5 (the 90-day maximum flow) for the flow regimes in the ANR zone type is also shared with the flow regimes of the MNR zone type.

Indices representing the main facets of the streamflow regime

As shown in Table 6.9 and as described in the sections above relating to the high information indices for the different zones type, none of the zone types listed indices representing all seven of the main streamflow characteristics. To address this factor, and following Olden and Poff (2003), the PCA was also used to identify high information indices which represent all the major streamflow characteristics of magnitude, duration, timing, frequency and rate of change for each streamflow type. This was achieved by selecting indices with the highest loadings on each of the significant principal component axes *for each of the seven main characteristics of the streamflow regime*.

This procedure provides a greater selection of high information, non-redundant indices (*i.e.* for each zone type, seven times the number of significant principal components rather than just 12 spread across the total number of significant principal components) from which researchers could draw on for eco-hydrological studies, and again the procedure followed that identified by Olden and Poff (2003). Using the LPWS zone type as an example, for each of the seven major streamflow characteristics, the representative index with the highest loading on each of the first four principal components axes was selected, since the Broken Stick Model had identified four statistically significant principal component axes for this zone type (*c.f.* Table 6.8).

The results of this analysis are provided in Table 6.10, which shows groups of two to five high information indices for each zone type that are independent from each other and that represent the seven main streamflow characteristics. Where there are only a few indices in the dataset to represent a particular facet of the streamflow regime (*e.g.* the frequency of flow events, n = 4), there is high commonality of these indices across all zone types. However, it can be seen that the relevance of some high information, non-redundant indices is common across the whole set and most of the zone types.

Magnitude of average flow conditions

Several high information calendar month indices are specific to single zone types, for example M_A3 (December) for the LPWS Group, M_A5 (February) for HPWS Group, M_A11 (August) for MNR Group and M_A10 (July) for ANR Group. Using this method of selection, in general, indices of the dispersion of "monthly" indicators contain more information about the variability of the Mkomazi River and its tributaries than indices of the central tendency of monthly flows. In particular, M_A24 (representing the variability in flows in September) is a high information, non-redundant index for the upper and middle reaches of the Mkomazi Catchment, since it is shared among the MPWS and MNR zone types (Table 6.10). This emphasises the relevance of maintaining the natural streamflow variability associated with the first of season rainfall events. Moreover, M_A25 (index CDB, the Desktop Reserve Model index representing overall variability in the streamflow regime) is important for streams in the upper Mkomazi Catchment.

Magnitude of extreme flow or disturbance conditions

M_L4 (index Q75, *i.e.* the Desktop Reserve Model coarse index associated with the low flows required to maintain ecosystem functioning in drought conditions, *c.f.* Section 3.3 of this Chapter) is the most common, high information, non-redundant index of the magnitude

Table 6.10 Hydrological indices with the highest absolute loadings on each of the two to five statistically significant principal components for reference hydrological zone types in the Mkomazi Catchment. Indices are assigned to seven main streamflow characteristics in accordance with the largest loadings exhibited on each significant component. Superscripts denote the first to fifth principal components. Plain font denotes indices of central tendency (medians). Bold font denotes indices of dispersion (CD). Some indices are highlighted. Purple denotes indices used in the Desktop version of the South African Building Block Methodology (see text); blue denotes Colwell's indices of predictability; green denotes the seasonal predictability of flooding; yellow denotes the seasonal predictability of non-flooding.

Streamflow Characteristic	Zone Type					
	Water Source Zone			Natural Recapture Zone		
	Low Precipitation	Moderate Precipitation	High Precipitation	Mainstream	Accumulated	All 52 Zones
Magnitude of flow events						
Average flow conditions	M_A3^1 $\mathbf{M_A18^2}$ $\mathbf{M_A13^3}$ $\mathbf{M_A16^4}$	M_A10^1 $\mathbf{M_A24^2}$ $\mathbf{M_A17^3}$ $\mathbf{M_A17^4}$	M_A5^1 $\mathbf{M_A25^2}$ $\mathbf{M_A20^3}$ $\mathbf{M_A19^4}$ M_A16^5	M_A11^1 $\mathbf{M_A24^2}$ $\mathbf{M_A21^3}$	M_A10^1 $\mathbf{M_A15^2}$	M_A5^1 $\mathbf{M_A23^2}$
Low or high flow conditions	M_L4^1 M_H1^2 M_L3^3 $\mathbf{M_L2^4}$	M_L4^1 $\mathbf{M_L2^2}$ M_L3^3 M_H1^4	M_L4^1 $\mathbf{M_L2^2}$ $\mathbf{M_L2^3}$ M_L3^4 $\mathbf{M_L2^5}$	M_L4^1 $\mathbf{M_L2^2}$ $\mathbf{M_L2^3}$	M_L4^1 $\mathbf{M_L2^2}$	M_H1^1 M_L4^2
Duration of flow events						
Low flow conditions	D_L4^1 D_L7^2 $\mathbf{D_L10^3}$ $\mathbf{D_L11^4}$	D_L3^1 $\mathbf{D_L12^2}$ $\mathbf{D_L8^3}$ $\mathbf{D_L11^4}$	D_L5^1 $\mathbf{D_L12^2}$ $\mathbf{D_L11^3}$ D_L7^4 $\mathbf{D_L9^5}$	D_L2^1 $\mathbf{D_L14^2}$ $\mathbf{D_L11^3}$	D_L5^1 $\mathbf{D_L9^2}$	D_L5^1 D_L4^2
High flow conditions	D_H4^1 $\mathbf{D_H8^2}$ D_H6^3 $\mathbf{D_H9^4}$	D_H5^1 $\mathbf{D_H9^2}$ $\mathbf{D_H10^3}$ $\mathbf{D_H12^4}$	D_H4^1 $\mathbf{D_H10^2}$ $\mathbf{D_H13^3}$ $\mathbf{D_H8^4}$ $\mathbf{D_H7^5}$	D_H4^1 $\mathbf{D_H9^2}$ $\mathbf{D_H12^3}$	D_H5^1 $\mathbf{D_H12^2}$	D_H5^1 D_H6^2
Timing of flow events	T_A1^1 T_H3^2 $\mathbf{T_H2^3}$ $\mathbf{T_H3^4}$	$\mathbf{T_H2^1}$ T_A1^2 T_H3^3 T_H1^4	$\mathbf{T_L2^1}$ T_H3^2 T_L1^3 T_A2^4 T_H1^5	T_A1^1 T_H3^2 T_H1^3	T_A2^1 T_H3^2	T_A1^1 T_A2^2
Frequency of flow events	F_H1^1 $\mathbf{F_L2^2}$ $\mathbf{F_L2^3}$ F_L1^4	F_L1^1 $\mathbf{F_H2^2}$ F_H1^3 F_H1^4	F_H1^1 $\mathbf{F_L2^2}$ F_L1^3 F_H1^4 F_L1^5	$\mathbf{F_L2^1}$ $\mathbf{F_H2^2}$ $\mathbf{F_H2^3}$	F_L1^1 $\mathbf{F_L2^2}$	$\mathbf{F_L2^1}$ F_H1^2
Rate of change in flow events	R_A2^1 $\mathbf{R_A4^2}$ $\mathbf{R_A6^3}$ R_A1^4	R_A2^1 R_A3^2 $\mathbf{R_A4^3}$ R_A3^4	R_A2^1 $\mathbf{R_A5^2}$ R_A3^3 $\mathbf{R_A4^4}$ $\mathbf{R_A6^5}$	R_A1^1 $\mathbf{R_A6^2}$ $\mathbf{R_A4^3}$	R_A1^1 $\mathbf{R_A6^2}$	R_A1^1 R_A3^2

of extreme flow or disturbance conditions, being shared across all zone types. M_L2 (variability in the IHA "baseflow" index) is also common among all the zone types. Conversely, M_L3 (index Alt-BFI, the Desktop Reserve Model index representing the baseflow component, and relatively short-term variability in the streamflow regime) is an important index for all three water source zone types (*i.e.* LPWS, MPWS and HPWS).

Duration of low flow conditions

D_L11 (variability in the 30-day minimum flow) is the most common high information, non-redundant index of the duration of low flow conditions across the different zone types, being shared among all the water source zone types (LPWS, MPWS and HPWS) as well as the MNR Group. Using this method of selection, in general, the indices of the dispersion around low flow disturbance are more relevant than indices of the central tendency (median) for describing the dominant patterns of variation provided by the entire set of indices, across all zone types.

Duration of high flow conditions

D_H12 (variability in high flow pulses) is a common high information index for both the natural recapture zone types as well as the MPWS zone type, whereas D_H13 (the seasonal predictability of non-flooding) is dominant for only the HPWS Group.

Timing of flow events

T_A1 (Colwell's predictability index) is a high information index among the MPWS, LPWS, MNR and "All-52" zone types, indicating its usefulness for characterising the flow regimes across a wide range of environmental conditions of the Mkomazi tributaries and main river system. T_A2 (the constancy component of Colwell's index of predictability) is highly relevant for zones in the ANR Group and its dominance shares some commonality with the zones in the HPWS Group. This feature indicates the importance of maintaining natural constancy in the flow regimes at both extremes of the environmental gradient of the Mkomazi river system. While not a dominant index at the catchment scale, T_H3 is important across all the zone types indicating that seasonal floods are an important component of the flow regime(s) in the Mkomazi Catchment.

Frequency of flow events

Indices of the frequency of flow events are poorly represented in the dataset. However, there are still some salient points to be made from the analysis of high information indices of these flow events across the different streamflow types. Indices relating to the frequency of high flow pulses (F_H1 and F_H2), are common dominant indices across all the zone types with the exception of the ANR Group, where indices relating to the frequency of low flow pulses (F_L1 and F_L2) are important. Index F_H1 has more relevance than F_L1 to the water source zone types.

Rate of change in flow events

R_A1 (the rate of rising river levels) is a high information, non-redundant index of the rate of change in flow events across all zone types, with the exception of the MPWS and HPWS zone types. As dominant indices, R_A2 (the rate of falling river levels) and R_A4 (variability in the rate of rising river levels) both have commonality among the water source zone types, whereas R_A1 and R_A6 (rate of rising river level and the number of hydrograph reversals) have commonality between the natural recapture zone types. These features indicate the relevance of maintaining natural variability in fluctuating river levels in the main Mkomazi River which results from the streamflow response to rainfall events in the upper catchment and which cause the river level to rise.

General Observations

The main shortcoming of the zoning approach applied in this Study is that it assumes that all streams within a zone share the same affinity for the hydrological conditions and processes reflected by the hydrological information (*i.e.* the point-based *ACRU* records) at the outlet of the zone. Moreover, the application of a zoning approach in this Study assumes that all streams have the same affinity for the high information indices identified in the statistical analysis. This is a spatial scale-related problem which will be revisited in Section 4 of this Chapter. Nonetheless, point-based information does reflect the hydrological processes and conditions which take place over a longer river reach and at a wider scale and should contain some measure of longitudinal (upstream-downstream), lateral (channel to riparian) and vertical (channel to groundwater) hydrological connectivity (Richter *et al.*, 1998). In addition, the benefits of the zoning approach in providing a link between societal and ecological functioning generally compensate for the shortcomings. This feature is explored in Section 4.

3.7 Summary

Despite the usefulness of the streamflow record at the DWAF gauging station U1H005, hydrological modelling is required to generate reference hydrological conditions for the entire Mkomazi Catchment. The *ACRU* records of streamflows generated at the outlet of each of the 52 reference hydrological zones throughout the catchment were used to investigate the attributes of the different streamflow regimes across a gradient of environmental conditions. Five different reference hydrological zone types, based on hydro-geographical attributes, were identified for the Mkomazi River and its tributaries. PCA provided this information and highlighted minimum sub-sets of high information, non-redundant indices for each zone type which could be used in EFA to assess any incompatibilities between ecological water needs and societal water needs which may arise as a result of water resource related developments.

4 ASSESSMENT OF ENVIRONMENTAL CHANGE AS A RESULT OF HYDROLOGICAL ALTERATION

4.1 Introduction

The societal activities of commercial forestry, irrigation, dryland and subsistence agriculture interrupt the hydrological cycle as well as hydrological connectivity at different spatial and temporal scales (*c.f.* Chapter 4, Section 5.3.3). Although present water use in the Mkomazi Catchment is generally conservative and undemanding, the impacts of these societal activities on the generation of streamflows in the Mkomazi river system were found to be considerable on a local scale in the Mkomazi 2001 Study (Taylor, 2001). The Mkomazi 2001 Study highlighted the middle and lower Mkomazi tributaries as being susceptible to hydrological drought, even under present land use conditions.

However, while the Mkomazi 2001 Study conducted a pilot study of fine-resolution, temporal indices (*i.e.* in so far as relating to the IHA indices, *c.f.* Table 4.5 in Chapter 4) of the hydrological variability at four IFR Sites on the mainstream Mkomazi River (*c.f.* Figure 6.2) as a result of various potential upstream water developments, the study was not extended to a catchment-wide analysis of the impacts of potential water development at these fine time steps.

In this Section, the degree of hydrological alteration as a result of present land use is assessed for river reaches at the scale of the 52 reference hydrological zones and at relatively fine time steps. The change in the hydro-status of each of the 52 hydrological zones, as a result of present societal activities, is assessed from the alteration in the high information indices of the reference hydrological conditions identified in Section 3. Thus, this Section addresses Steps 2 and 3, "Assess ecological freshwater requirements" and "Assess societal freshwater requirements" of an application of the proposed framework for ecologically sustainable water resources management outlined in Chapter 4 of this thesis (*c.f.* Figure 4.9) for the Mkomazi Catchment.

4.2 Existing Hydrological Alteration at the Scale of Reference Hydrological Zones

Long-term hydrological records of streamflows generated under conditions of "present land use" contain information relating to the impacts of societal change on the streamflow regime. The *ACRU* records of "present land use" streamflows generated for the Mkomazi 2001 Study, simulated with the CSIR's land use classification defined in accordance with Thompson's (1996) land classification and the interpretation of the 1996 LANDSAT TM image (*c.f.* Section 2.3 of this Chapter, Figure 6.5), were revisited for each of the 52 "hydronomic zones". For each zone, the *ACRU* record of streamflows under reference hydrological conditions was compared with the *ACRU* record of present land use streamflows to assess societally induced, hydrological alteration, based on the differences in the hydrological indices (*c.f.* Section 3.5 of this Chapter) representing the streamflow regime of each time series.

4.2.1 Methods of assessing hydrological alteration

For each of the 52 zones, the time series of the *ACRU* record of present land use streamflows from 1952 to 1995 was appended to the *ACRU* record of streamflows under reference hydrological conditions from 1952 to 1995 (*i.e.* streamflows simulated with baseline land cover, defined as Acocks Veld types, Acocks, 1988) to form one time series spanning a hypothetical 88-year record. This allowed the same climatic and geophysical conditions to prevail over both records. The IHA model (Richter *et al.*, 1996) described in Chapter 5 was applied to perform a statistical analysis of the 88-year record, in which the *ACRU* reference conditions' streamflows were regarded as being generated under unimpacted land conditions (*i.e.* pre-societal development), whereas the *ACRU* present land use streamflows were regarded as being generated after land use change (*i.e.* post societal development). In this way it was possible to assess the degree of hydrological alteration that had occurred between the two time periods (Richter *et al.*, 1996). The hydrological alteration was calculated as:

((post-development – pre-development)/ pre-development) (Equation 1)

This calculation is performed routinely for the 33 indices comprising the IHA method by the IHA software (Smythe Scientific Software, Boulder, Colorado, USA) and provided as a "deviation factor" in the IHA output files. However, for the purposes of this Study, the values were recalculated for each of the 33 IHA indices to address the anomaly of point precision described in Chapter 5, Appendix 5A1. In addition, the hydrological alteration was calculated separately for the indices representing Alt-BFI and CDB (M_L3 and M_A25, *i.e.* the Desktop Reserve Model indices of short-term variability and of overall variability, *c.f.* Table 5.2 in Chapter 5), Q75, (M_L4, *i.e.* the Desktop Reserve Model coarse index associated with drought or stress conditions), Colwell's indices of predictability and of constancy (T_A1 and T_A2; Colwell, 1974), the seasonality of flooding and of non-flooding (T_H3 and D_H13; Poff *et al.*, 1989), and for the 33 indices of dispersion in the IHA indices (*c.f.* Chapter 5, Section 3.3.2.). Index HFI (M_H1, the index of high flow, *c.f.* Table 5.2) does not feature in the sub-sets of high information indices for any of the five different zones (*c.f.* Table 6.9) and was not included in this part of the study.

When the value of an index under post-development streamflows is equal to that under pre-development conditions, the hydrological alteration is zero. A positive deviation in the hydrological alteration indicates that the index under post-development conditions is higher than under pre-development conditions, whereas a negative deviation indicates that the index under post-development conditions is lower than under pre-development conditions.

The hydrological alteration values for each of the 52 zones are point-based, and as such only measure hydrological alteration in a temporal dimension rather than reflecting a spatial dimension of alteration at the scale of the zones. However, point-based information does reflect the hydrological processes and conditions which take place over a longer river reach and at a wider scale and should contain some measure of longitudinal (upstream-downstream), lateral (channel to riparian) and vertical (channel to groundwater) dimensions. The difficulty in expressing the information in different dimensions lies in determining how far upstream, or downstream, such conditions extend or influence streamflows. Richter *et al.* (1998) suggested a method to address this shortcoming, whereby they mapped the degree of alteration in both upstream and downstream river reaches between eight stream gauging stations on two major rivers in the upper Colorado River basin in Colorado and Utah in the USA, based on the average alteration (*i.e.* across the six IHA indices with the greatest alteration) detected within each river reach. In the method, Richter *et al.* (1998) extrapolated the hydrological alteration detected at a particular gauging station in both an upstream and downstream dimension until a confluence, impoundment or reach of higher alteration was encountered. The method devised by Richter *et al.* (1998) is adapted here for the 52 zones of the Mkomazi Catchment. The hydrological alteration detected at the outlet of each zone for each of the indices, based on the statistical analysis of the *ACRU* records and Equation 1, was assigned to one of seven classes in accordance with its value as indicated in Table 6.11.

Table 6.11 Classes assigned to the range of existing hydrological alteration in the Mkomazi Catchment

Hydrological Alteration (Value)	**Alteration Class (Degree of change)**
> 1.00	Substantial
0.81 – 1.00	Very Large
0.61 – 0.80	Large
0.41 – 0.60	Moderate
0.21 – 0.40	Little
0.01 – 0.20	Very Little
0.00	No change

From this dataset of hydrological alteration across the entire set of indices, an *average hydrological alteration* was assessed across the indices of high information (*i.e.* 12 indices, *c.f.* Section 3.5.4 of this Chapter) for (a) "All-52 zones" and (b) each of the five different reference hydrological zone types. These average scores were used to spatially map the hydrological alteration of the high information indices of the variability of the Mkomazi river system. First, the average hydrological alteration for all "All-52 zones" (*i.e.* the indices of high information of variability at a catchment-wide scale) was mapped for the Mkomazi catchment. This was achieved by mapping the class assigned to the average hydrological alteration at the outlet of each zone to include the accumulated upstream contributions of that particular zone. This method assumed that not only are all streams within a zone characterised by the same hydrological conditions (*c.f.* Section 3.5.4 of this Chapter) but also that they share the same hydrological alteration. However, given the relatively small areal extents of most of the zones this was considered to be acceptable.

Thereafter, the average hydrological alteration across the high information indices was assessed and mapped for each of the five different reference hydrological zone types. In contrast to the catchment-wide analysis described above, this second analysis represented an investigation of the alteration in the dominant indices of hydrological variability for each of the five different zone types. Again, this was achieved by mapping the class assigned to the hydrological alteration at the outlet of each zone to include the upstream contributions. However, in this instance the indices of high information used in the calculation of hydrological alteration differed across the catchment and were specific to each of the five zone types.

4.2.2 Results of the analysis

Alteration in the high information indices at a catchment wide scale

The zones with the least average hydrological alteration within each of the zones types and those with the greatest average hydrological alteration, as a result of present land use conditions, are provided in Table 6.12. In the following discussion the term "greatest hydrological alteration" is used to denote "the greatest of the averages of hydrological alteration among the high information indices", whereas the term "lowest hydrological alteration" is used to denote "the lowest of the averages of hydrological alteration among the high information indices". This distinction is made in recognition of the fact that the zones described may not experience either the greatest or the lowest overall, or even average, alteration *in all the indices*. However, they do experience either the greatest or

Table 6.12 Measures of existing hydrological alteration at a catchment-wide scale for selected hydrological zones in the Mkomazi Catchment

Zone type	Zone No	High Information Indices across "All 52-zones"												Average (Dimensionless)
		M_A3	M_A4	M_A5	M_A6	M_A7	M_A8	M_A25	D_H4	D_H5	D_H6	T_A2	T_H3	
HPWS	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	13	-0.04	-0.04	-0.04	-0.04	-0.06	-0.07	+0.10	-0.03	-0.03	-0.10	-0.09	0.00	0.05
MPWS	16	0.00	-0.04	-0.06	-0.04	+0.03	+0.05	-0.05	-0.04	-0.03	+0.13	+0.03	0.00	0.04
	26	-0.32	-0.21	-0.23	-0.26	-0.41	-0.65	+21.09	-0.16	-0.20	-0.54	+0.26	0.00	2.03
LPWS	39	+0.04	-0.02	+0.01	+0.13	+0.27	+0.30	-0.23	-0.07	+0.03	+0.03	+0.03	0.00	0.10
	43	-0.83	-0.85	-0.77	-0.79	-0.95	-1.00	+17.87	-0.48	-0.57	-0.21	+0.42	0.00	2.06
MNR	14	-0.02	-0.01	-0.02	-0.01	0.00	-0.02	+0.06	-0.01	-0.01	-0.03	-0.03	0.00	0.02
	35	-0.11	-0.08	-0.07	-0.06	-0.08	-0.11	+0.04	-0.07	-0.08	-0.33	-0.03	0.00	0.09
ANR	52	-0.12	-0.08	-0.10	-0.08	-0.06	-0.08	+0.02	-0.09	-0.09	-0.20	-0.03	0.00	0.08
	45	-0.12	-0.08	-0.10	-0.10	-0.09	-0.12	+0.08	-0.11	-0.10	-0.17	-0.03	0.00	0.09

Note: 1. A positive deviation indicates that the index under post-development conditions is higher than under pre-development conditions

2. A negative deviation indicates that the index under post-development conditions is lower than under pre-development conditions

3. Zero value indicates that there is no alteration.

4. Yellow shading indicates the zone within each zone type with the least average hydrological alteration

5. Green shading indicates the zone within each zone type with the greatest average hydrological alteration

6. Average values are based on the absolute value of the deviation in each index.

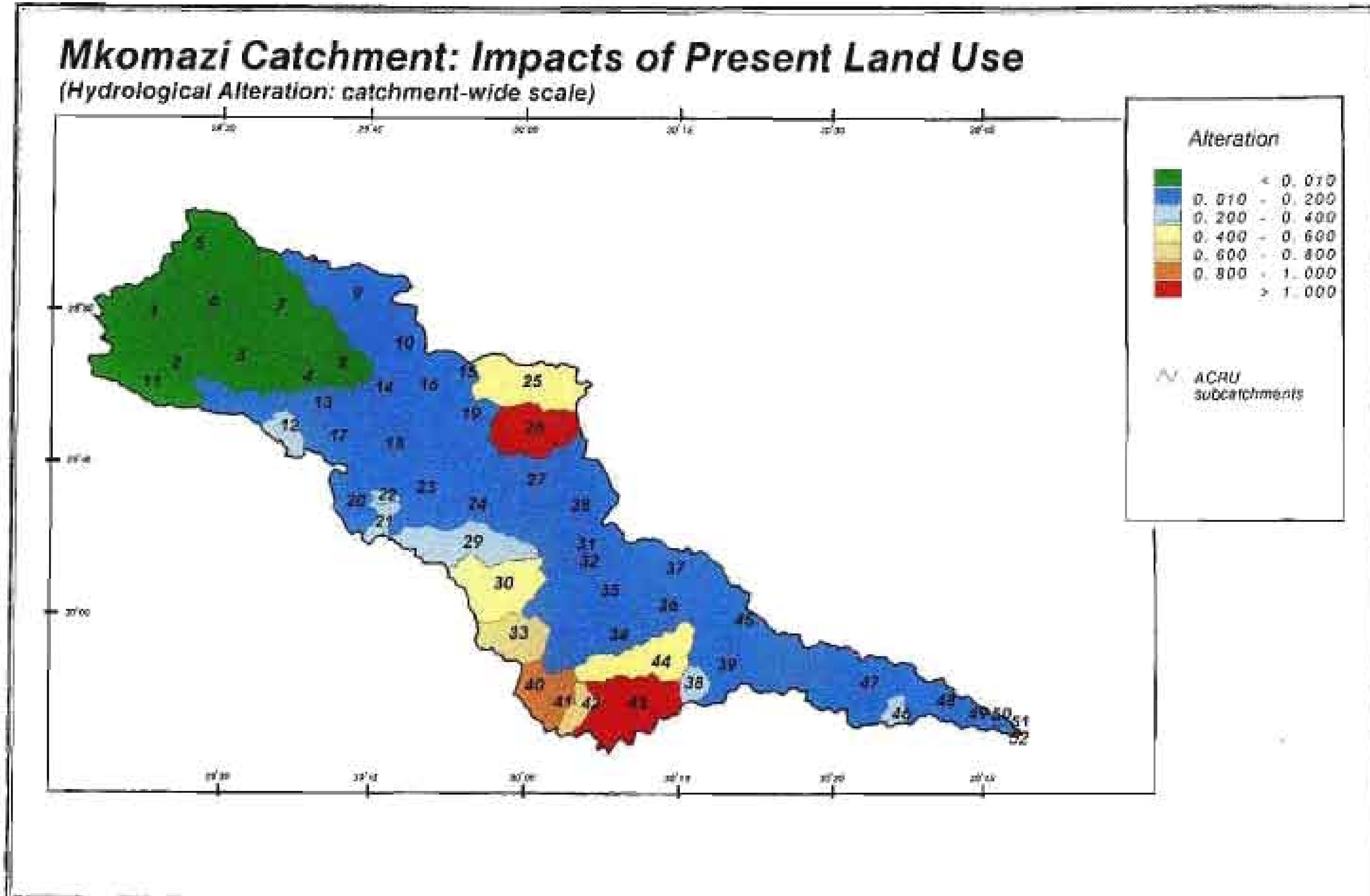


Figure 6.12 Alteration in the high information hydrological indices as a result of present land use conditions, across a catchment-wide scale in the Mkomazi Catchment

the lowest of the averages of hydrological alteration *in the high information indices*, as relating to the "All-52 zones".

Figure 6.12 indicates the degree of hydrological alteration among the 52 hydrological zones of the Mkomazi Catchment. While there is some evidence that upstream societal activity has an influence on the degree of alteration experienced by any downstream zone (*e.g.* Zones 26 and 25 rank at positions 2 and 9 in the greatest altered zones; Zones 43, 40, 41, 42 and 44 rank at positions 1, 3, 4, 5 and 8), many zones with no upstream societal activities also have relatively high alteration, *precisely because they have no upstream contributions*. These "high ranking" altered zones are located in either the LPWS or the MPWS zone type, and it appears that the impact of societal activity on the upstream-downstream relationship of hydrological connectivity is exacerbated by the climatic regime.

As expected, the greatest hydrological alteration is experienced along the tributaries of the Mkomazi. The greatest hydrological alteration is generally experienced in the zones where the reference conditions are characterised as being of the LPWS type, although extremely high alteration is also reported for Zone 26 (*c.f.* Figure 6.12 and Table 6.12), characterised by the MPWS type.

There is minimal hydrological alteration (range of 0.00 to 0.05) in the zones of the HPWS type as there is little societal impact in this part of the catchment. However, present land use has resulted in small reductions in some of the most variable streamflow components among the streamflow regimes found in the upper part of the catchment (*i.e.* average flows from December through May (M_A3 to M_A8), the 30-day and 90-day maximum flows (D_H4 and D_H5), the high flow duration (D_H6) and the seasonal predictability of the flow regime (T_A2). In particular, the extent of reductions in (a) average flows in the winter months (*e.g.* M_A7 at -0.06 and M_A8 at -0.07 for April and May respectively) and (b) the duration of high flow pulses has lead to a less seasonally predictable flow regime (T_A2) and an increase in the overall variability (M_A25) of the streamflow regime(s) in Zone 13 (*c.f.* Figure 6.12 and Table 6.12).

A similar hydrological state exists for the streamflow regime(s) in Zone 26 (*c.f.* Table 6.12), although the degree of alteration is much greater than is experienced by the streamflow regime(s) in Zone 13. In Zone 26, which forms part of the MPWS Group, 42% of the zone has been altered to cultivate a combination of commercial forestry and irrigation, with nearly equal areal extents of each being represented (*c.f.* Figure 6.5 and 6.6). The effect of these practices on the constancy of the flows (T_A2) and the overall variability (M_A25) of these streamflow regimes is pronounced. However, in this instance the streamflow regime(s) become more seasonally predictable and yet exhibit more overall variability. Conversely, the increases in (a) average flows in winter months and (b) the durations of high pulses in Zone 16 (also part of the MPWS Group) most probably result from the degraded grassland (*c.f.* Figure 6.5 and 6.6) since this is the major change in land use within the zone, and is a problem which is exacerbated by local rural settlement. As natural vegetation is dormant during winter months and the degraded grassland is likely to be denuded, the hydrological processes of evapotranspiration and vegetation interception will be omitted from the energy pathway (*c.f.* Figure 1.2) of hillslope processes and soils will tend to be compacted, with the effect that any rainfall that occurs in winter months would tend to generate overland type flow which quickly reaches the channel. In this way, the streamflow regimes(s) in this zone display slightly higher seasonal predictability in the flow regime and less overall variability. The MPWS Group exhibits the greatest *range* of hydrological alteration among the zones (0.04 – 2.03, Table 6.12). This factor emphasises that even for these relatively high rainfall areas of the catchment, societal activities can lead to substantial alteration of the natural variability of the flow regime(s) found there.

Although the *range* of hydrological alteration among the zones in the LPWS Group (0.10 – 2.06) is slightly less than those zones in the MPWS Group, the greatest hydrological alteration, in terms of absolute values, is experienced by the zones in the LPWS Group. This alteration need not be attributed solely to commercial cultivation practices, since land conditions in Zone 39 experience alteration from degradation of the natural grassland and some expansion of thicket and bushland, replacing the grassland (*c.f.* Figure 6.6). This emphasises the need for stewardship practices to conserve the natural environmental processes. In addition to the restrictions imposed on the energy pathways described above, substantially increased runoff in the end of wet season months, as a result of reduced infiltration and evapotranspiration, is likely to limit the off-stream ecological processes required to maintain the nutrient pathways that occur under natural conditions (*c.f.* Figure 1.2). Unsurprisingly, the overall variability (M_A25) of the flow regime(s) in Zone 39 is reduced. Conversely, the overall variability in Zone 43 is substantially increased as a result of societal activities. The streamflow regime in Zone 43 experiences the greatest hydrological alteration of all zones, although examination of present land use conditions (Figure 6.6) indicates that this deviation is largely attributable to changes in land use in the upstream zones. The streamflow regime(s) in Zones 42, 41 and 40 experience average hydrological alteration of 0.77, 0.81 and 0.91 respectively. As concluded in the Mkomazi 2001 Study, these zones are nearly fully utilised (*i.e.* "closed", *c.f.* Chapter 4, Section 5.2), with little available water for allocation (Taylor, 2001). Intensive irrigation is practised in the zones upstream of Zone 43, with a considerable number of small farm dams in operation. As a consequence, the streamflow regime(s) have become much more seasonally predictable (T_A2) and yet, any rainfall that occurs has the effect of increasing the overall variability (M_A25) of the streamflow regime(s).

There are reductions of flows for the entire length of the main Mkomazi River as a result of present land use. However, the degree of alteration in the indices of high information of the natural variability is relatively conservative for the zones in both the MNR and the ANR Groups. However, societal activities have a large influence on the duration of high flow events (D_H6) in these "natural recapture" zones, implying that the length of the flood season has already been somewhat reduced by upstream land use. While this is more pronounced for the zones of the MNR group (*e.g.* -0.33 for Zone 35), even Zones 45 and 52 located further downstream, experience a greater reduction in this flow component than the others. This feature has relevance for the assessment of the Ecological Reserve (ER) of the Mkomazi River since Zone 35 is also the location of IFR Site 3 (*c.f.* Figure 6.2). Attention to include the natural range associated with the duration of the high flow pulses would be beneficial to maintaining the natural variability of the streamflow regime in the natural recapture zones.

With the exception of Zones 26 and 43, the hydrological alteration in the constancy of the flow regime (T_A2) is similar across all zones on the main Mkomazi River. This is a reflection of the perennial character of the main Mkomazi River, a feature which is unimpeded by present societal activities in the catchment.

There is no change in the predictability of flooding (T_H3) for any of the zones in the Mkomazi Catchment as a result of present land use. However, rather than reflecting the effects of societal activities in the catchment, this feature is related to general climatic conditions. Index T_H3 is assessed as the "maximum proportion of all floods over the period of record that fall within any 60-day period" (*c.f.* Table 5.2). This indicates that climate patterns which govern the timing of the generation of flows in the water source zones have remained relatively constant over the time period in the model simulations (the 44-year period from 1952 to 1995).

Alteration in the high information indices at a zone type scale

The analysis of the degree of hydrological alteration in the indices of high information specific to each of the five different zone types revealed a similar pattern of deviation to the coarser catchment-wide analysis described above (*c.f.* Figure 6.13). However, the investigation of hydrological alteration in the streamflow regimes of the Mkomazi River network at this finer, zone type scale indicates that a greater number of zones can be classified as being "largely" to "substantially" altered by present societal activities in the catchment. Again, the greatest hydrological alteration among the selected indices is experienced along the tributaries of the Mkomazi, particularly in the zones where the reference conditions are characterised as being of the LPWS type. However, again the greatest alteration experienced by any zone is that reported for Zone 26 (*c.f.* Table 6.13), characterised by the MPWS type. Generally, the lowest alteration is reported for the zones of the HPWS type (range of 0 to 0.27), although both Zones 10 and 13 experience considerably more alteration than the remaining HPWS zones and more alteration than many of the MPWS zones. The reasons for the reductions or increases in any of the flow components are similar to those described above. Rather than repeat the same information, the remainder of this section compares and contrasts the two spatially different assessments of the degree of alteration of the Mkomazi Catchment. Thus only the salient points deduced from Table 6.13 will be discussed.

As a result of the finer spatial resolution of the second analysis, several indices appear in Table 6.13 which are absent from Table 6.12. The indices shown in Table 6.13 are those hydrological indices with the largest absolute loadings on each statistically significant principal component shown for each zone type in Table 6.9. Table 6.13 indicates that there is a greater reduction in the flows towards the end of the dry season months (*e.g.* -0.06 for June (M_A9) in Zone 10, characterised by the HPWS Group; -0.98 and -0.99 for July (M_A10) and August (M_A11) for Zone 26, characterised by the MPWS Group) than in the immediately preceding months. Moreover, there is increased inter-annual variability in

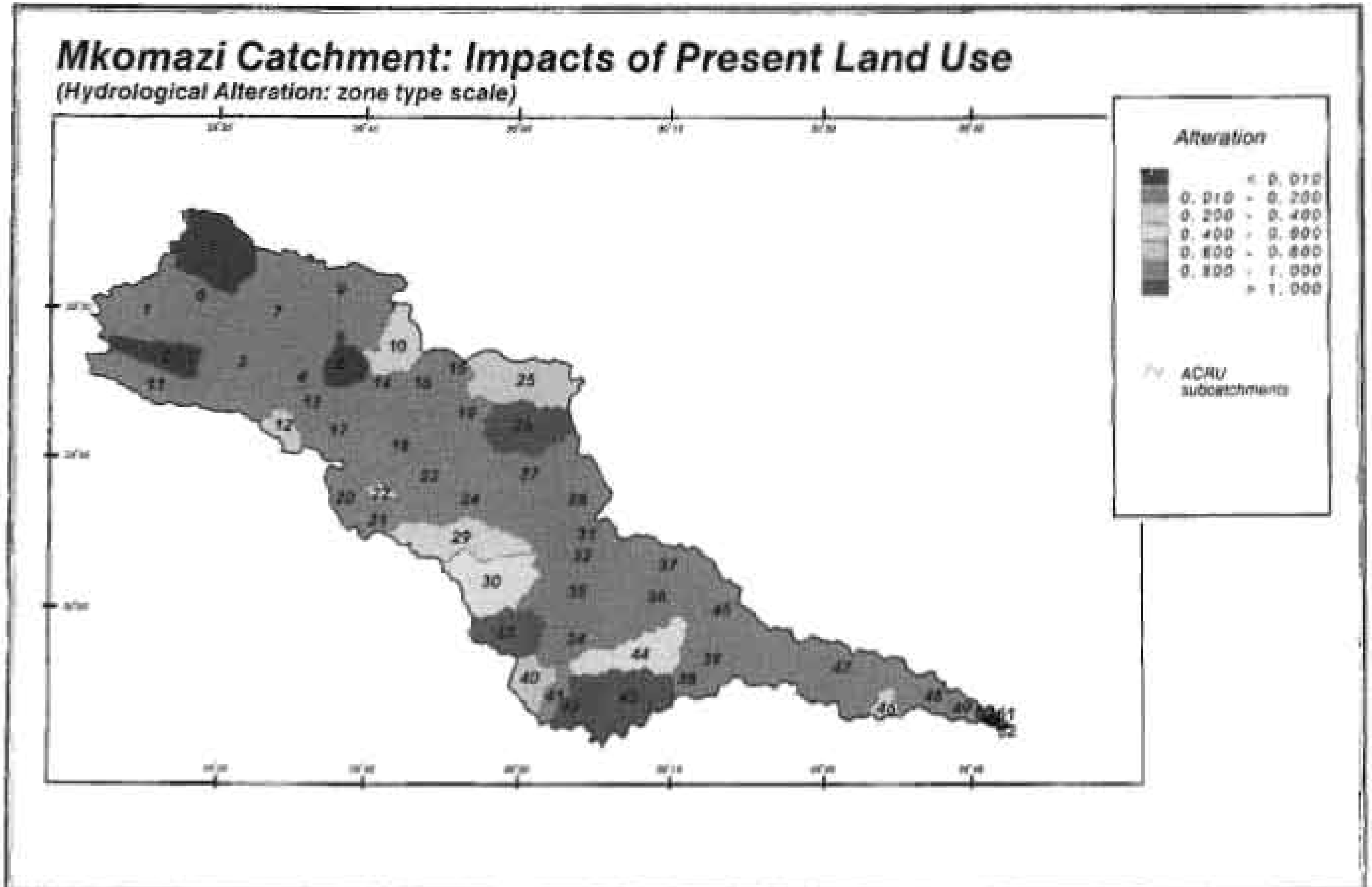


Figure 6.13 Alteration in the high information hydrological indices as a result of present land use conditions, across specific zone types in the Mkomazi Catchment

Table 6.13 Measures of existing hydrological alteration across specific zone types, shown for selected zones in the Mkomazi Catchment

Zone type	Zone No	High Information Indices												Average (Dimensionless)
		M_A4	M_A5	M_A6	M_A8	M_A9	M_A24	M_A25	D_H4	D_L9	D_L11	T_L1	R_A4	
HPWS	5	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.02	0.00	0.00	0.00
	10	-0.04	-0.04	-0.04	-0.05	-0.06	+0.24	+0.16	-0.03	2.12	0.38	-0.04	-0.09	0.27
		M_A6	M_A7	M_A9	M_A10	M_A11	M_A19	M_A20	M_A24	M_L4	D_L12	T_H1	R_A4	
MPWS	16	-0.04	+0.03	+0.04	+0.05	+0.01	-0.02	+0.04	-0.09	+0.07	+0.13	0	-0.05	0.05
	26	-0.26	-0.41	-0.78	-0.98	-0.99	+0.53	+1.53	+36.36	-0.99	+33.70	+0.29	-0.29	6.43
		M_A3	M_A4	M_A5	M_A13	M_L2	D_H2	D_H3	D_H4	D_H5	D_H6	T_H3	R_A4	
LPWS	39	+0.04	-0.02	+0.01	-0.19	-0.34	-0.12	-0.13	-0.07	+0.03	+0.03	0.00	-0.02	0.08
	33	-0.65	-0.64	-0.65	+36.88	-0.01	-0.30	-0.34	-0.43	-0.50	-0.34	0.00	-0.15	3.41
		M_A11	M_A12	M_A24	M_L2	M_L4	D_L1	D_L2	D_L4	D_H1	D_H4	D_H5	F_H2	
MNR	18	-0.06	-0.07	+0.05	0.00	-0.05	-0.22	-0.15	-0.08	-0.02	-0.02	-0.02	0.00	0.06
	35	-0.17	-0.19	+0.05	0.00	-0.15	-0.28	-0.24	-0.18	-0.05	-0.07	-0.08	+0.22	0.14
		M_A6	M_A10	M_A11	M_A12	D_L2	D_L5	D_L9	D_L11	D_H5	F_L2	R_A1	R_A2	
ANR	52	-0.08	-0.13	-0.18	-0.18	-0.19	-0.16	-0.01	+0.09	-0.09	-0.09	-0.42	+0.08	0.14
	45	-0.10	-0.18	-0.14	-0.22	-0.23	-0.16	+0.03	+0.17	-0.10	-0.35	-0.42	+0.12	0.18

Notes: 1. A positive deviation indicates that the index under post-development conditions is higher than under pre-development conditions
2. A negative deviation indicates that the index under post-development conditions is lower than under pre-development conditions
3. Zero value indicates that there is no alteration.
4. Blue shading indicates the zone within each zone type with the least average hydrological alteration
5. Purple shading indicates the zone within each zone type with the greatest average hydrological alteration
6. Average values are based on the absolute value of the deviation in each index.

these months. The reason for this is that both commercial forestry and irrigation are practised in these zones (*c.f.* Figure 6.6), with the result that there is greater difference between dry and wet years in winter months than experienced under reference hydrological conditions. The degree of alteration experienced in the variability of September flows (M_A24) in Zone 26 as result of these practices is considerable (+36.36), indicating that the natural variability in this flow component has been severely compromised. In addition, the reduction of flows in these low flow months as a result of commercial forestry and irrigation has led to increased variability in the duration of seasonal low flow conditions (*i.e.* +33.70 for D_L12 which is the variability in the 90-day minimum flow).

At this finer spatial resolution, Zone 33 surpasses Zone 43 as being the zone with greatest average hydrological alteration in the LPWS Group. Nearly 40% of this zone is allocated to commercial forestry, whereas nearly 20% is allocated to commercial irrigation (*c.f.* Figure 6.5 and 6.6). Table 6.13 indicates that even in wet months (M_A3, M_A4 and M_A5, *i.e.* December, January and February respectively) these practices have resulted in substantial reductions in the flow regime(s) in this zone. Again, the MPWS Group exhibits the greatest *range* of hydrological alteration among the zones (0.05 – 6.43, Table 6.13). This factor emphasises the benefits of knowledge of the natural streamflow regime at a local scale in order to make confident assessments of any potential alteration.

Indices which describe the rate of rise in streamflow conditions are important for describing the variability among the streamflow regimes in four of the five different zone types. The range of scores for R_A1 (-0.42 to -0.42, Table 6.13) indicates that all zones in the ANR Group (on the main Mkomazi River) experience moderately high reductions in the rate of rising river levels as a result of present land use. The variability of the rate of rise in streamflow conditions (R_A4) is a high information index for each of the water sources zone types. Table 6.13 indicates that in the tributaries of the Mkomazi there is a fairly wide range of alteration between the least impacted and most impacted zones within each of the Groups. Even in the HPWS zones, present land use has reduced the inter-annual variability of the rate of rise in streamflow conditions.

Table 6.13 indicates that there is no change in the timing of the 1-day minimum flow (T_L1) over the range of zones in the HPWS Group or in the variability associated with the IHA "baseflow" index (M_L2) over the range of zones in the MNR Group as a result of present land use, since in each instance the greatest and least impacted zones report zero change. However, examination of the remaining zones reveals otherwise in each instance. For example, while 10 of the zones in the HPWS Group report no change in T_L1, Zone 10 attracts a score of -0.05. While five of the zones in the MNR Group report no change in M_L2, Zone 24 attracts a score of +0.14. These features indicate that T_L1 is relatively insensitive to present land use in the zones in the HPWS Group, whereas M_L2 is relatively insensitive to present land use in the zones in the MNR Group. Conversely, M_L2 is moderately sensitive to present land use in the zones in the LPWS Group (range of -0.01 to -0.34, Table 6.13), with the greatest alteration being experienced by Zone 39. This feature emphasises the need for stewardship practices, even in the absence of any seemingly "streamflow reduction activity" (*c.f.* Figure 6.6).

Indices describing the duration of low flows are important for describing the variability among the zones in both the MNR and the ANR Groups. Table 6.13 indicates that there are modest reductions in the shorter minimum flow events (*i.e.* D_L1, D_L2 and D_L4, *i.e.* the 1-day, 3-day and 30-day minima) for zones in the MNR Group as well as in D_L2 and D_L5 (90-day minimum) for zones in the ANR Group. These reductions arise not only from present land use conditions within the zones in the main Mkomazi River, but also as a result of reductions in streamflows in the Mkomazi tributaries. Among the other indices most affected by upstream land use are F_H2 (inter-annual variability in the number of high flow pulses) for the zones in the MNR Group and F_L2 (inter-annual variability in the number of low flow pulses) for zones in the ANR Group. The increased variability in the frequency of high flow pulses (F_H2) in the MNR zones and the decreased variability in the frequency of low flow pulses (F_L2) in the ANR have mostly likely resulted in a flashy character to the lower Mkomazi River.

4.3 Summary

Analysis of the impacts of water development at relatively fine time steps and spatial resolution indicates that existing alteration of high information indices of variability within the different reference hydrological zone types varies greatly over the Mkomazi Catchment and river network. The greatest hydrological alteration is generally experienced in the zones where the reference conditions are characterised as being of the LPWS type. The least hydrological alteration is generally experienced in the zones where the reference conditions are characterised as being of the HPWS type.

With the exception of Zones 1, 5 and 33, a portion of the hydrological alteration experienced by each of the hydrological zones described above is attributable to upstream societal practices. This feature is inevitable when dealing with a hierarchical river network, particularly where the determination of the zonation has been influenced by societal practices. Rather than invalidating any of the discussion above, this situation emphasises the potential conflict over upstream-downstream water use and development that is common for any catchment where resources are scarce. However, the upstream-downstream hydrological relationship is complicated by climatic variation within the Mkomazi Catchment. The following Section addresses this dilemma by investigating "sub-hydronomic zones" discretised to represent the different societal land use activities described above and in Chapter 4, Section 5.3.4.

5 ECOSYSTEM GOODS AND SERVICES: TRADE-OFF POTENTIAL

5.1 Introduction

As stated Chapter 4, the provision and maintenance of ecosystem goods and services is cited as justification for the protection of catchment resources. The value of ecosystem goods and services, in both the short-term and the long-term, is the basis for ecosystem approaches to water management. However, in developing catchments such as the Mkomazi, stakeholders often lose sight of the long-term benefits (*e.g.* biodiversity and sustainability) in favour of the short-term benefits (*e.g.* timber and food production). There is a defined need for the benefits of ecosystem approaches to freshwater resources to be clearly demonstrated. The proposed framework for ecologically sustainable water resources management presented in Chapter 4, Section 5.4 is pursued in this Section to highlight the benefits of the hydronomic zoning approach for demonstrating that societal well-being and biodiversity are interlinked.

5.2 The Upstream-Downstream Relationship within the Mkomazi Catchment

The analysis in Section 3 of this Chapter identified different groups of reference hydrological zones at a catchment scale, whereas the analysis in Section 4 of this Chapter focussed on the present hydro-status within the different zones types. However, each analysis was performed on accumulated streamflows and included the impacts of upstream societal activities. An important area of collaboration among stakeholders is to understand not only how cumulative impacts of societal activity impact on the natural variability of streamflow regimes, but also how each individual activity may have different impacts.

As discussed in Chapters 1 and 2, river systems can be described in terms of inter-active pathways (Pringle, 2003) and energy transfer routes along one temporal dimension (for different time scales) and three spatial dimensions (*i.e.* in the longitudinal dimension from headwater to estuary; the latitudinal dimension from river channel to floodplain and vertical dimension from river channel through to aquifer, Ward and Stanford, 1989). Pringle (2003) contends that "consideration of the dynamic interactions along these four dimensions has proven to be a very effective conceptual spatial framework to understand human impacts on river ecosystems".

In this Section an even finer spatial or organisational resolution (hereafter referred to "organisational scale", *c.f.* Chapter 4, Section 5.3.3) than the reference hydrological zones is applied to the assessment of the different societal activities which operate in the Mkomazi Catchment in order to filter out the effects of the upstream alterations (*i.e.* the longitudinal spatial connectivity). Hydronomic "sub-zones" are defined within the reference hydrological zones, since this finer resolution represents the scale at which stakeholders are most likely to trade short-term benefits for longer-term sustainability. Consideration is given to the different impacts of societal activities on the hydrological cycle and is directed to the catchment to river relationship, focusing on precipitation as the basic water resource.

The impacts of each individual societal activity on the high information hydrological indices which characterise the streamflow regimes to different reference zone types are investigated. Highlighting the differences among the hydrological impacts of different societal activities at this organisational scale is expected to equip Mkomazi stakeholders with improved information relating to their water use within the Catchment. Thus, the investigation in this Section focuses on Steps 4 to 8 of the proposed framework in Chapter 4, Section 5.4.

5.3 Management Targets for Hydronomic Zones at a Fine, Organisational Scale

As illustrated in Figure 4.9 of Chapter 4, ecologically sustainable management of catchment water resources should address the freshwater requirements of both ecological and societal systems. The Indicators of Hydrological Alteration, IHA (Richter *et al.* 1996) were introduced in Chapter 4, Section 5.4 as a method for identifying any potential incompatibility or conflict between the ecological and societal needs for freshwater (Step 4 in Figure 4.9). Understandably, comparing ecosystem flow requirements with the streamflow regime resulting from meeting societal needs, by applying pre- and post-development change in the hydrograph as demonstrated by the IHA method is most effective when used in conjunction with ecological data relating to community structure, distribution and species life-cycles. A main thrust of this thesis is that humans should be regarded as an integral component of their environment, thus societal functioning and ecological functioning are interlinked. Thus, any river management strategies, or more pertinently catchment management plans, should be designed so that a compromise between societal and ecological functioning can be achieved.

As indicated in Chapter 4, defining societal freshwater needs is relatively straightforward when compared to assessing ecological freshwater needs. Matching the hydrological regime with the ecosystem goods and services that can be expected from any specific Ecological Reserve Category was highlighted in Chapter 4 Section 5.5 as being one of the main challenges to the implementation and management of the South African Reserve. While the Threshold Model for sustainability indicators (*c.f.* Figure 4.9, Godfrey and Todd, 2001) could be applied to set management targets based on empirical observations or knowledge of ecological response to different hydrological flow parameters, this approach becomes complex where ecological data are poor or non-existent.

The Range of Variability Approach, RVA (Richter *et al.*, 1997) was introduced in Chapter 4, Section 5.5 as a method for linking the gaps between applied river management and any theory of aquatic ecological functioning. The benefit of the RVA approach is that in the absence of ecological information, water managers can aim to maintain annual values of the different flow parameters within a pre-set target range which defines a portion, or all, of the natural range of variability in any particular flow parameter. This method for defining management (or restoration) targets prescribed on the basis of the natural variability in the streamflow characteristics of the 33 flow parameters of the IHA method is revisited in this Section, since it is appropriate for defining management targets for hydronomic zones at a relatively fine organisational scale.

5.3.1 Methods of analysis

The management targets

Where there is a paucity of ecological information the developers of the RVA recommend that preliminary management targets can be set, which can be adjusted when monitoring results and ecological observations become available (Richter *et al.*, 1997). The developers recommend that the target range is based on selected percentile levels, or a single multiple of the parameter standard deviations, for the natural or pre-development streamflow regime. It is not the intention that the river attain the target range every year, but rather that the target range is attained with the same frequency as occurred in the natural or pre-development streamflow regime (Richter *et al.*, 1998). The developers recommend that in the absence of any ecological information, the 25th to 75th percentile range is selected for preliminary targets, since attainment of an RVA target range defined by these percentile values of any particular parameter would be expected in only 50% of the years. Monitoring of the ecosystem response to the preliminary targets should identify critical flow thresholds for components of the river ecosystem and allow subsequent refinement of the flow-based management targets (Richter *et al.*, 1997).

Richter *et al.* (1998) define the degree to which the management target range is *not* attained as "a measure of the hydrological alteration" (*c.f.* Section 4.2 of this Chapter for a different measure of hydrological alteration, based on the *deviation* of the value of hydrological indices of central tendency, dispersion, predictability, seasonality and overall variability of pre-development conditions to post-development conditions). In accordance with Richter *et al.*, (1998) the hydrological alteration is calculated as:

((observed occurrences-expected occurrences)/expected occurrences) (Equation 2) where observed occurrences are the number of years in which the observed value of the hydrological parameter falls within the targeted range and expected occurrences are the number of years in which the value is expected to fall within the targeted range.

Hydrological alteration is equal to zero when the observed number of yearly occurrences of post-development values falling within the RVA target range equals the expected frequency. A positive value for the hydrological alteration indicates that the annual parameter values fell within the RVA target window more often than expected, whereas a negative value indicates that the annual values fell within the RVA target window less often than expected (Richter *et al.*, 1998). This second measure of hydrological alteration is the focus of the remainder of this Section.

In common with the first measure of hydrological alteration (Equation 1) described in Section 4.2, the calculation of Equation 2 is performed across the record by the IHA software (Smythe Scientific Software, Boulder, Colorado, USA) for the 33 intra-annual hydrological parameters comprising the IHA method (Richter *et al.*, 1996) and in this instance is provided as "hydrological alteration" in the RVA output files. However, in order to address the anomaly of point precision described in Chapter 5, Appendix 5B1, the scale factor in the "Set-up" dialogue box in the IHA application was used to work in units of $10^{-3}m^3.s^{-1}$ so that the smallest datum was in the order of 1. Again the values in the RVA output files were re-converted before applying the calculation of hydrological alteration. In addition to the 33 intra-annual hydrological parameters comprising the IHA method, the hydrological alteration associated with the annual values of Alt-BFI (based on the Desktop Reserve Model baseflow index) was also calculated across each record of the zones investigated.

Societal activities and influences

As described in Section 2.4 of this Chapter, present societal needs in the Mkomazi Catchment comprise the BHNR as well as agri-business. The Mkomazi 2001 Study concluded that "present" domestic water demand (*i.e.* the BHNR), together with livestock abstractions within the Mkomazi Catchment, has little consequence on the mainstream river, representing only 0.61% of annual accumulated flows even in the lowermost region of the catchment. While, it is anticipated that the domestic water demand of people living on the Mkomazi tributaries is likely to require a greater proportion of streamflows, particularly in low flow months, the present sparse population density in the Mkomazi Catchment suggests that the BHNR demand on catchment water resources is likely to remain relatively inconsequential in the near future. The study of water poverty and rural people is outside the scope of this thesis, since there are many contributing political, economic, environmental and societal factors to be considered besides the availability of water for direct human consumption. A study performed by the University of Southampton in England concluded that for a variety of reasons, including preference of taste and social interaction, rural inhabitants in the Mkomazi Catchment, often travel distances further than the nearest water tap for their daily requirements (INCO-DC, 2000). Thus, despite the hardship invoked by drought, rural inhabitants will travel further to seek adequate supplies of water. In addition, at the time of this Study (2005), there was no information available regarding the "dormant" proposals to construct the rural water supply sites (Nzinga, Bulwer, Gomane and Ngwadini) shown on Figure 6.2, indicating that at present there is no priority requirement for their development. Moreover, the BHNR will most likely be met from sources beyond the hydrological zones where people live. As such, the impacts of the BHNR are unlikely to have a substantial influence on the upstream-downstream relationship at the site where it is consumed. For these reasons, the impacts of the BHNR were omitted from the study on societal impacts conducted for this Section.

The main societal activities and influences in the Mkomazi Catchment are commercial forestry, commercial and subsistence dryland agriculture, commercial irrigation, degraded grassland, alien invasive riparian vegetation and encroaching valley bushland. The impacts of these activities and influences were investigated for each of the three water source zone types, namely the High Precipitation Water Source, Moderate Precipitation Water Source and Low Precipitation Water Source zone types described in previous Sections. Each of these societal impacts, or influences, was assigned to a "hydronomic sub-zone" in accordance with the classification of their disruption to the catchment hydrological processes (*c.f.* Chapter 4, Section 5.3.3). Thus, the different societal activities and influences were assigned as follows:

(a) commercial forestry was assigned to the ***Deep-rooted Streamflow Reduction sub-Zone***;

(b) commercial and subsistence dryland agriculture (referred to jointly as rain-fed agriculture) was assigned to the ***Short-rooted Streamflow Reduction sub-Zone***;

(c) commercial irrigation was assigned to the ***Supplementary sub-Zone***;

(d) degraded grassland was assigned to the ***Recession sub-Zone***;

(e) riparian alien vegetation was assigned to the ***Transmission sub-Zone*** and

(f) valley bushland (or thicket and bushland) was assigned to the ***Succession sub-Zone.***

In each instance the impacts of any particular societal activity or influence in a hydronomic sub-zone were compared with the "natural" or reference streamflow conditions of selected zones. The reference hydrological zones selected were those identified in Section 4.2.2 as having the least and the greatest alteration as a result of present land use conditions. The water use coefficients applied to the reference hydrological zones as well as those applied to the land use and cover in the hydronomic sub-zones were those used in the Mkomazi 2001 Study, which were taken from Schulze *et al.* (1999) and Schulze (2000). The values are provided in Appendix 6C.

As in Section 4, in the following discussion the term "greatest hydrological alteration" is used to denote "the greatest of the averages of hydrological alteration among the hydrological parameters assessed", whereas the term "lowest hydrological alteration" is used to denote "the lowest of the averages of hydrological alteration among the hydrological parameters assessed". This distinction is made in recognition of the fact that the hydrological zones described may not experience either the greatest or the lowest overall, or even average, alteration *in all the parameters*. However, they do experience either the greatest or the lowest of the averages of hydrological alteration *in the parameters assessed.*

The *ACRU* simulation model record of streamflows generated by any particular land use was compared with the *ACRU* record of streamflows generated under "natural" or reference hydrological conditions (*c.f.* Section 3 of this Chapter). This was facilitated by the model configuration of the Mkomazi 2001 Study (Taylor, 2001) shown in Figure 6.14, whereby each land use, or sub-zone, had been modelled discretely as a separate area within each of the 52 hydronomic zones. The land use categorisation shown in Figure 6.14 for two hydronomic zones was applied in the Mkomazi 2001 Study to each of the 52 zones, giving 468 linked hydrological units or sub-zones, each with their own input parameters. The benefit of this configuration is that the areal extent of each land use category, or sub-zone, and the parameters associated with different management practices, can be altered accordingly within the configuration when the hydrology of different land use scenarios is simulated without the need to reconfigure the model. For example, for assessment of the change in water yield from an increase in commercial forestry in a given hydronomic zone, the land use category of plantation (*ACRU* category 2 in Table 6.14) would be increased to represent the proposed areal extent. Correspondingly, the areal extent of the category representing the land use being converted to forestry (say, grassland, *ACRU* category 6 in Table 6.14) would be decreased by the equivalent areal extent.

A further advantage of isolating the different land uses into units or sub-zones is that, although the zonal climatic and soils characteristics are applied, the hydrological responses of each sub-zone are modelled distinctly as an entity. The routing / cascading of simulated streamflows resulting from the individual land use categories is also indicated in Figure 6.14. In particular, the zone discretisation and inter-sub-zone routing configuration used in the Mkomazi Catchment is indicated in Figure 6.15. The numbered configuration of the Mkomazi, by zone, is shown in Figure 6.7.

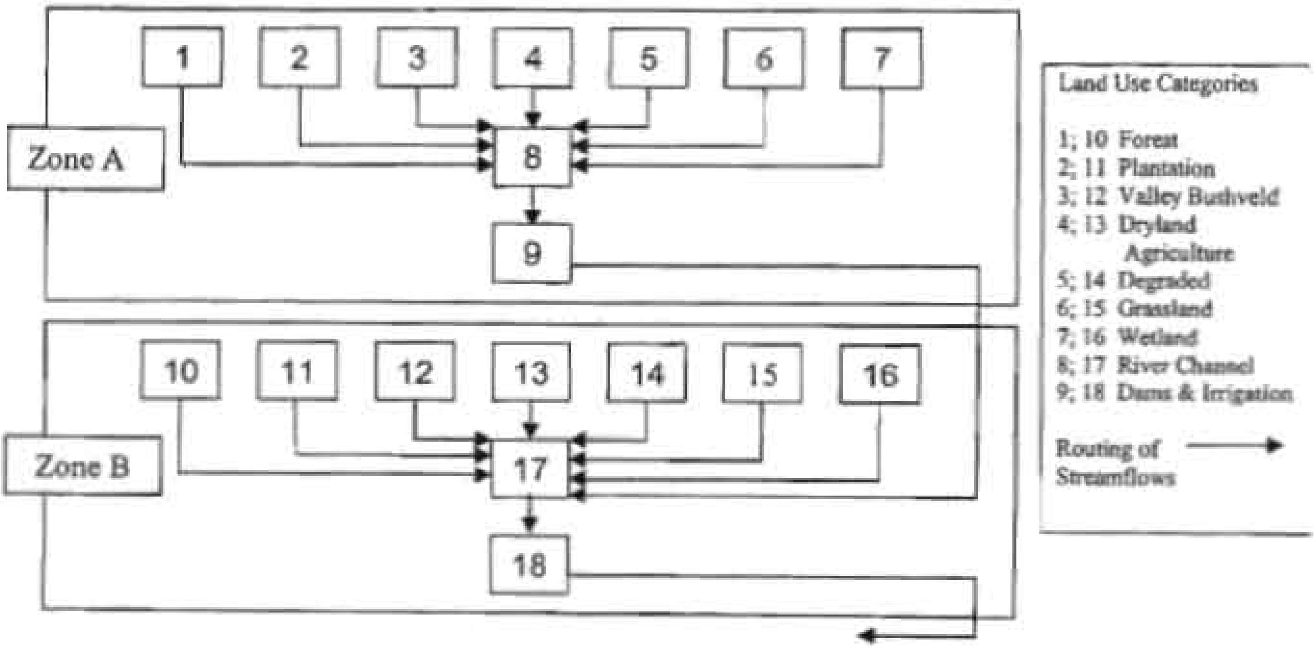


Figure 6.14 Schematic representation of the *ACRU* model hydrological zone and sub-zone configuration of the Mkomazi Catchment (modified from Taylor, 2001)

Notwithstanding the benefits of the model configuration in facilitating the "expansion" or "areal decline" of each of the land uses investigated, it was considered appropriate for this Study to proceed with the evaluation of *present land use conditions* on the streamflow regimes of the reference water sources zones. The areal extents of each land use investigated for the selected hydrological zones are provided in Table 6.15. Information relating MAP and altitude for each zone is provided in Table 6.1. It is apparent from Table 6.15 that there are differences among the areal extents of the land uses within each reference hydrological zone. However, the analysis of hydrological alteration in this section of the study is not compromised since all calculations are based on the relative differences of observed versus expected occurrences of yearly values of each of the hydrological indices investigated (*c.f.* Equation 2).

Table 6.14 Mkomazi Catchment: Land use categorisations (Taylor, 2001)

No	Categorisation used in *ACRU* model	CSIR (1996) land use classifications
1	Forest	Forest; Forest & Woodland
2	Plantation	Forest Plantation
3	Valley Bushveld	Thicket & Bushland
4	Dryland Agriculture	Cultivated permanent: commercial sugarcane; Cultivated temporary; commercial dryland; Cultivated temporary: semi-commercial / subsistence dryland
5	Degraded	Barren rock; Degraded: thicket & bushland; Degraded unimproved grassland; Urban / built-up land: residential; Urban / built-up land: residential (small holdings; bushland); Urban / built-up land: transport
6	Grassland	Shrubland & low fynbos; Unimproved grassland
7	Wetland	Wetland
8	Riparian	None identified
9	Dams and Irrigation	Improved grassland; Waterbodies; Cultivated temporary: commercial irrigated

However, the model configuration did contribute to some difficulties in the separation of the impacts of alien invasive riparian vegetation and the irrigated land use in each of the reference hydrological zones investigated. Figures 6.14 and 6.15 show that alien invasive riparian vegetation was allocated to the "riparian / channel" sub-zone in (*i.e.* the penultimate sub-zone within each zone, whereas irrigated crops and dam storage were both assigned to the sub-zone representing the outlet of each zone. In both instances, the assessment of hydrological alteration was problematic since in the present land use simulations, streamflows routed through these sub-zones included upstream contributions. For example, in the case of irrigation, upstream water was available to the plants in accordance to the irrigation scheduling invoked by the irrigation routine in the *ACRU* model.

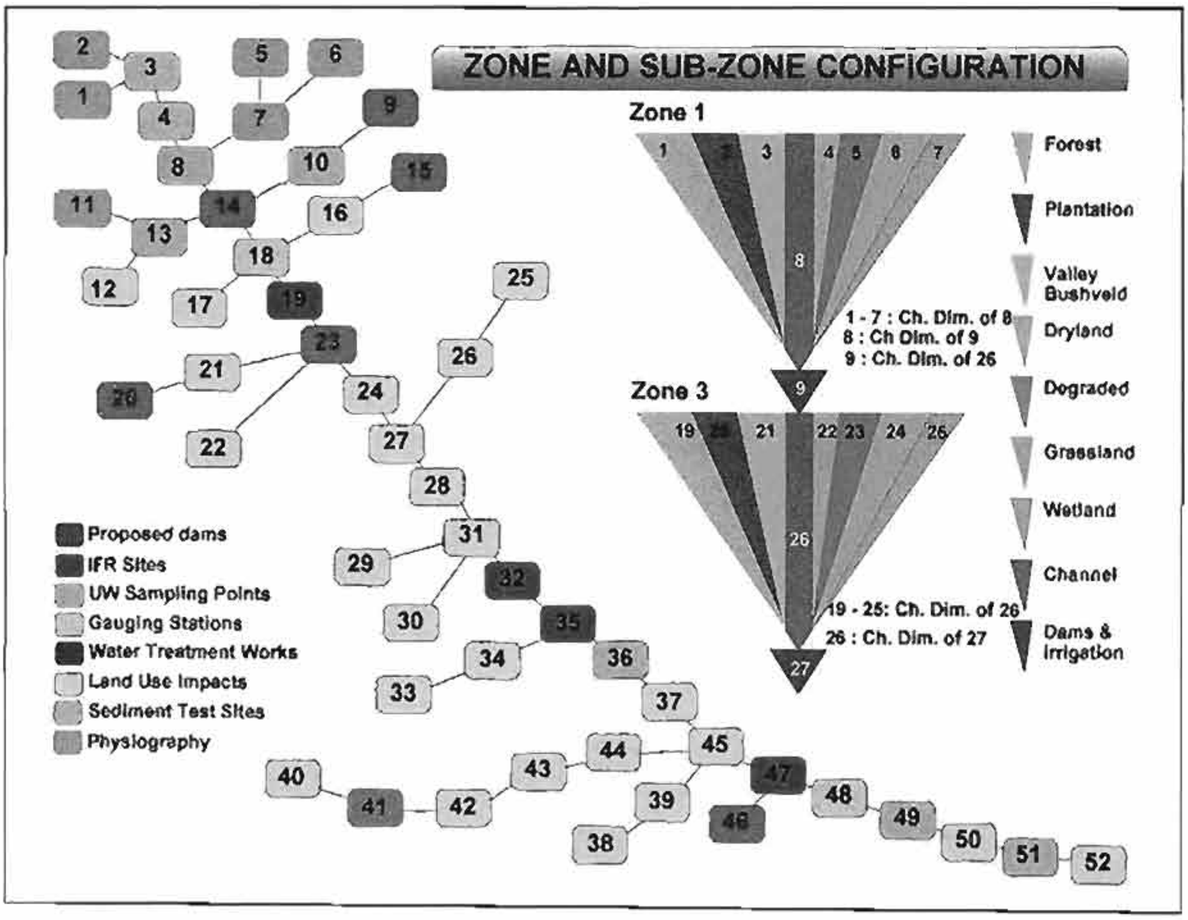


Figure 6.15 Schematic representation of the Mkomazi Catchment hydrological zones and inter-sub-zone configuration (modified from Taylor, 2001)

These criteria of the *ACRU* model routing/cascading presented a dilemma regarding an equitable comparison among the different societal activities and land uses within the zones of the Mkomazi Catchment. The problem of upstream contributions which incorporate changes to the streamflow regime as a result of upstream land uses was approached in the following manner for both the "channel" and zone "outlet" sub-zones:

Table 6.15 Areal extents of different land uses applied in the present land use simulations of the Mkomazi Catchment

Reference Zone		Hydronomic Sub-zone Type						
Type	No	Total Zone Area (km^2)	Interception		Supple-mentary (km^2)	Trans-mission (km^2)	Recession (km^2)	Succession (km^2)
			Deep-rooted (km^2)	Shallow-rooted (km^2)				
HPWS	5	142.97	0.00	0.00	0.00	1.08	0.00	0.00
	10	77.44	4.19	19.24	2.64	1.14	4.03	1.88
MPWS	16	70.94	0.42	1.99	0.00	0.92	8.52	7.12
	26	106.48	24.29	3.45	20.38	0.81	0.00	4.20
LPWS	33	76.07	29.90	2.64	13.38	0.69	0.00	0.41
	39	56.04	0.92	0.19	0.00	0.53	10.30	24.95

(a) Within each of the hydrological zones tested, the routing of the cells upstream from both the "riparian / channel" and the "outlet" sub-zone was adjusted so that upstream contributions of streamflows were "diverted" to downstream zones which were not accounted for in the analysis. This re-routing isolated the hydrological response in the "channel" sub-zone to that of the alien invasive riparian vegetation, whereas the hydrological response in "outlet channel" was limited to the impacts of the irrigative practice.

(b) Thereafter, the *ACRU* model was re-run to simulate the separate impacts of alien invasive riparian vegetation and of the dams and irrigation scenarios to generate daily records of streamflows for each hydrological sub-zone investigated.

(c) The next step comprised applying the same re-routing procedure to the *ACRU* reference record.

(d) Lastly, comparison was made between the *ACRU* reference hydrological condition records of streamflows and the records described in (b) to assess the hydrological alteration as a result of alien invasive riparian vegetation and of irrigation practices.

While this approach makes available to the plants (*i.e.* alien invasive trees in the "channel" sub-zone and irrigated crops in the "outlet" sub-zone) only the water that is generated within the sub-zone and, therefore, replicates the method applied to the other land uses investigated in this section, there are other considerations. Clearly, a river channel exists as a direct result of upstream contributions. Therefore, excluding upstream contributions of streamflows in the approach described above does not reflect reality. In addition, it has to be emphasised that in this specific configuration of the *ACRU* model, the alien invasive trees in the "channel cell" do not simulate the processes inherent to riparian vegetation in that neither the upstream stormflows, subsurface flows and baseflows which increase water availability, nor the channel transmission processes, are available to the trees by this routing approach. Nonetheless, the hydrological attributes relating to vegetative water use, interception loss, root mass in the soil profile as well as the coefficient of initial abstraction of riparian vegetation (wattle) are reflected in the sub-zone. Thus, while this may be an imperfect approach to simulating the impacts of alien invasive riparian vegetation *per se*, it does provide an indication of the likely impacts of this land cover on the components of the streamflow regime which contribute to the high information indices of the different zones within the Mkomazi Catchment. At present, this land cover has little societal, and far less ecological, use within the Catchment. Thus, any indication of the potential for trading the water use of alien riparian trees is likely to be beneficial to any catchment management plans.

In addition, in the case of irrigated crops, limiting the availability of water in this way does not truly reflect the practice of irrigation, since in reality irrigation would not be operated under such circumstances. The purpose of this Study is to investigate ways in which stakeholders can be empowered to make better decisions regarding their water use. Even considering only the short-term benefits of ecosystem approaches, this entails some measure of the volume of different ecosystem goods that could be expected under different scenarios. The net effect of adopting this "model-routing" approach whereby there are no upstream contributions to the irrigation scheduling as input to the *ACRU* model, would incur an irrigation deficit and most likely result in a very much reduced crop yield compared to the present land use conditions.

5.3.2 Results of the analysis

The results of the analysis of hydrological alteration, defined in this Section as the degree to which the RVA target range is not met for the selected intra-annual hydrological parameters, produced copious amounts of useful information regarding the different hydronomic sub-zones within each of the different reference hydrological zone types. For the sake of completeness, the following sections reproduce seasonal distributions of streamflows showing the hydrological alteration as a result of different land uses for each of the zones tested. However, only the hydrological alteration in those months featured in the relevant lists of high information indices will be discussed in any detail (*c.f.* Table 6.9).

In addition, in instances where a high information index is an index of dispersion, overall variability or seasonality rather than an index of central tendency, the results discussed in this section are restricted to those of the hydrological alteration of the annual values of the contributing parameter, since indices of dispersion, overall variability and seasonality have only one value over an entire record. However, it is to be expected that hydrological alteration in the contributing parameter will also impact on the more overall index. For example, the index M_A25 (CDB, the Desktop Reserve Model index of overall variability) is an important index for describing the variability of the streamflow regimes in the HPWS zone type (*c.f.* Table 6.9). In this instance the hydrological alteration in the annual values of M_L3 (Alt-BFI, one of the contributing components of M_A25), as a result of different societal activities, is investigated for Zones 5 and 10 (experiencing the least and the greatest existing alteration respectively) in this zone type (*c.f.* Table 6.13). Unfortunately, it was not possible to perform a similar investigation for the index T_H3 (the seasonality of flooding) which is important to the LPWS zone type, since it is based on general conditions over the entire record period.

The following sections reproduce the hydrological alteration attributable to each land use examined and show the extent to which annual values of the hydrological parameters under post-development conditions fell within three RVA categories, *viz*:

(a) *below the RVA middle category*, for cases when the post-development annual value was less than the 25th percentile value of annual values under pre-development conditions,

(b) *within the RVA middle category*, for cases when the post-development annual value was within the target range of the 25th to 75th percentile,

(c) *above the RVA middle target*, for cases when the post-development annual value was greater than the 75th percentile of annual values under pre-development condition.

It must be noted that these three RVA categories are not of equal size. Categories (a) and (c) both represent one quarter of the annual values, whereas category (b) represents one half of the annual values. Hence, the degree of hydrological alteration within any of the three categories (below, above and within the RVA target range) has to be considered simultaneously with the alteration within each of the other categories.

The hydrological alteration of annual values of the hydrological parameters contributing to the high information indices (*c.f.* Section 3.5.4) is reproduced in the following sub-sections for the zones identified in Section 4.2 as having the least and the greatest existing hydrological alteration (for median and general conditions across the 44-year record at the zone scale) within each water source zone type. The information depicted in the graphs of hydrological alteration in the following sub-sections was interpreted by the degree of non-attainment of annual values of the hydrological parameters incurred under the different land uses. Table 6.16 provides a general summary of the interpretation.

Table 6.16 Summary of the interpretation of the graphs depicted in the analysis of the degree of Hydrological Alteration of selected parameters of the streamflow regimes in the Mkomazi Catchment

RVA Category	Direction of Alteration	Interpretation of Occurrence
Below RVA middle category	Positive	Annual values fall below the RVA target window of the 25th to 75th percentile of values more often than expected (*i.e.* in more than 25% of post-development years)
	Negative	Annual values fall below the RVA target window of the 25th to 75th percentile of values less often than expected (*i.e.* in less than 25% of post-development years)
Within RVA middle category (the RVA target window)	Positive	Annual values fall within the RVA target window more often than expected (*i.e.* in more than 50% of post-development years)
	Negative	Annual values fall within the RVA target window less often than expected (*i.e.* in less than 50% of post-development years)
Above RVA middle category	Positive	Annual values fall above the RVA target window of the 25th to 75th percentile of values more often than expected (*i.e.* in more than 25% of post-development years)
	Negative	Annual values fall above the RVA target window of the 25th to 75th percentile of values less often than expected (*i.e.* in less than 25% of post-development years)

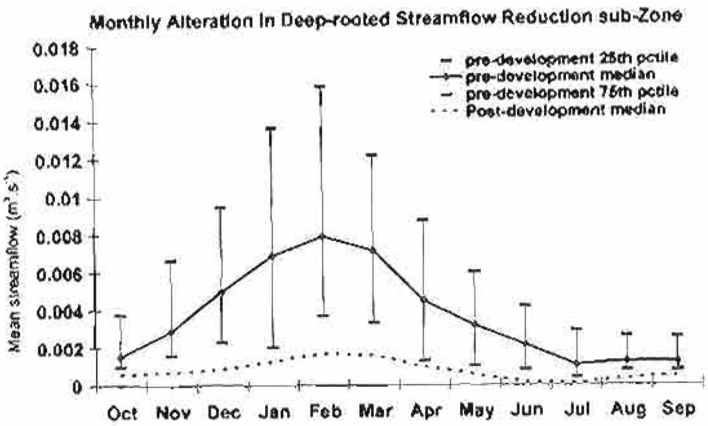
5.3.2.1. Hydrological alteration in the flow parameters contributing to the high information indices of the LPWS zone type

The subset of high information indices of the variability among the streamflow regimes in the Low Precipitation Water Source zone type was shown in Table 6.9 to comprise M_A3 to M_A5 (average flows in December, January and February), M_A13 (average flows in October), M_L2 (variability in the IHA "baseflow" index), D_H2 to D_H5 (the 3-day, 7-day, 30-day and 90-day maxima), D_H6 (the duration of the high flow pulse), T_H3 (seasonality of flooding) and R_A4 (variability in the rise rate of streamflows). Zone 33 was identified as having the greatest existing hydrological alteration (at the zone scale, and for general and median conditions) from reference conditions among the zones in the LPWS zone type. Zone 39 was identified as having the least existing alteration among the zones in this Group.

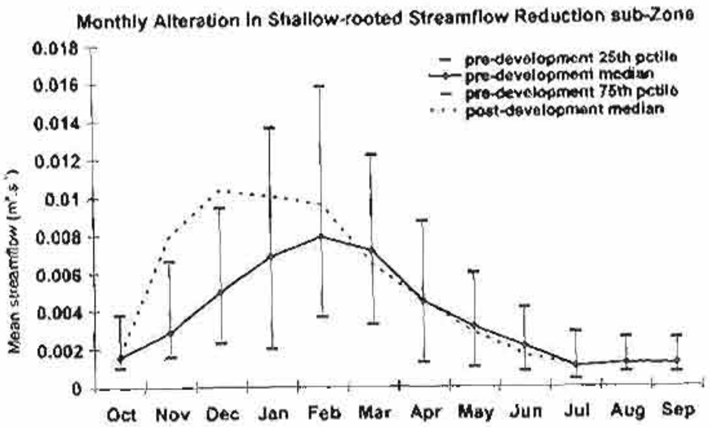
Monthly alteration in Zone 33

The seasonal distribution of streamflows resulting from the change in land use from reference hydrological conditions in Zone 33 is shown in Figure 6.16 for commercial forestry, rain-fed agriculture, commercial irrigation, alien invasive riparian vegetation and thicket and bushland. These land uses have been allocated to the Deep-rooted Streamflow Reduction, Shallow-rooted Streamflow Reduction, Supplementary, Transmission and Succession sub-Zones respectively (*c.f.* Chapter 4, Section 5.3.4 and this Chapter, Section 5.3.1). As indicated in Table 6.15, the areal extents of these sub-zones differ. In order to show comparable alterations, the graphs in Figure 6.16 depict the monthly alteration that would occur if only 1 km^2 was developed from reference conditions.

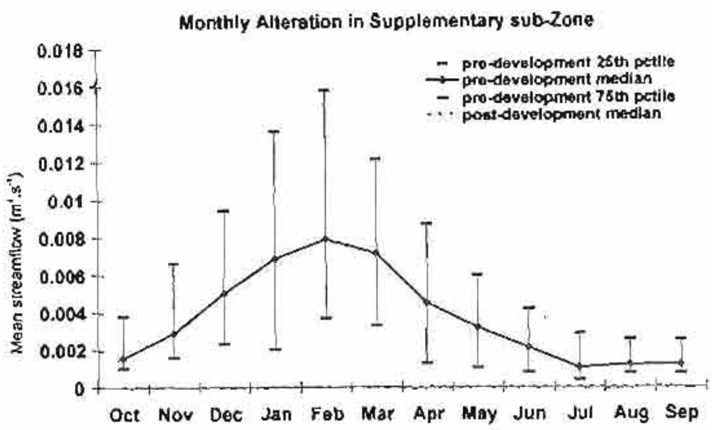
The monthly alteration in Zone 33 varies considerably among the sub-zones, with the greatest reductions in streamflow arising from the practice of commercial irrigation (Figure 6.16c). According to the *ACRU* model simulations and the methods applied in Section 5.3.1 of this Chapter to evaluate the water use of irrigation in this Low Precipitation Water Source Zone, this agri-business depletes any available streamflow in an average year. Zone 33 has no upstream zone contributing to farm dam storage. Consequently, this practice requires contributions from other sub-zones, either from within Zone 33 or from neighbouring zones (*e.g.* Zone 35 on the mainstream Mkomazi River, *c.f.* Figure 6.13).



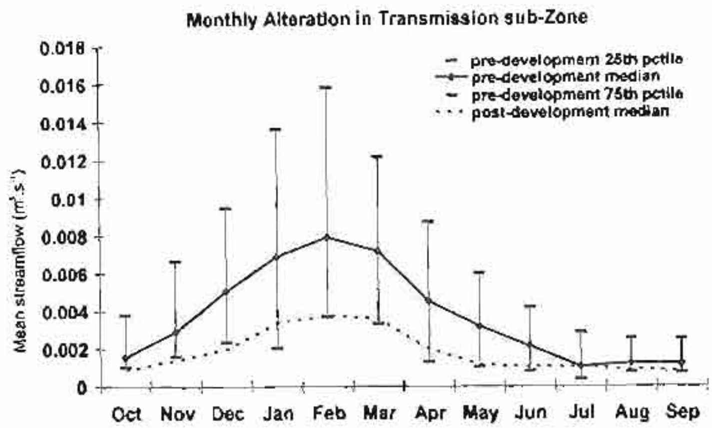
(a)



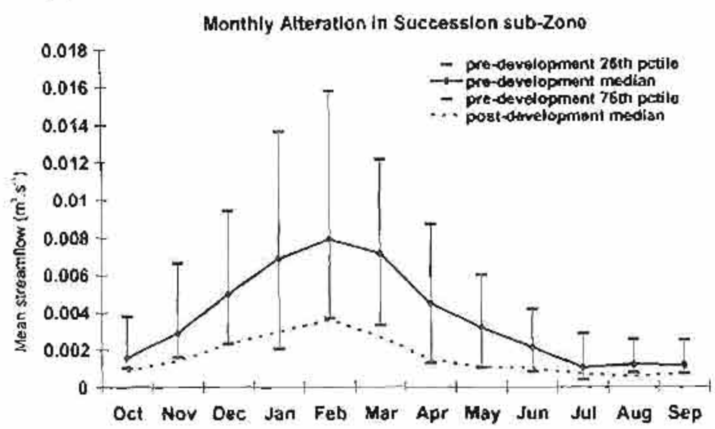
(b)



(c)



(d)



(e)

Figure 6.16 Monthly alteration of the seasonal distribution of streamflows of Zone 33 in the Low Precipitation Water Source Zone of the Mkomazi Catchment as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) commercial irrigation, (d) alien invasive riparian vegetation and (e) thicket and bushland. All values are based on alteration of transformation of an assumed 1 km^2 from reference hydrological conditions.

In an average year, commercial forestry reduces average monthly streamflows to below the 25th percentile of values under reference conditions in each month of the year, although the greatest *absolute* alterations occur in the wetter summer months (Figure 6.16a). Likewise, in an average year, both alien invasive riparian vegetation and thicket and bushland reduce average monthly streamflows to more or less the 25th percentile of values under reference hydrological conditions for most months of the year (Figures 6.16d and 6.16e). The similarities in water use between these two opportunistic land covers are evident from Figures 6.16d and 6.16e, with the main difference being that thicket and bushland in this zone utilises more water at the end of the high flow season (March and April) whereas alien invasive riparian vegetation uses more water in the low flow season (August to September).

According to the *ACRU* model menu for the Mkomazi, the rain-fed crop in Zone 33 is maize. It has been suggested in South African water resources management that commercial farming of this crop should be declared as a Streamflow Reduction Activity (SFRA). However, this proposal appears to be misplaced. Figure 6.16b indicates that, according to the *ACRU* model simulations there is additional streamflow generation in this zone when compared to reference hydrological conditions at the start of the growing season for this crop. This is as a result of patches of bare soil replacing the naturally occurring grassland cover. While there may be increased soil moisture loss from the upper soil profile, these patches of bare soil generate more stormflow during rainfall events than would occur under natural conditions. As the growing season progresses these patches reduce in size and the crop plant evapotranspires at a greater rate than the grassland it replaced. By the end of the growing season (March) in an average year, average monthly streamflows under rain-fed agriculture are lower than would be expected under natural conditions. After the plants have been harvested, there is little difference between the two land covers in the low flow winter months as a result of climatic conditions (*e.g.* absence of rainfall on the fallow land as apposed to the dormant grassland).

Hydrological alteration in Zone 33

The degree to which the RVA target range (defined by the 25th to 75th percentile of pre-development values under reference hydrological conditions) is not attained for selected hydrological parameters of the streamflow regime in Reference Zone 33, as a result of different societal activities, is shown in Figure 6.17 for the sub-zones described

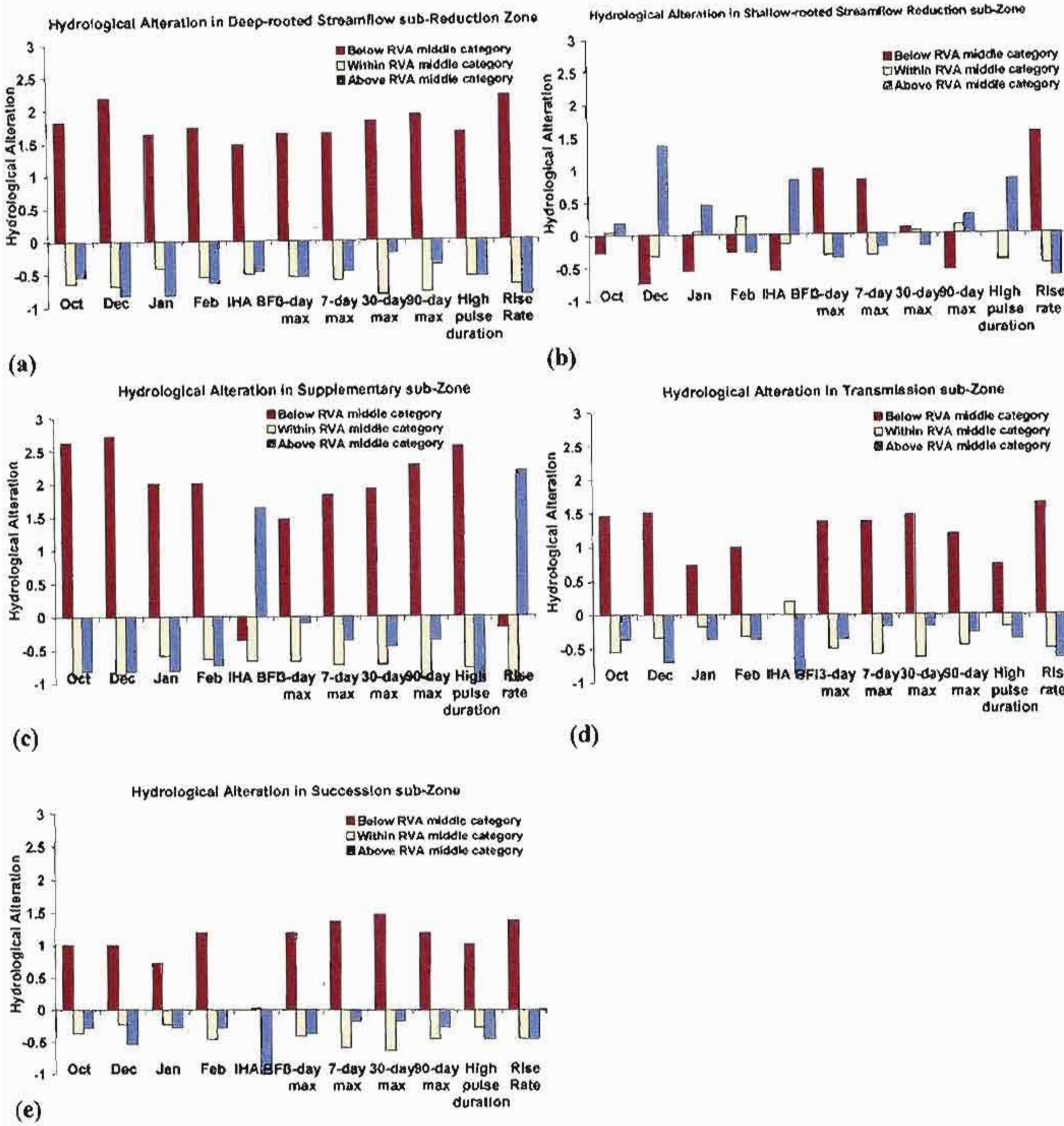


Figure 6.17 Hydrological alteration of selected hydrological parameters of the streamflow regime in Zone 33 as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) commercial irrigation, (d) alien invasive riparian vegetation, (e) thicket and bushland. Hydrological parameters selected on their contribution to the subset of high information indices for the Low Precipitation Water Source Zone Type (of which Zone 33 has the greatest existing alteration) within the Mkomazi Catchment.

immediately above. Unlike the analysis of the seasonal distribution of streamflows described above, the graphs in Figure 6.17 represent hydrological alteration resulting from the "actual area" *i.e.* the areal extent allocated to each of the different activities as shown in Table 6.15. However, in this instance the areal extent is inconsequential[1] since the graphs, which all result from the calculation of the hydrological alteration in Equation 2 (*c.f.* Section 5.3.1), are comparable to each other.

While there are differences among the degree of hydrological alteration as a result of the different land uses, there is some similarity between the impacts of commercial forestry, alien invasive riparian vegetation and thicket and bushland on the selected hydrological parameters. With the exception of the IHA baseflow parameter, the greatest hydrological alteration for each of these land covers lies in the number of years in which the values of the flow parameters are less than their respective RVA target window, as a result of considerable reductions to the annual range of variation in streamflow magnitude (*c.f.* Figure 6.17a, 6.17d and 6.17e). For each of these land uses, the RVA target window is achieved in less than 50% of the post-development years for virtually all of the flow parameters. While there is a decline in the baseflow regime (as derived by the IHA "baseflow" index[2]) as a result of both alien invasive riparian vegetation and thicket and bushland, the RVA target window is still met in more than 50% of the years (*c.f.* Figure 6.17d and 6.17e). Nonetheless, the impacts of commercial forestry (*c.f.* Figure 6.17a) are greater than those of both alien invasive riparian vegetation and of thicket and bushland and can be attributed largely to the different tree species "planted", in the respective sub-zones, with pine, riparian wattle and small to medium sized trees and shrubs occupying the Deep-rooted Streamflow Reduction, Transmission and Succession sub-Zones respectively. In accordance with the *ACRU* model input, pine trees have higher hydrological attributes (*e.g.* water use coefficient and interception loss) than riparian wattle. In turn, riparian wattle has higher hydrological attributes than thicket and bushland, according to the *ACRU* model input.

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[1] Caution should be applied when analysing the hydrological alteration in the parameters relating to low flow conditions (*i.e.* low flow season months; the minimum and shorter multi-day minima extremes; low flow pulse counts and their durations; number of days with zero flow; the IHA baseflow and the date of minimum flow event) generated from small areal extents as a result of anomalies in point precision from model output files.

[2] The author recognises that there is likely to be some anomaly regarding the hydrological alteration in the IHA baseflow parameter for land uses with small areal extents (see Footnote 1 above).

The greatest hydrological alterations among the different land uses are shown for commercial irrigation (Figure 6.17c) which results in substantial reductions in the annual range of variation in the magnitude of streamflows. However, unlike the impacts of commercial forestry and of alien invasive riparian vegetation, there is a substantially elevated "baseflow component" (as calculated by the IHA "baseflow" parameter) accompanied by increases in the rate of rising streamflow level as a result of commercial irrigation (Figures 6.18a and 6.18b). Just as the Desktop Reserve model baseflow index, BFI, (Hughes and Hannart, 2003) does not represent any catchment generating mechanism (*c.f.* Chapter 5, Section 3.3.2), neither does the IHA baseflow index, which is calculated as the proportion of the 7-day minimum flow to the annual mean of daily flows. The reason that the values of this index are so much higher due to commercial irrigation results from substantial reductions in the mean daily flow for the year when compared to reductions in the 7-day minimum flow. The substantial increase in the rate of rising streamflows results from differences in antecedent moisture conditions under irrigative practices when compared with "natural conditions". Supplementary water is available to the plants in accordance with the irrigation scheduling incorporated in the *ACRU* model with the effect that for many months of the year the surface soil layer is wetter than under "natural conditions". These features represent substantial alterations from the natural range of variation in the both the baseflow regime and the rate of change in streamflow regime. Conversely, the duration of the high flow pulses is now much less variable, with most years experiencing much shorter pulses (*c.f.* Figure 6.17c).

Rain-fed agriculture has the effect of moderately increasing the magnitude of streamflows at the start of the growing season (October to January), thereby elevating the "baseflow component" (according to the IHA measure of the baseflow regime), and increasing the variability of the duration of high flow pulses (Figure 6.17b). The shorter maxima streamflow events all experience small reductions, whereas in low flow years, the decline in rising river level is considerable. In general, there is a greater likelihood of the RVA target window being attained in at least half the post-development years under this land use than any of the others. Nonetheless, the impacts of rain-fed agriculture on the streamflow regime of this LPWS zone are more complex than those of other land uses. Increases over the natural range of the selected streamflow parameters are just as detrimental to natural hydro-ecological functioning as are reductions.

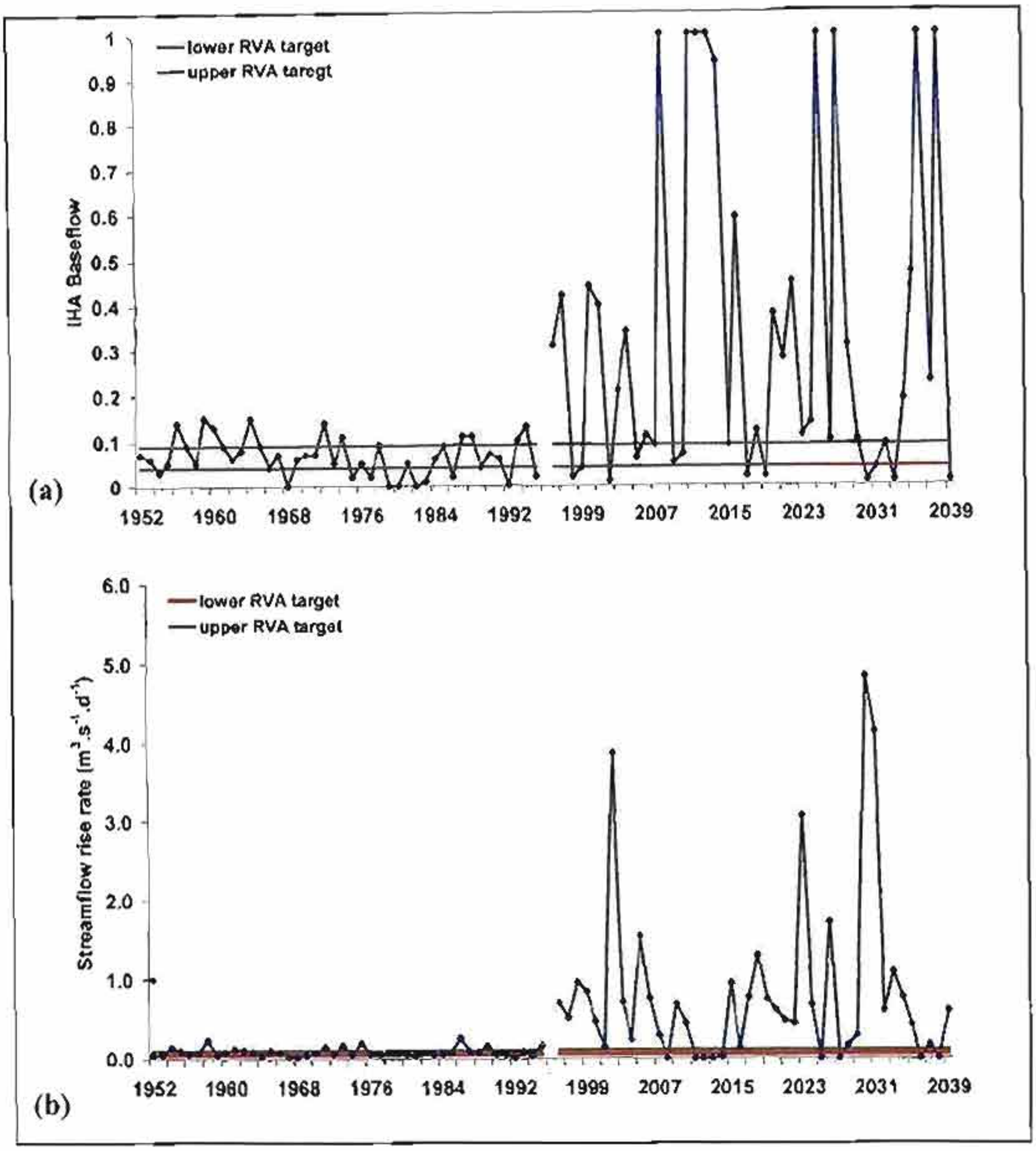

Figure 6.18 Increased variability in (a) the "baseflow regime" (as calculated by the IHA baseflow parameter) and (b) the rate of change of rising streamflows associated with commercial irrigation in Zone 33 of the Mkomazi Catchment

Monthly alteration in Zone 39

The seasonal distribution of streamflows resulting from the change in land use from reference hydrological conditions in Zone 39 is shown in Figure 6.19 for commercial forestry, rain-fed agriculture, alien invasive riparian vegetation, degraded grassland and thicket and bushland. These land uses have been allocated to the Deep-rooted Streamflow Reduction, Shallow-rooted Streamflow Reduction, Transmission, Recession and

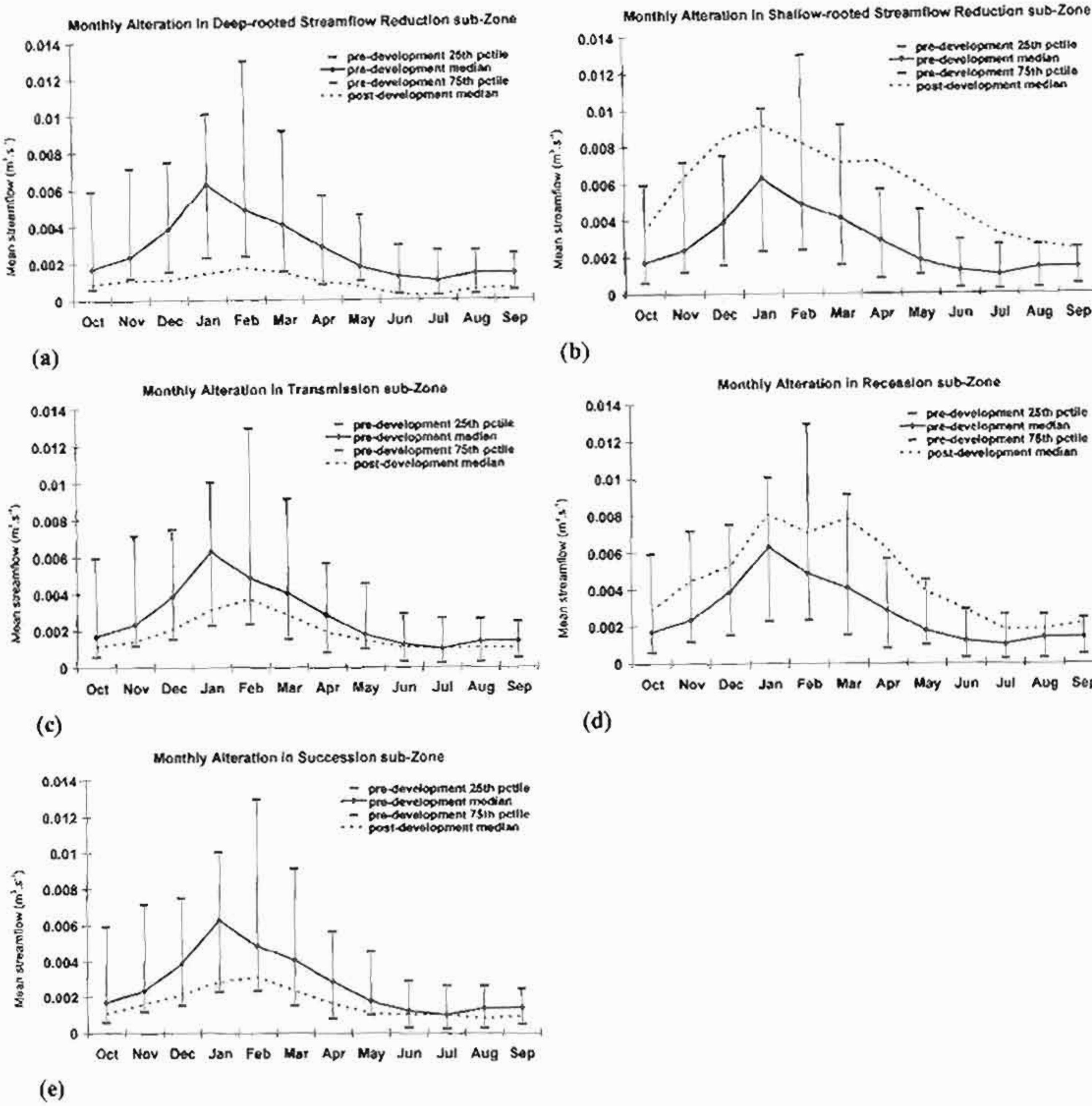

Figure 6.19 Monthly alteration of the seasonal distribution of streamflows of Zone 39 in the Low Precipitation Water Source Zone of the Mkomazi Catchment as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) alien invasive riparian vegetation, (d) degraded grassland and (e) thicket and bushland. All values based on alteration of transformation of a unit of 1 km^2 from reference hydrological conditions.

Succession sub-Zones respectively (*c.f.* Chapter 4, Section 5.3.4 and this Chapter, Section 5.3.1). While Table 6.15 shows that areal extents of these sub-zones differ, the graphs in Figure 6.19 depict the monthly alteration that would occur if only a unit of 1 km^2 was developed from reference hydrological conditions in order to draw realistic comparisons between the zones.

While Zone 39 was reported to have the least overall alteration among the high information indices identified for the LPWS zone type, the seasonal distribution of streamflows resulting from the various land uses operating in this zone indicate that in most years neither commercial forestry (Figure 6.19a) nor rain-fed agriculture (Figure 6.19b) can be practised while maintaining the average monthly streamflows within the RVA target window. As a result of commercial forestry, in an average year, average monthly streamflows during wet season months are lower than the lower RVA target of the 25th percentile of pre-development values. Conversely, rain-fed agriculture (in this instance, subsistence agriculture) results in average monthly streamflows being greater than the upper RVA target of the 75th percentile of pre-development values in the dry season months. Both rain-fed agriculture and degraded grassland (Figure 6.19d) result in increased streamflow generation in their respective sub-zones throughout the year, but for different reasons. First it should be emphasised that within a zone the *ACRU* model is not spatially explicit, *i.e.* it does not recognise the spatial distribution of any particular land use within a zone. In the model input, reference conditions in Zone 39 comprise area-weighting of Valley Bushveld and Ngongoni Veld as a land cover (*c.f.* Figure 6.5). Thus, reference conditions in this zone incorporate the hydrological attributes of both grassland and thicket and bushland. Consequently, in addition to the patches of bare soil which lead to increased soil water evaporation rates after ploughing at the start of the growing season (described above), clearing the natural grassland (and thicket and bushland) for crop production has itself resulted in increased streamflow generation (Figure 6.19b), with disruption of the natural hydrological processes of precipitation interception, plant water use through evapotranspiration and hydrological partitioning in the soil profile. Similarly, degradation of the natural grassland has increased streamflow generation (Figure 6.19d). However, in this instance there is year-round vegetative cover, albeit sparse grassland, with no tilling of the soil profile. As a consequence, the evapotranspiration losses from degraded grassland are relatively less than those from the practice of rain-fed agriculture.

Similarly to commercial forestry, alien invasive riparian vegetation in Zone 39 has reduced streamflows, although the degree of alteration is less severe, with the lower RVA target being exceeded throughout the year in most years (Figure 6.19c). Figure 6.5 indicates that reference conditions in this zone comprised a modest patch of "Valley Bushveld" (Acocks, 1988). Encroachment of this land cover in Zone 39 (Figure 6.6.) has considerably reduced the magnitude of average monthly streamflows (Figure 6.19e), although again not to the same extent as commercial forestry. As described for land cover conditions in Zone 33, there are strong similarities between the reductions invoked by alien invasive riparian vegetation and thicket and bushland. This emphasises the need for stewardship approaches to manage the opportunistic encroachment of terrestrial plant species which have neither an ecological nor economic value to stakeholders.

Hydrological alteration in Zone 39

The degree to which the RVA target range is not attained for selected hydrological parameters of the streamflow regime at Reference Zone 39 as a result of different societal activities is shown in Figure 6.20 for the sub-zones described immediately above. The graphs in Figure 6.20 represent hydrological alteration resulting from the actual areal extent allocated to each of the different activities as shown in Table 6.15.

The greatest alteration in the hydrological parameters in Zone 39 results from commercial afforestation (with substantial reductions in streamflows in the wet season months from December to February, in the duration of high flow pulses and in the rate of rising streamflows). Nonetheless, there are more years in post-development conditions where the RVA target window is met for average flows in October and for the "baseflow" component of the streamflow regime as calculated by the IHA method (also see Footnote 2, Page 6-91). However, the impact of commercial forestry in Zone 39 results in decreased variability of the baseflow regime, with this attainment being met at the expense of annual occurrences in high flow years (*c.f.* Figure 6.20a). The presence of deep-rooted trees in this sub-zone results in reduction of the magnitude of all high flow events ranging from the 3-day maximum to the seasonal high flow event.

Encroachment of thicket and bushland (*c.f.* Figure 6.20e) has similar impacts to commercial forestry on the streamflow regime in Zone 39. However, the alteration is lower for all the flow parameters. Average flows in January and February can be met in

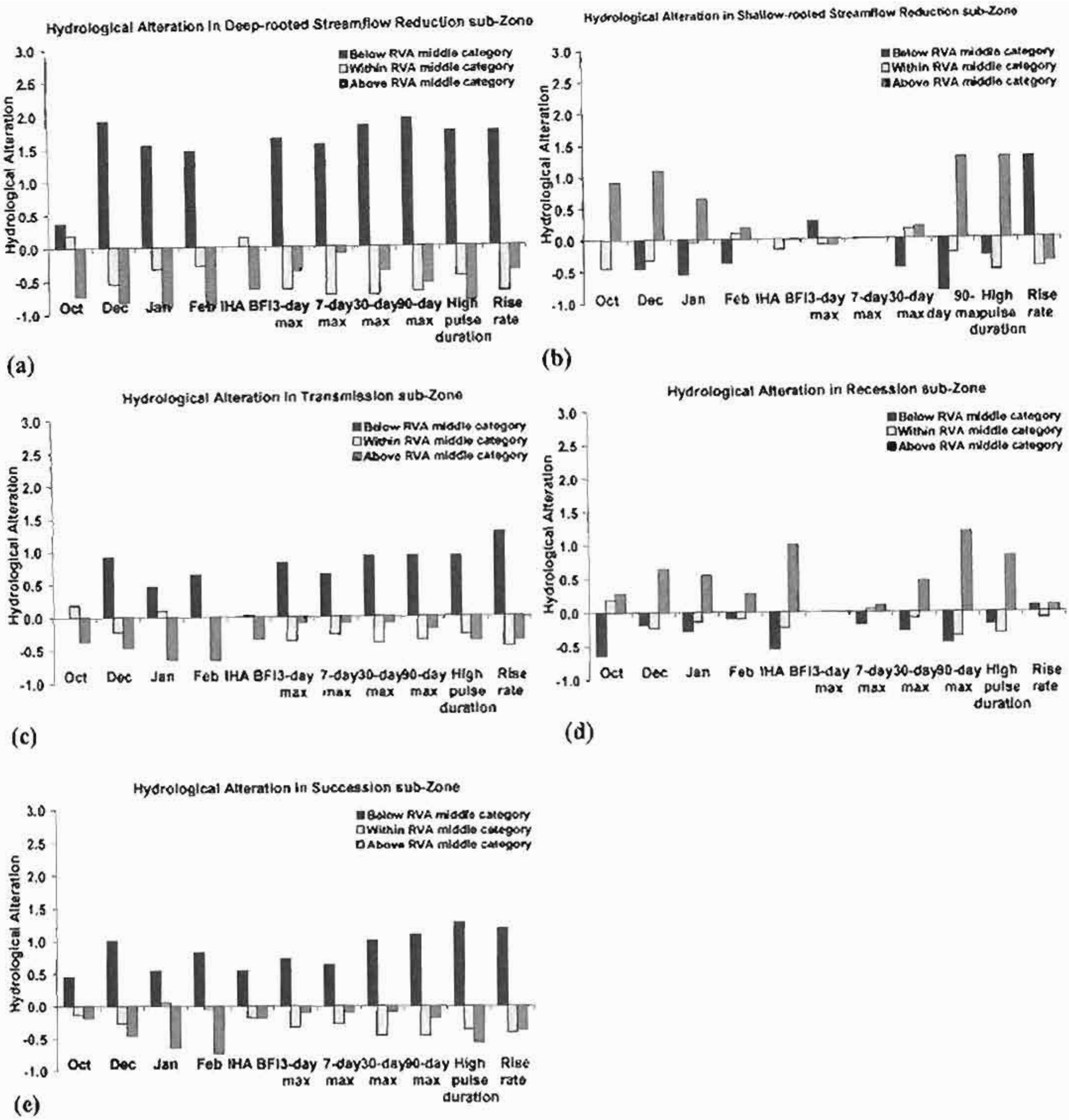

Figure 6.20 Hydrological alteration of selected hydrological parameters of the streamflow regime in Zone 39 as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) alien invasive riparian vegetation, (d) degraded grassland and (e) thicket and bush land. Hydrological parameters selected on their contribution to the subset of high information indices for the Low Precipitation Water Source Zone Type (of which Zone 39 has the least alteration) within the Mkomazi Catchment.

close to 50% of the post-development years, although this is at the expense of flows in high flow years (*c.f.* Figure 6.20e and Figure 6.21a). The greatest hydrological alterations as a result of this natural succession are manifest in the high flow events, with the RVA target window being attained less often than under reference conditions, particularly for the 30-day and 90-day maxima flow events, the duration of high flow pulses (*c.f.* Figure 6.21b), and rate of rising river levels. This is as a result of higher interception of precipitation and higher evapotranspiration than under reference conditions. Unlike commercial forestry, thicket and bushland reduces the "baseflow" regime to the extent that the RVA target window is achieved less often than under reference conditions. However, the overall variation in the "baseflow" regime is less severe under thicket and bushland.

The impacts of alien invasive riparian vegetation in Zone 39 are very similar to those of thicket and bushland. However, there is some difference between the two land uses regarding the resultant "baseflow" regime with baseflows under alien invasive riparian vegetation bearing a closer resemblance to reference hydrological conditions (Figures 6.20c and 6.20e) (also see Footnote 2, Page 6-91).

Rain-fed agriculture (comprising "semi-commercial / subsistence" crops according to the CSIR land use classification of 1996) in Zone 39 results in considerable increases in the natural range of streamflows as the growing season progresses (*c.f.* December in Figure 6.20b). However, there are fewer years where the RVA target window is attained, as a result of reduced variability of monthly streamflows at this time. December is ecologically important for aquatic ecosystems in the Mkomazi Catchment, since this month provides the first-of-season floods which are critical to various biotic life stages. Increases in streamflow over the natural range at this time of the year could have serious implications for the completion of ecological processes. As the growing season progresses, the RVA target window is attained more often, with more than 50% of post-development years meeting the target range in February. However, this is generally at the expense of streamflows in the low flow years, implying reduced variability of the streamflow regime at the height of the growing season (January / February in Figure 6.20b). The "baseflow" regime experiences little change as a result of rain-fed agriculture (also see Footnote 2, Page 6-91). Crops with shorter roots do not generally have a substantial impact on this streamflow component. Similarly, shallow-rooted crops have little impact on the shorter high flow events. However, over the duration of the growing season, and coinciding with

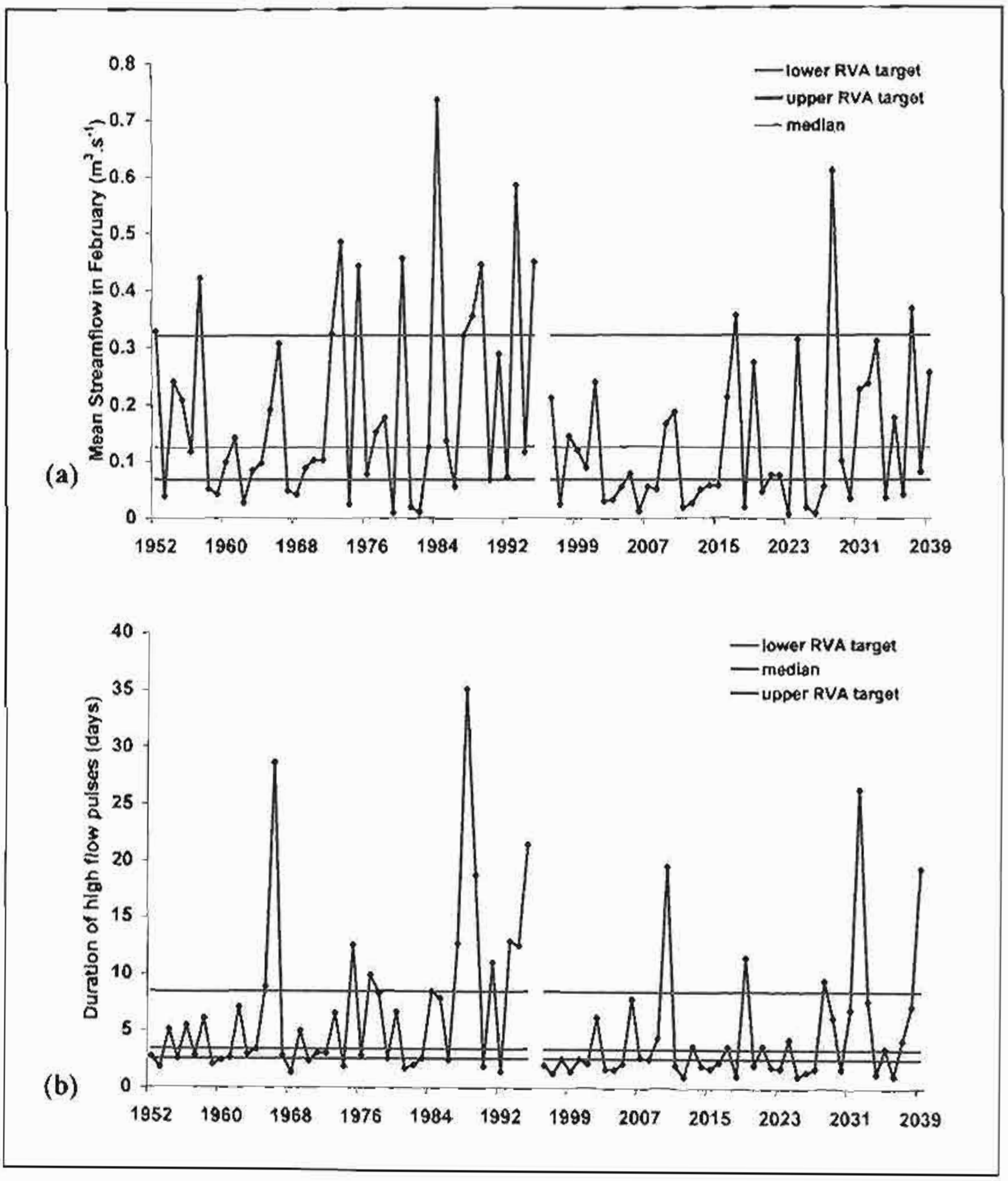

Figure 6.21 Reduced variability in (a) mean streamflows in February and (b) mean duration of high flow pulses associated with thicket and bushland in Zone 39 of the Mkomazi Catchment

the 90-day maximum flow, they are unable to intercept high flow events to the same degree as the natural grassland they replace, with the result that there is a substantial increase in the high pulse duration in post-development years. Nonetheless, rain-fed agriculture results in similar reductions to the rise rate of streamflows as both commercial forestry and thicket and bushland.

Degraded grassland (*c.f.* Figure 6.20d) results in similar impacts on the flow parameters to those described for rain-fed agriculture. However, at the start of the wet season there are smaller relative increases in streamflow generation as a result of the greater biomass of, albeit degraded, grassland, compared to the patches of bare soil which occur under rain-fed agriculture at this time. Moreover, the RVA target window is attained in more than 50% of post-development years in October (Figure 6.20d). However, this is at the expense of streamflows in the lower RVA category, thereby reducing variation in low flow years. The impacts of degraded grassland in Zone 39 also results in elevation of the "baseflow" regime as a result of lowered interception and evapotranspiration compared to reference hydrological conditions. The alteration to high flow events is similar to those experienced under rain-fed agriculture. However, there is little change to the rising streamflow rates under degraded grassland as this land cover incurs less disruption to the soil profile and hydrological partitioning than any of the other societally induced land cover changes.

5.3.2.2 Hydrological alteration in the flow parameters contributing to the high information indices of the MPWS zone type

The subset of high information indices of the variability among the streamflow regimes in the Moderate Precipitation Water Source zone type was shown in Table 6.9 to comprise M_A6, M_A7, M_A9 to M_A11 (average flows in March, April, June, July, and August), M_A19, M_A20 and M_A24 (variability in April, May and September), M_L4 (the RESDSS simple low flow index), D_L12 (variability in the 90-day minimum flow), T_H1 (date of the annual maximum flow) and R_A4 (variability in the rise rate of streamflows). Zone 26 was identified as having the greatest existing alteration (at the zone scale, and for general and median conditions) from reference conditions among the zones in the MPWS zone type. Zone 16 was identified as having the least existing alteration among the zones in this Group.

Monthly alteration in Zone 26

The seasonal distribution of streamflows resulting from the change in land use from reference hydrological conditions in Zone 26 are shown in Figure 6.22 for commercial forestry, rain-fed agriculture, commercial irrigation, alien invasive riparian vegetation and thicket and bushland. These land uses have been allocated to the Deep-rooted Streamflow Reduction, Shallow-rooted Streamflow Reduction, Supplementary, Transmission and Succession sub-Zones respectively (*c.f.* Chapter 4, Section 5.3.4 and this Chapter, Section 5.3.1). As in the previous Section, and in order to make reasonable comparisons among the different land uses, the graphs in Figure 6.22 depict the monthly alteration that would occur if only a unit of 1 km^2 was developed from reference conditions.

The monthly alteration among the sub-zones in Zone 26 varies considerably, with the greatest alteration occurring as a result of commercial irrigation and the least alteration occurring under rain-fed agriculture. The practice of rain-fed agriculture in Zone 26, results in increased generation of streamflows at the start of the growing season. However, the degree of alteration is not as great as that which occurs in Zones 33 and 39 (*c.f.* Figures 6.16b and 6.19b), and average monthly streamflows for all months remain well within the RVA target range in most years. Commercial forestry cannot be practised in Zone 26 without the average flows in the wet season (October through March) failing to meet the RVA target window in at least 50% of post-development years (*c.f.* Figure 6.22a). Alien invasive riparian vegetation and encroaching thicket and bushland both have less impact than commercial forestry, although there are streamflow reductions in both cases. Commercial irrigation reduces streamflows to such an extent that without upstream contributions, the streamflow regime all but diminishes (*c.f.* Figure 6.22c).

Hydrological alteration in Zone 26

The degree to which the RVA target range is not attained for selected hydrological parameters of the streamflow regime in Zone 26 as a result of different societal activities is shown in Figure 6.23 for the sub-zones described immediately above. The graphs in Figure 6.23 represent hydrological alteration resulting from the "actual area" areal extent allocated to each of the different activities as shown in Table 6.15.

The greatest alteration in the hydrological parameters in Zone 26 results from commercial irrigation, with substantial reductions in the natural range of the magnitudes of streamflows

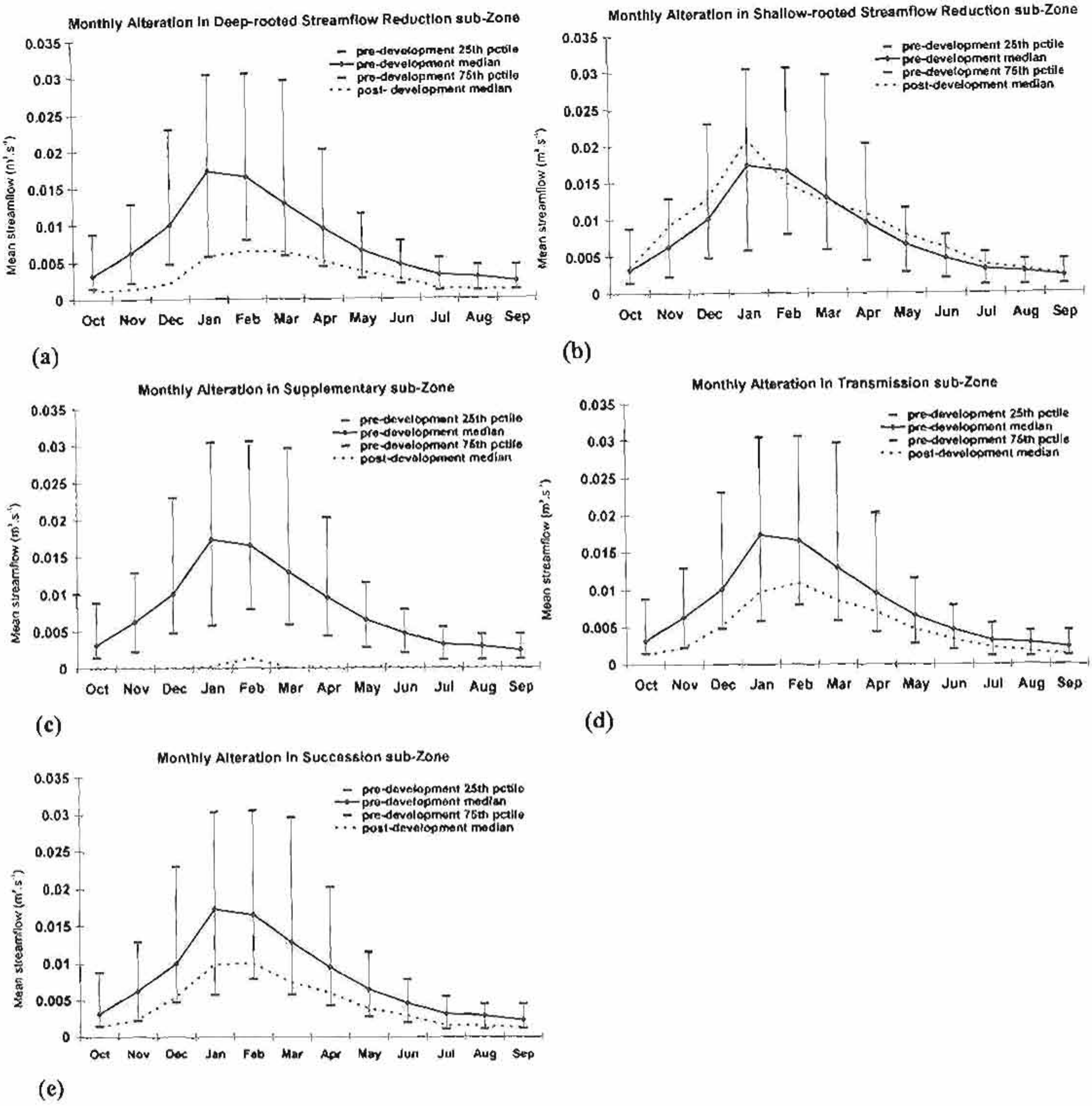

Figure 6.22 Monthly alteration of the seasonal distribution of streamflows of Zone 26 in the Medium Precipitation Water Source Zone of the Mkomazi Catchment as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) commercial irrigation, (d) alien invasive riparian vegetation and (e) thicket and bush land. All values are based on alteration of transformation of a unit of 1 km^2 from reference hydrological conditions.

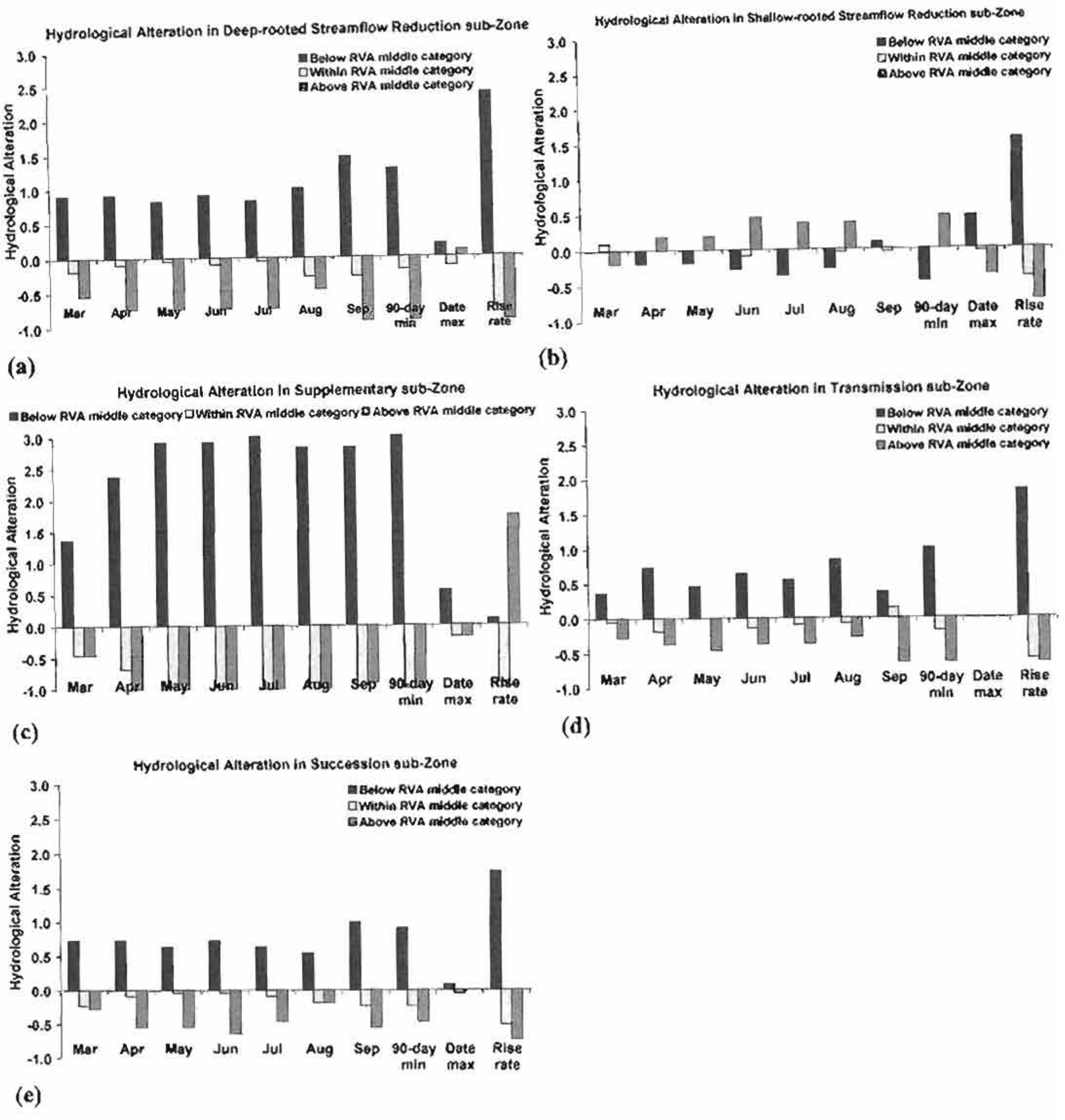

Figure 6.23 Hydrological alteration of selected hydrological parameters of the streamflow regime in Zone 26 as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) commercial irrigation, (d) alien invasive riparian vegetation, (e) thicket and bush land. Hydrological parameters are selected on their contribution to the subset of high information indices for the Medium Precipitation Water Source Zone Type (of which Zone 26 has the greatest existing alteration) of the Mkomazi Catchment.

in the dry season (*c.f.* Figure 6.23c). Most of the high information indices of the streamflow regimes in the MPWS zone type are derived from low season flow parameters. These parameters experience high alteration as a result of irrigation practices which use any available storage water to compensate for the absence of rainfall in the dry season. Moderate precipitation is available for storage in this MPWS zone in most years and, therefore, the alteration to average flows in March is less severe than in the ensuing months. The degree of hydrological alteration appears to lessen towards the end of the dry season. However, this is most likely as a result of the absence of any available water in storage rather than an alleviation of the irrigation practice. There is little impact on the date of the maximum flow. This flow parameter is relatively insensitive to even severe alteration to the streamflow regime, since it is linked with individual rainfall events rather than land use conditions. The only flow parameter that experiences any increase in its range of variation under irrigation is the rise rate of streamflows, with most of the annual values now occurring above the 75th percentile of pre-development conditions. This is a consequence of the supplementary input of water to this sub-zone in the growing season which augments the streamflow response to natural precipitation events and thereby increases the variability associated with this parameter.

The similarity in the extent of streamflow reductions for commercial forestry (*c.f.* Figure 6.23a), alien invasive riparian vegetation (*c.f.* Figure 6.23d) and encroaching thicket and bushland (*c.f.* Figure 6.23e) results in similarities between the impacts of these land uses on the selected streamflow parameters. In each instance the RVA target window is met in close to 50% of post-impact years in the dry season months of April through to July, at the expense of flows in wetter years. In these dry season months, pine trees (as planted in the Deep-rooted Streamflow Reduction sub-Zone) utilise more water resources than the tree and shrub species found in the Transmission and Succession sub-Zones. There is little difference between these three societal influences regarding the date of the maximum flow. This confirms the insensitivity of this flow parameter to different land covers.

Rain-fed agriculture produces the least hydrological alteration in Zone 26 and the RVA target window is attained in 50% of post-development years for most of the selected flow parameters (*c.f.* Figure 6.23b). In concurrence with Figure 6.22b, there is some elevation of the flow regime, particularly towards the end of the dry season, after the crops (a mix of maize and subsistence agriculture, according to the *ACRU* land use classification) have been harvested. However, the alteration to the rise rate of river levels resembles that shown for thicket and grassland (*c.f.* Figure 6.23d) and to a lesser extent for commercial forestry (*c.f.* Figure 6.23a) and alien invasive riparian vegetation (*c.f.* Figure 6.23d). This reduction in the natural range of this flow parameter is as a result of increased interception of summer rainfall and increased evapotranspiration when the plants are growing most vigorously.

Monthly alteration in Zone 16

The seasonal distribution of streamflows resulting from the change in land use from reference hydrological conditions in Zone 16 is shown in Figure 6.24 for commercial forestry, rain-fed agriculture, alien invasive riparian vegetation, degraded grassland and thicket and bushland. These land uses have been allocated to the Deep-rooted Streamflow Reduction, Shallow-rooted Streamflow Reduction, Transmission, Recession and Succession sub-Zones respectively (*c.f.* Chapter 4, Section 5.3.4 and this Chapter, Section 5.3.1). As in the previous Sections, and in order to make reasonable comparisons among the different land uses, the graphs in Figure 6.24 depict the monthly alteration that would occur if only a unit of 1 km^2 was developed from reference hydrological conditions.

Among the land uses tested, commercial forestry (*c.f.* Figure 6.24a) has the greatest impact on the streamflow regime in Zone 16, with average monthly streamflows reduced to below the 25th percentile of pre-development streamflows, throughout the year in most years. Both rain-fed agriculture (*c.f.* Figure 6.24b) and degraded grassland (Figure 6.24d) result in increased streamflows, with average monthly streamflows being generally higher than the pre-development median values. However, rain-fed agriculture results in considerably higher flows at the start and at the end of the dry season, whereas at the height of the growing season, post-development years experience a reduction in streamflows. Again, there is little difference between alien invasive riparian vegetation and encroaching thicket and bushland (*c.f.* Figures 6.24c and 6.24e). Both of these land covers have a similar impact on the streamflow regime of Zone 16 to that of commercial forestry. However, under alien invasive riparian vegetation and thicket and bushland, average monthly streamflows in most years are higher than the 25th percentile value of pre-development monthly streamflows virtually throughout the year.

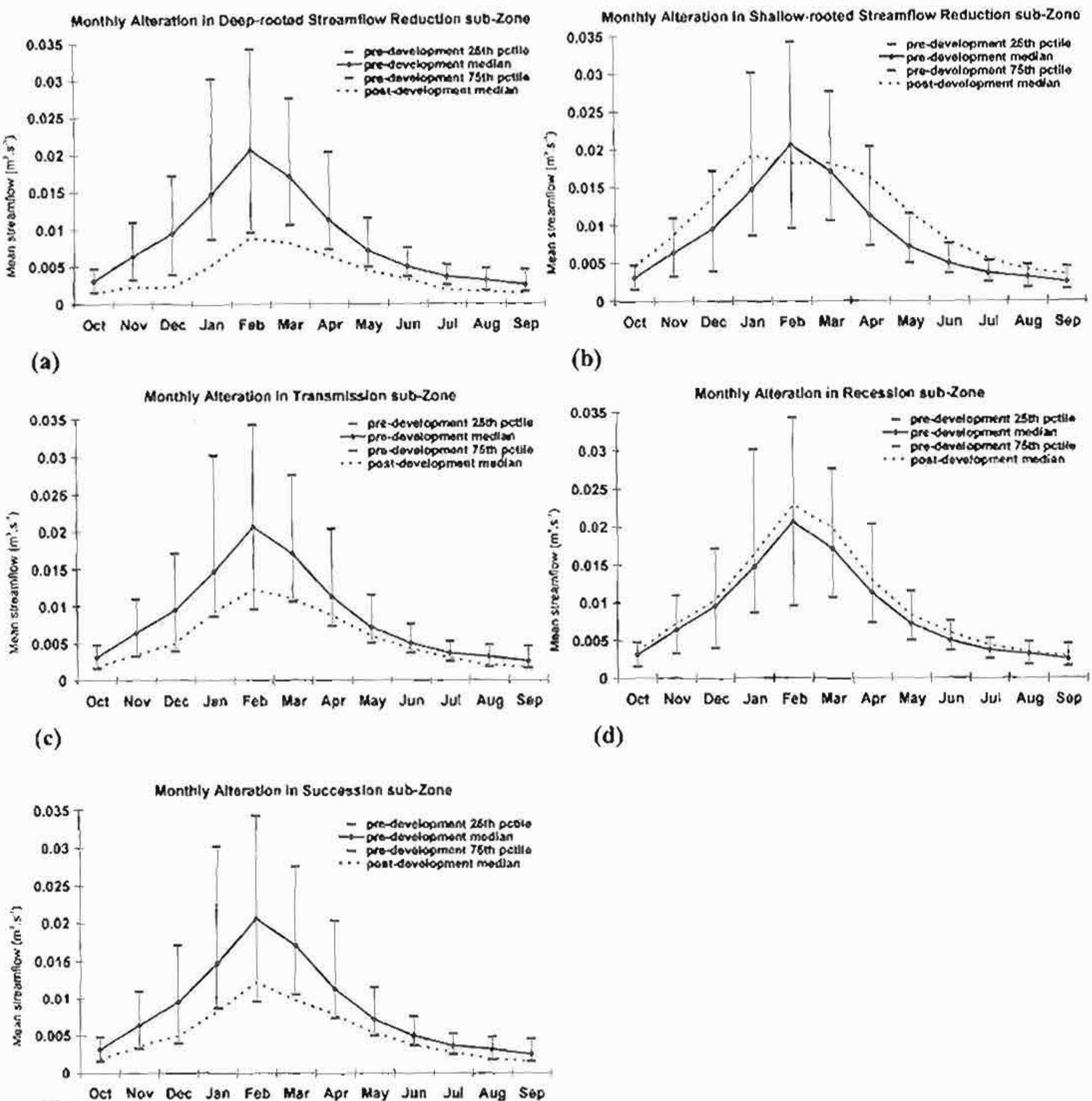

Figure 6.24 Monthly alteration of the seasonal distribution of streamflows of Zone 16 in the Medium Precipitation Water Source Zone of the Mkomazi Catchment as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) alien invasive riparian vegetation, (d) degraded grassland and (e) thicket and bush land. All values are based on alteration of transformation of a unit 1 km^2 from reference hydrological

Hydrological alteration in the flow parameters

The greatest hydrological alteration of the selected flow parameters in Zone 16 results from commercial forestry. Average monthly flows at the height of the dry season in this MPWS zone experience considerable reductions in low flow years (Figure 6.25a). As a consequence of these deep-rooted trees, the rise rate of river levels is substantially altered with most years experiencing less than the 25th percentile of pre-development flow rates. However, in common with the Zone 26 in the MPWS zone type, there is little impact on the date of the annual maximum flow event as a result of commercial forestry.

As expected, the practice of rain-fed agriculture (*i.e.* subsistence agriculture in this instance) has moderate to considerable alteration in the dry season months, particularly in June and July, in high flow years (*c.f.* Figure 6.25b). This occurs subsequent to harvesting, when any appreciable rainfall events generate increased streamflow generation as a result of reduced land cover. This feature is also reflected by the streamflows associated with the seasonal (90-day) minimum flow event. However, the evapotranspiration processes associated with the additional biomass in the growing season result in some postponement of the date of the annual maximum flow event. In common with commercial forestry, alien invasive riparian vegetation and thicket and bushland in this zone, the rise rate of streamflows experiences reductions in the natural range. This disruption to the rate of change of streamflows results from reductions in streamflows for increased, albeit highly seasonal, biomass production (*e.g.* average monthly flows are reduced in February, *c.f.* Figure 6.24b) and, consequently, a dampening of the hydrograph (*c.f.* Figure 6.24b).

Degraded grassland in Zone 16 results in only small alterations to the selected flow parameters. At the start and at the end of the dry season the RVA target window is attained in 50% of post-development years (*c.f.* Figure 6.25d). Nonetheless, the general trend in alteration across the different parameters indicates slight increases in all years. The greatest alteration among the flow parameters is the increased range of the 90-day minimum flow. However, even this is relatively inconsequential (*c.f.* Figure 6.26), particularly when compared to the alterations of this parameter invoked by the other land uses in this zone. The greatest divergence from the other land uses in Zone 16 relates to the change in the rise rate of streamflows, whereby under degraded grassland conditions this flow parameter experiences little change from reference conditions. In common with

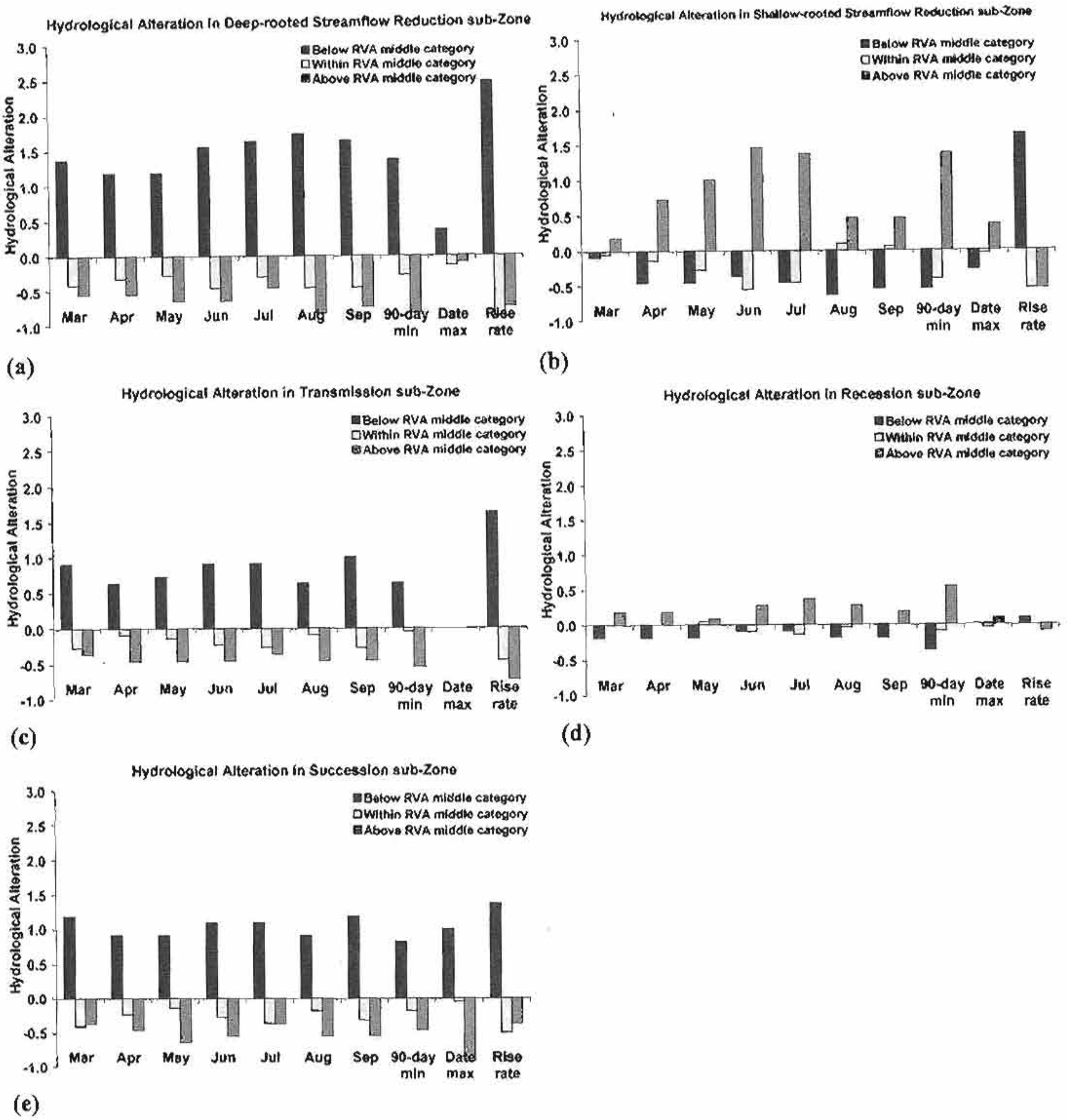

Figure 6.25 Hydrological alteration of selected hydrological parameters of the streamflow regime at Zone 16 as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) alien invasive riparian vegetation, (d) degraded grassland and (e) thicket and bush land. Hydrological parameters are selected on their contribution to the subset of high information indices for the Medium Precipitation Water Source Zone Type (of which Zone 16 has the least existing alteration) of the Mkomazi Catchment.

the other land uses, date of the annual maximum event is relatively unimpacted, since this event is a function of individual rainfall events.

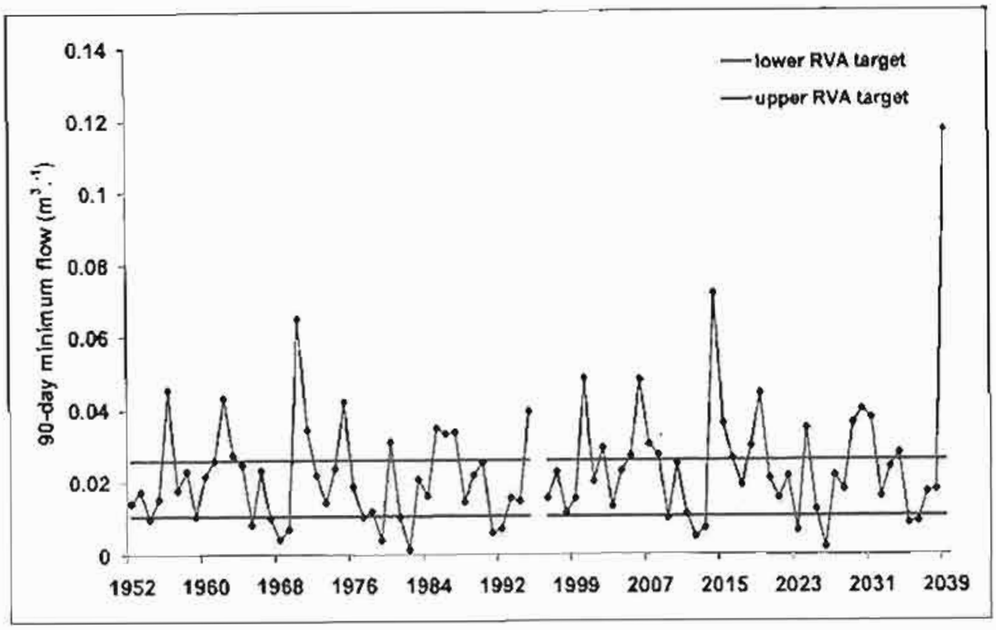

Figure 6.26 Increased range in the 90-day minimum flow associated with degraded grassland in Zone 16 of the Mkomazi Catchment

The pattern of alteration in the selected flow parameters under both alien invasive riparian vegetation and encroaching thicket and bushland is very similar, although slightly less to that experienced as a result of commercial forestry. There are only two noteworthy exceptions to this feature. First, in both instances, the impacts in dry season months are relatively less severe than under commercial forestry. Second, while the degree of attainment of date of the annual maximum flow for both land uses is similar (*i.e.* attained in nearly 50% of post-impact years, Figure 6.25c and 6.25e), the natural range of this parameter is much reduced by thicket and bushland, and as a consequence, has become more predictable.

5.3.2.3 Hydrological alteration in the flow parameters contributing to the high information indices of the HPWS zone type

The subset of high information indices of the variability among the streamflow regimes in the HPWS zone type was shown in Table 6.9 to comprise M_A4 to M_A6, M_A8, and M_A9 (average flows in January, February, March, May and June), M_A24 (variability in

September flows), M_A25 (CDB, the Desktop Reserve Model overall variability), D_H4 (the 30-day maximum flow); D_L9 and D_L11 (variability of both the 3-day and the 30-day minimum flow), T_L1 (date of the minimum flow) and R_A4 (rise rate of streamflows). Zone 10 was identified as having the greatest existing alteration (at the zone scale, and for general and median conditions) from reference hydrological conditions among the zones in the HPWS zone type. Zone 5 was identified as having the least existing alteration among the zones in this Group.

Monthly alteration in Zone 10

The seasonal distribution of streamflows resulting from the change in land use from reference hydrological conditions in Zone 10 are shown in Figure 6.27 for commercial forestry, rain-fed agriculture, commercial irrigation, alien riparian vegetation, degraded grassland and thicket and bushland. These land uses have been allocated to the Deep-rooted Streamflow Reduction, Shallow-rooted Streamflow Reduction, Supplementary, Transmission, Recession and Succession sub-Zones respectively (*c.f.* Chapter 4, Section 5.3.4 and this Chapter, Section 5.3.1). As in the previous Sections, and in order to make reasonable comparisons among the different land uses, the graphs in Figure 6.27 depict the monthly alteration that would occur if only a unit of 1 km^2 was developed from reference hydrological conditions.

As expected, the greatest alteration to the seasonal distribution of streamflows among the land uses tested results from commercial irrigation (*c.f.* Figure 6.27c). Even in this high precipitation water source zone, the practice of irrigation severely reduces the flow regime for most of the year. In most years, average monthly streamflows are lower than the 25th percentile of pre-development values throughout the year, with the exception of February at the height of the wet season. However, as described in Section 5.3.1 of this Chapter, there would most certainly be a reduced crop yield when compared to normal irrigation practice which incorporates utilising upstream contributions to the water storage in this zone (*c.f.* Figure 6.10). Streamflow reductions under commercial forestry in Zone 10 are such that the median of average monthly flows for most months of the year is lower than the 25th percentile of pre-development values. With the exception of the winter months of May, June and July, average monthly streamflows fall within the RVA target window throughout the year, in most years under rain-fed agriculture. The least alteration in the seasonal distribution of streamflows is attributable to degraded grassland. However, as

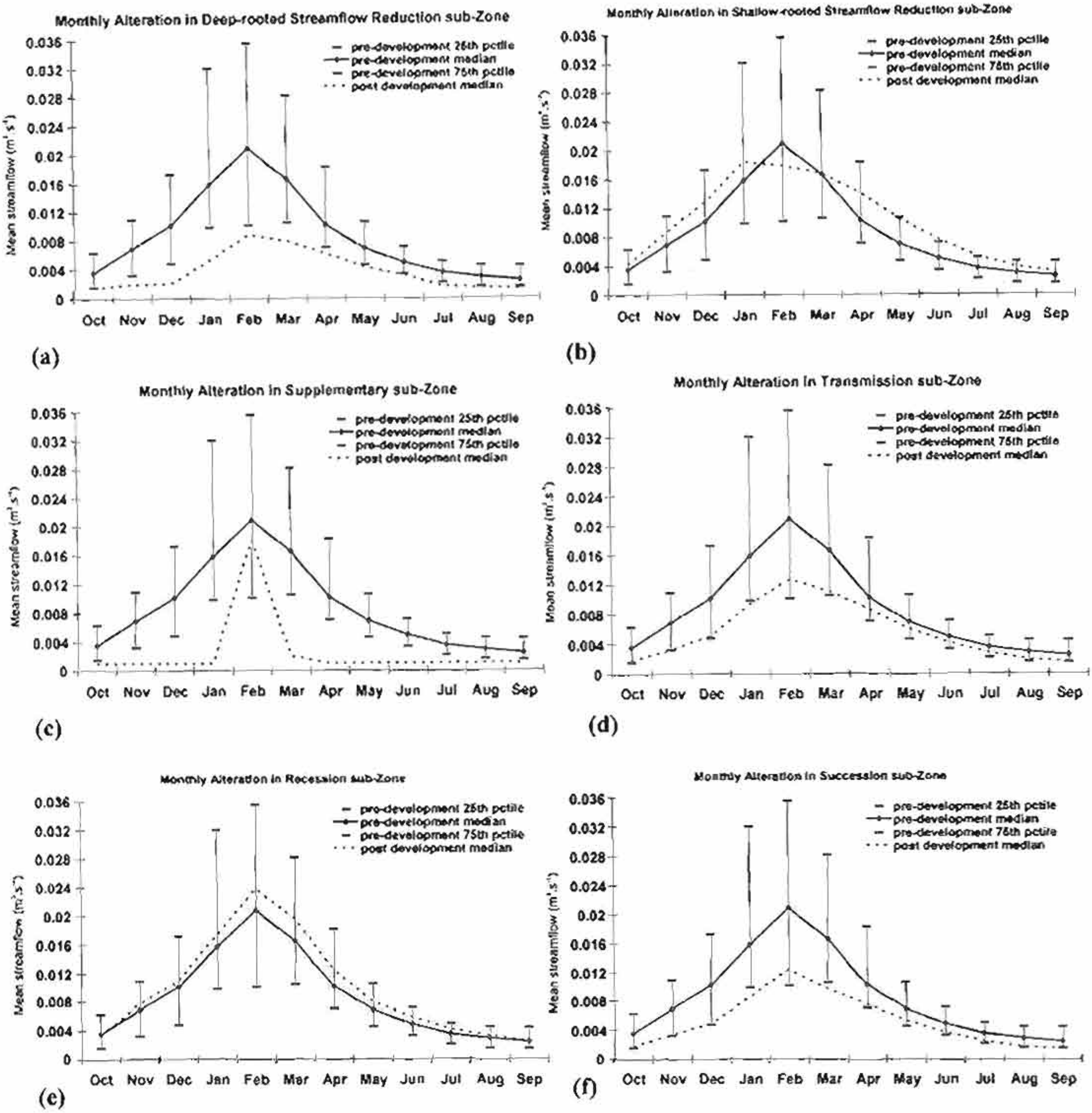

Figure 6.27 Monthly alteration of the seasonal distribution of streamflows of Zone 10 in the High Precipitation Water Source Zone of the Mkomazi Catchment as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) commercial irrigation, (d) alien invasive riparian vegetation, (e) degraded grassland and (f) thicket and bush land. All values are based on alteration of transformation of a unit of 1 km^2 from reference hydrological conditions.

described above, increases in the range of annual values of average monthly streamflows are just as detrimental to natural eco-hydrological functioning as reductions.

Hydrological alteration in Zone 10

The greatest hydrological alteration in Zone 10 for the selected flow parameters arises from commercial irrigation (Figure 6.28c). This "hypothetical" practice conducted in the absence of upstream contributions to water storage results in the decimation of streamflows throughout the dry season months, leading to the degeneration of the baseflow regime (as calculated by the Alt-BFI parameter based on the Desktop Reserve Model baseflow index of short-term variability, *c.f.* Chapter 5, Section 2.3.3.). Such a serious alteration in the baseflow regime is likely to disconnect hydrological connectivity between terrestrial subsurface flows and the stream channel. As a consequence, aquatic ecosystems would become disconnected from the landscape and become vulnerable to a disturbance regime not previously experienced. Moreover, the timing of the minimum flow event occurs much earlier in the year. This alone can have serious ecological consequences, as time shifts of even a few weeks can mismatch the synchronisation of aquatic life cycles with habitat requirements. As expected the rate of rising river levels is altered so that this flow parameter experiences considerably more variability than under pre-development conditions. This translates into the river experiencing a much more fluctuating and flashy flow regime than it previously did.

Commercial forestry also reduces streamflows in Zone 10 (*c.f.* Figure 6.28a). With this practice, reductions are evident throughout the year as a result of increased biomass and roots utilising water from greater depths within the soil profile. Nonetheless, the impact on baseflow, as calculated by the Desktop Reserve Model "baseflow" index, is negligible. However, under commercial forestry the timing of the minimum flow is sufficiently postponed to merit concern over the impacts of the extended low flow season for aquatic ecosystems.

In this high rainfall zone, hydrological alteration to average monthly flows as a result of rain-fed agriculture is, generally, negligible when assessing the attainment of the RVA target window (*c.f.* Figure 6.28b). The greatest alteration among the monthly flows tested is the moderate reduction in the variability of flows in the winter months of May and June. While there is some reduction in the natural range of the 30-day minimum flow, there is

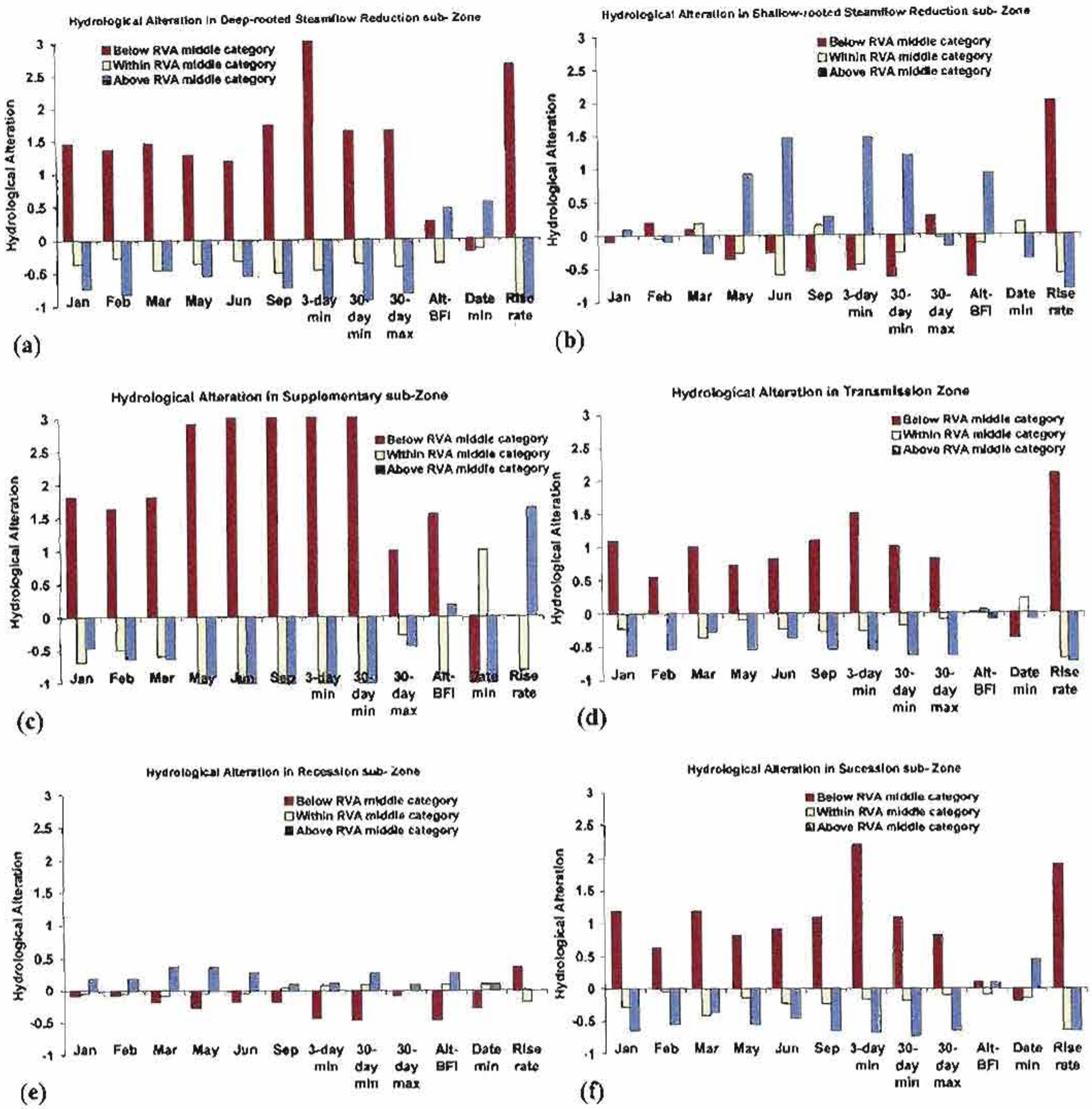

Figure 6.28 Hydrological alteration of selected hydrological parameters of the streamflow regime at Zone 10 as a result of (a) commercial forestry, (b) rain-fed agriculture, (c) commercial irrigation (d) alien invasive riparian vegetation, (e) degraded grassland and (f) thicket and bush land. Hydrological parameters selected on their contribution to the subset of high information indices for the High Precipitation Water Source Zone Type (of which Zone 10 has the greatest alteration) of the Mkomazi Catchment.

little impact on the baseflow regime (as calculated by the Alt-BFI parameter, *c.f.* Figure 6.29a). Nonetheless, there is advancement the timing of the minimum flow, particularly in high flow years (*c.f.* Figure 6.29b).

Degraded grassland in Zone 10 results in the RVA target window for the selected parameters being met in most years. While there are likely to be other non-streamflow related issues relating to degraded grassland cover (see discussion in Chapter 4, Section 5.3.4), this land cover has little impact on any of the selected flow parameters (*c.f.* Figure 6.28e). In particular, there is barely any change to the timing of the minimum flow event or to the rate of rising river levels.

The similarity between the hydrological alteration as a result of alien invasive riparian vegetation and that of encroaching thicket and bushland that was reported for zones in the MPWS and LPWS zone type is also evident for this high rainfall zone (*c.f.* Figures 6.28d and 6.28f).

Monthly alteration in Zone 5

Zone 5 was shown in Section 4 (*c.f.* Tables 6.12 and 6.13) to be the least altered zone in the HPWS group, as a result of its close-to-natural hydrological status. As a consequence, the only land use tested for this zone was alien invasive riparian vegetation. The alteration of the seasonal distribution of streamflows, as a result of this invasion of the Transmission Zone within Zone 5, is shown in Figure 6.30. As in the previous Sections, the graph in Figure 6.30 depicts the monthly alteration that would occur if a unit of only 1 km^2 was developed from reference hydrological conditions.

Figure 6.30 indicates that in this high precipitation water source zone, average monthly streamflows under alien invasive riparian vegetation are reduced to close to the 25th percentile of pre-development values, throughout the year, in most years. The greatest reductions in monthly streamflows are experienced at the start of the high flow season when these trees renew their evapotranspiration processes.

Hydrological alteration in Zone 5

In concurrence with Figure 6.30, Figure 6.31 indicates that reductions in the magnitude of streamflows in Zone 5 as a consequence of alien invasive riparian vegetation are more

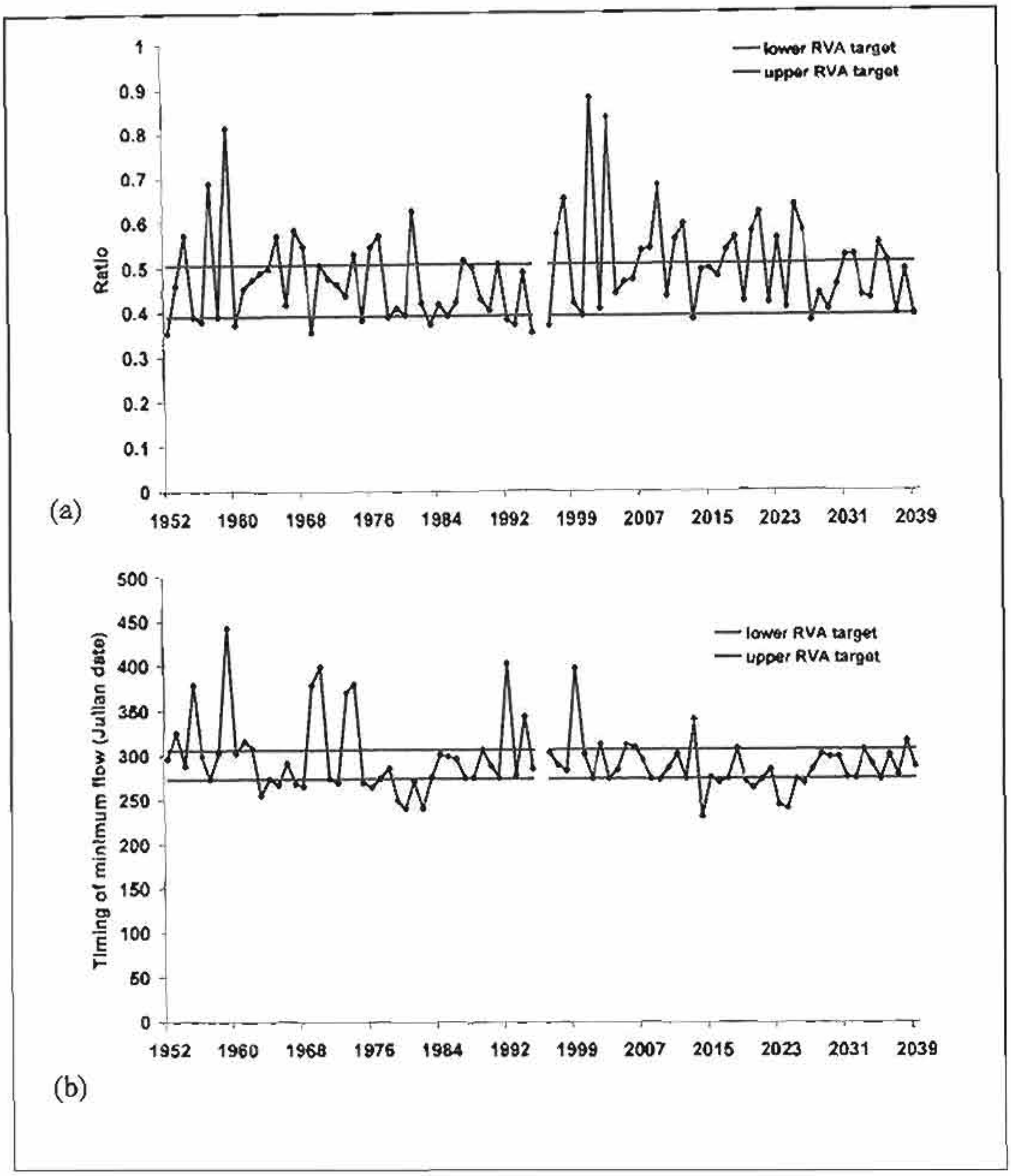

Figure 6.29 Increased variability in the baseflow regime as calculated by the Alt BFI flow parameter (a) and reduced variability in the timing of the minimum flow event (b) associated with rain-fed agriculture in Zone 10 of the Mkomazi Catchment. Note the Julian date in which those (8) years fell within the following year were reassigned a higher value (the addition of 366 days) so that they fell within the same season as the year in question.

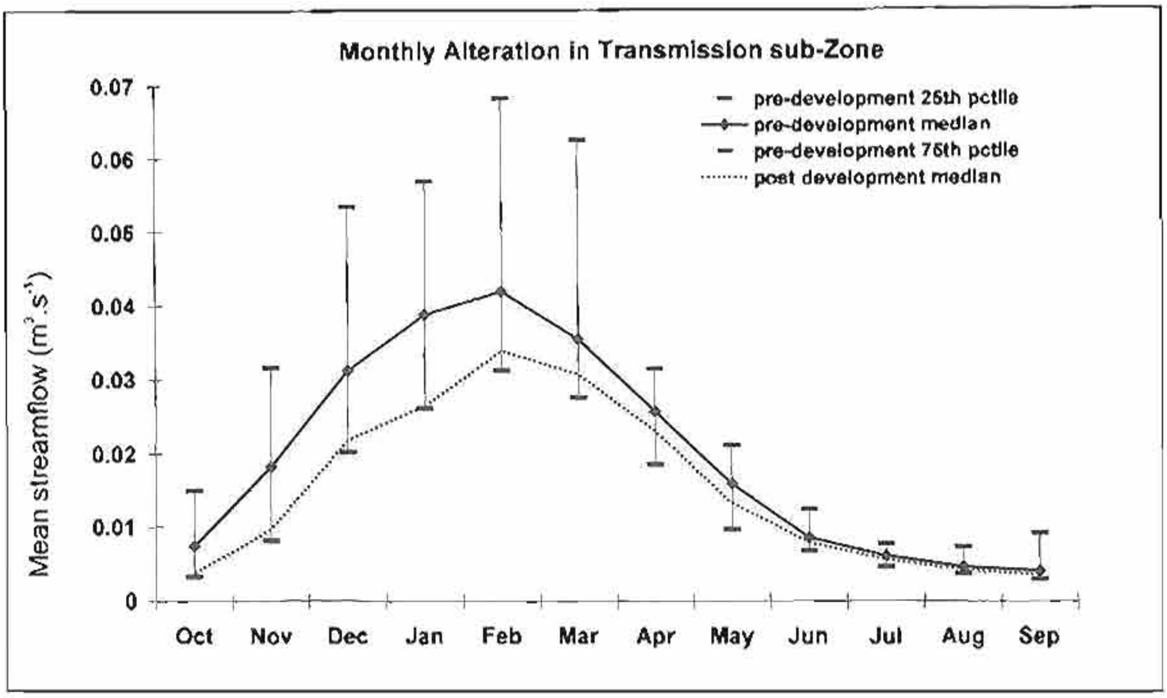

Figure 6.30 Monthly alteration of the seasonal distribution of streamflows of Zone 5 in the High Precipitation Water Source Zone of the Mkomazi Catchment as a result of alien invasive riparian vegetation. All values based on alteration of transformation of a unit of 1km^2 from reference hydrological conditions.

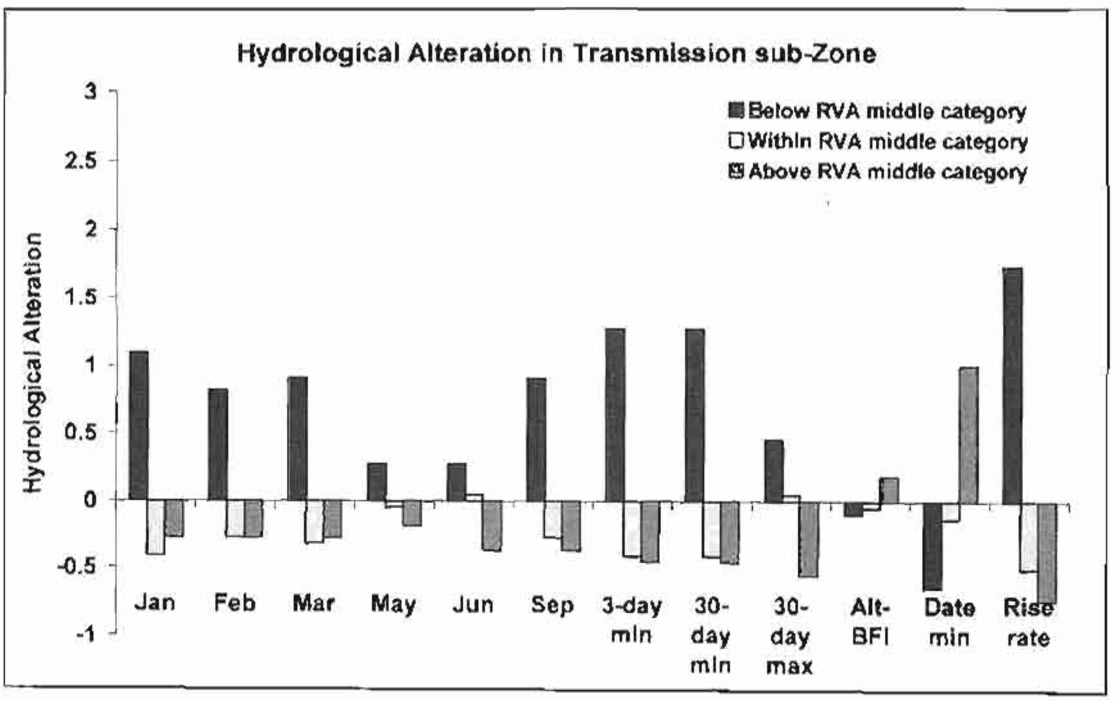

Figure 6.31 Hydrological alteration of selected hydrological parameters of the streamflow regime at Zone 5 as a result of alien invasive riparian vegetation. Hydrological parameters are selected on their contribution to the subset of high information indices for the High Precipitation Water Source Zone Type (of which zone 5 has the least existing alteration) of the Mkomazi Catchment.

substantial in wet season months than dry season months. There is greater likelihood of the RVA window being met in May and June than any other month. Nonetheless, there are still reductions in the natural range of flows in the low flow season (*e.g.* in September, *c.f.* Figure 6.31) accompanied by reduced range associated with the 30-day minimum flow event (Figure 6.32a). Alien invasive riparian vegetation has the potential to postpone the timing of the minimum flow event so that it occurs later in the year (*c.f.* Figure 6.32b). Ecologically, this could lead to aquatic life cycles being desynchronised with environmental conditions. The greatest alteration as a result of these trees is associated with the rate of rise of streamflows. This finding concurs with the discussion around this streamflow parameter for large plant species in previous Sections.

5.4 Trade-off Potential for the Hydronomic Zones

The Study in Section 5.3 highlights the degree of hydrological alteration for different societal activities and impacts in the tributaries of the Mkomazi Catchment. Allocating the main societal activities and impacts to sub-zones facilitated the assessment of each activity on the streamflow regime of the zone within which it was located. The degree to which any particular activity alters the streamflow regime through disruption of the hydrological cycle can be expected to be valuable to stakeholders in any decision-making forum associated with maximising the generation of ecosystem goods and services within the Mkomazi Catchment. However, as stressed throughout this thesis, hydrological connectivity operates at different organisational resolutions, or scales, in catchments. The concept of the upstream-downstream relationships of the energy and nutrient pathways of natural ecosystems has been referred to in several instances throughout this thesis (Chapters 1, 2, and 4 as well as this Chapter). The complexities arising from the synergy of societal and climatic factors were highlighted in Section 4 of this Chapter. Thus, in this Section, the upstream-downstream hydrological relationships among different water users in the tributaries of the Mkomazi Catchment are examined rather by focusing on the trade-off potential of the different societal activities within and among the zones on the tributaries of the Mkomazi Catchment.

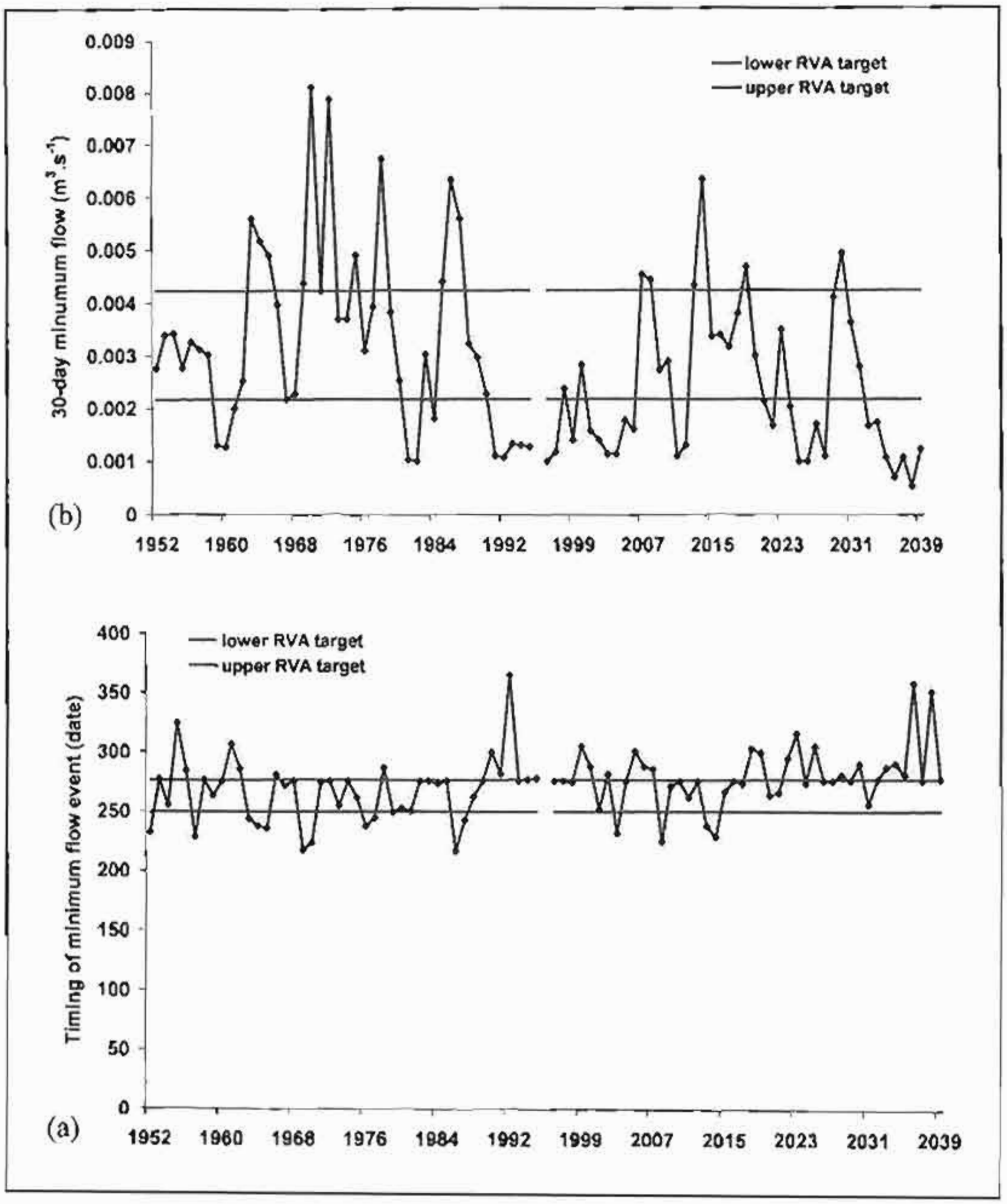

Figure 6.32 (a) Reduced range of the 30-day minimum flow event and (b) reduced variability in the timing of the minimum flow event associated with alien riparian invasive vegetation in Zone 5 of the Mkomazi Catchment

5.4.1 Methods of analysis

The results of the study of hydrological alteration in the different sub-zones described in three immediately preceding sub-sections were examined in conjunction with the ranking of degree of hydrological alteration, in terms of Equation 2 of Section 5.3.1, attributable to each of the hydronomic zones on the tributaries of the Mkomazi Catchment (*c.f.* Figure 6.13). First, comparison of the hydrological alteration of the selected streamflow parameters, resulting from the different societal activities and influences, was made among the different reference hydrological zone types. This step provided information regarding the suitability of each of the activities and influences to the different hydrological zone types. Given this information, potential trade-offs *among the different zone types* can be identified for the common good of the catchment. Thereafter, comparison of each of the activities was made between the zones experiencing the greatest and the least hydrological alteration within each reference hydrological zone type. This step provided information regarding potential trade-offs that could be invoked *within each of the zone types*.

Both analyses were restricted by the differences among the reference hydrological zone types regarding the selected flow parameters representing the indices of high information for each zone type (*c.f.* Table 6.9). For example, average flows in high flow season months are important for characterising the streamflow regimes of the LPWS zone type, whereas average flows in both low flow season and high flow season months are important for distinguishing the streamflow regimes of the MPWS and HPWS zone types (*c.f.* Table 6.9). However, the rate of rising streamflow is important to all zone types and this parameter provides a strong contender in the comparison of the different societal influences on the streamflow regimes of the different zones. In addition, not all the societal activities and influences are practised or are present in all of the hydrological zones tested. Nonetheless, with the exception of Zone 5 there is a sufficient mix of different activities for the analyses.

5.4.2 Results of the analysis

Ideally stakeholders require information relating to biomass (*i.e.* crop yield), economic return on crop production, any "unforeseen" impacts and the sustainability of each agri-business to make judicious decisions. As in previous sub-sections, the results presented here relate solely to the hydrological alteration of the hydrological parameters contributing to the high information indices for each zone type. Thus, the analyses are limited. Nonetheless, it is contended that protecting some degree of the natural range of variability in each of the selected parameters is a straightforward approach to maximising both the short-term and long-term benefits of the freshwater resources in the Mkomazi Catchment.

5.4.2.1 Comparison of societal activities among the hydrological zone types

Commercial forestry is practised in five of the zones tested (*c.f.* Table 6.15). Figures 6.17a, 6.20a, 6.23a, 6.25a and 6.28a indicate that, in general, there is little difference among the zone types regarding the hydrological alteration that can be attributed to this agri-business. With the exception of Zone 26 in the MPWS zone type, the differences among these geo-climatic regions are negligible in terms of the degree to which the RVA target window is met for the selected flow parameters. The RVA target window for Zone 26 is attained in close to 50% of the years for average flows in the low flow season months and, while it may be tempting to conclude that this zone is the most suitable for commercial forestry, this attainment is mostly at the expense of the magnitude of flows in high flow years. Consequently, there can be just as much divergence from the natural variability in high flow years in Zone 26 as there is in Zone 10 in the HPWS zone type (*c.f.* Figure 6.33 for average flows in May).

Rain-fed agriculture is also practised in five of the zones tested (*c.f.* Table 6.15). However, there is greater divergence among the zone types as a result of this activity. The hydrological response to rain-fed agriculture in Zones 10 and 16 is quite distinct from that in the other zones. This will be discussed further below in the comparison between Zone 16 with Zone 26. Figure 6.28b indicates that Zone 10 in the high rainfall region experiences similar hydrological alteration in low flow season months to Zone 16 (*c.f.* Figures 6.25b), which is the least altered zone in the MPWS zone type, and less alteration in high flow season months than Zones 33 and 39 (*c.f.* Figures 6.17b and 6.20b) in the LPWS zone type.

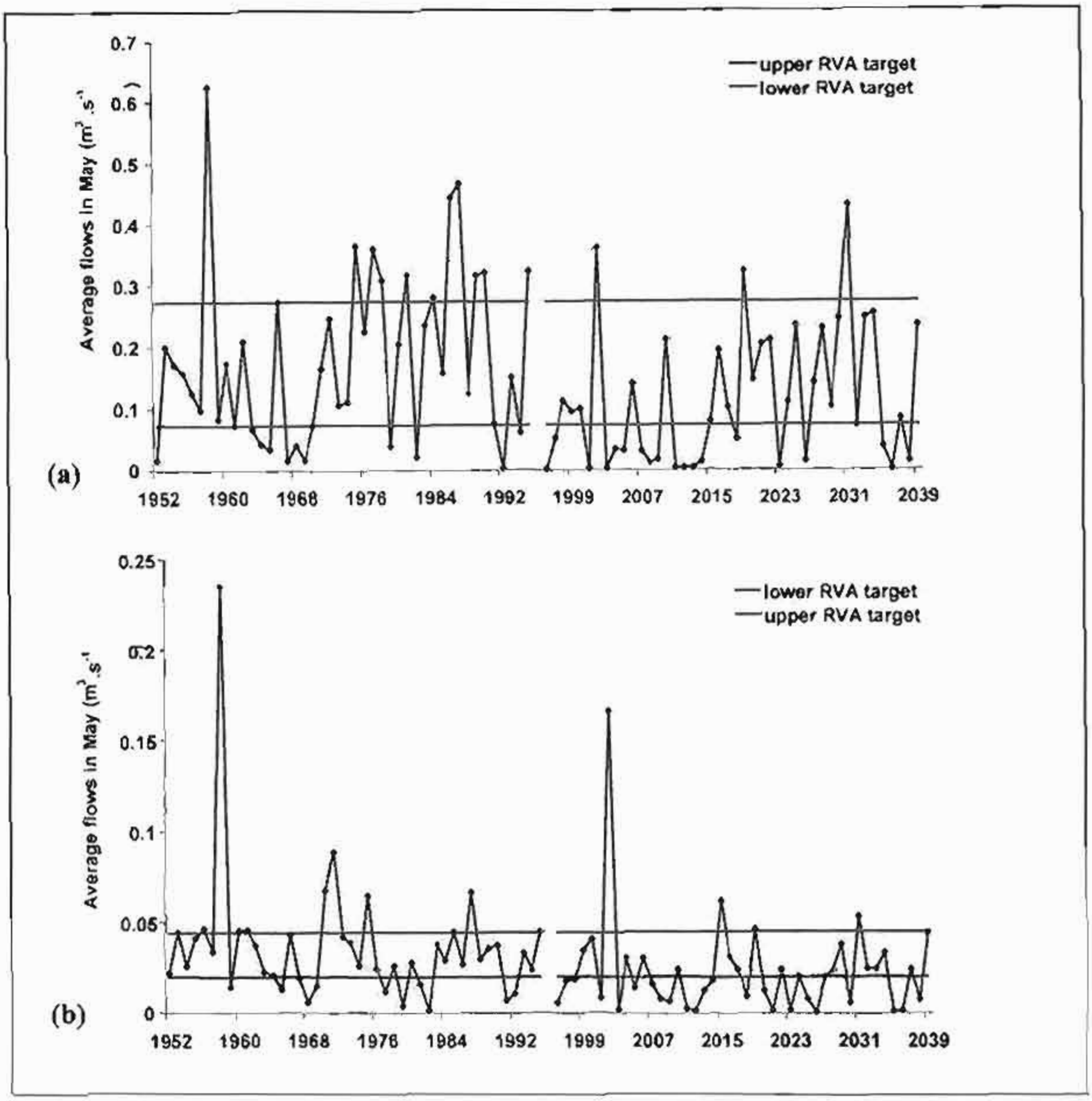

Figure 6.33 Reduced variability in high flow years of the average flows in May associated with commercial forestry in (a) Zone 26 and (b) Zone 10 of the Mkomazi Catchment.

In addition, because the hydrological alteration as a result of rain-fed agriculture in Zone 26, both above and below the RVA target range, is also relatively low in both high flow years and low flow years for most of the selected flow parameters, these streamflow components are closer to pre-development conditions. The exception to this feature is that the rate of change in rising streamflows is just as impacted in Zone 26 as in the other zones. However, this is largely a result of the sensitivity of this parameter to the (moderate) rainfall regime of this MPWS zone type. In general, the analysis of Figures 6.17b, 6.20b, 6.23b, 6.25b and 6.28b indicates that Zone 26 in the MPWS zone type is the most appropriate zone of those tested for sustaining low hydrological alteration as a result of rain-fed agriculture. As mentioned above, a wide array of socio-economic features also need to be considered before any trade-offs among the zones can be contemplated. Nonetheless, this finding is expected to have relevance to the formulation of catchment management plans.

Commercial irrigation is practised in only three of the zones tested (*i.e.* Zones 10, 26 and 33). However, all three water source zone types are represented in the analysis. The discussion in Section 5.3.2.2 revealed that the streamflow regimes of all zones were highly altered by this practice. While there is little difference among the different zones regarding most of the monthly flows and rate of rising streamflow level, the hydrological alteration for average flows in January and February and for the 30-day maximum flows in Zone 10 of the HPWS zone type are less than those in Zone 33 of the MPWS zone type. Thus, it can be deduced that, of the zones tested, Zone 10 in the high rainfall water source area is the most "appropriate" for this operation, at least in terms of minimising the impacts of the practice on the most important components of the streamflow regime found in this zone type. However, there other factors which require consideration before commendations can be made regarding the trade-off potential among the different water source zone types regarding this practice. For example, 50% of the catchment's irrigated land is currently located jointly among Zones 40, 41, 42, 43 and 44, all of which are in the lowest rainfall region. While irrigation is practised precisely *where* there is little rainfall, there may additional hydro-geographical or socio-economic reasons why such a large extent of the catchments irrigation is practiced in these five *tributary* zones. In addition to the socio-economic factors which drive this agri-business, assessing the influence of this practice in different water source zone types is further complicated by the hydrological connectivity embedded in the hierarchical structure of the catchment and river system network. This upstream-downstream relationship is evident for Zones 40, 41, 42, 43 and 44 in the LPWS zone type, and to a lesser extent for Zones 25 and 26 in the MPWS zone type where these hydronomic zones rank in the top nine zones of the most altered streamflow regimes (*c.f.* Figure 6.13). However, all other things being equal, it does seem inappropriate to consider any further development involving irrigation practices in zones of the LPWS zone type.

Unlike the three economically active business and livelihood operations of commercial forestry, rain-fed agriculture and irrigation, the three remaining land covers in the analysis have no value to stakeholders. Moreover, the influence of alien riparian invasive vegetation, encroaching thicket and bushland and degraded grassland all negatively influence the biodiversity of the Mkomazi Catchment and can bear great costs to stakeholders (*e.g.* degradation results in increased sediment yield and "flashy" floods). These negative impacts are either the direct results of societal activity or they are influenced by societal practices. The most powerful incentive for trade-offs among and within each of the different water source zone types lies in the removal of alien vegetation, control of the changes in natural succession processes and rehabilitation of the denuded landscape.

The South African Working for Water Programme for clearing alien invasive riparian vegetation should afford priority to zones in the high rainfall region, since this is where the greatest alterations to the streamflow regimes are experienced, particularly in the high flow season in Zone 10 (*c.f.* Figure 6.28 for average flows in January compared with the same parameter in Figures 6.17 and 6.20 for Zones 33 and 39 in the LPWS zone type). Efforts to restore natural vegetative cover from degraded conditions should be prioritised in any affected zones within the LPWS since this is where the influence of the denuded landscape on the streamflow regime is greatest. For example, alteration in the natural range of the average flows in January and February as well as the 30-day maximum flow as a result of degraded conditions in Zone 39 is greater than that experienced in Zone 10 in the HPWS zone type (*c.f.* Figures 6.20d and 6.28e). However, the situation is complex, with alteration in average flows in the low flow season months in Zone 10 in the HPWS zone type being greater than that experienced in Zone 16 in the MPWS zone type (*c.f.* Figures 6.28e and 6.25d). Stewardship approaches which monitor and control the encroachment of thicket and bushland should be initiated for the HPWS zone type since it is in this high rainfall area that these shrubs and trees have the greatest alteration. For example, the alteration in the natural range of the average flows in January in Zone 10 in the HPWS zone type is greater than that experienced in Zone 39 in the LPWS zone type *c.f.* Figures 6.28f and 6.20e), whereas the alteration in the low flow season months in Zone 10 is greater than that experienced in Zone 26 in the MPWS zone type (*c.f.* average flows in March, May, June and September in Figures 6.28f and 6.23e). Moreover, this thicket and bushland has been dispersed throughout the catchment to areas that are beyond its natural habitat range (*c.f.* Figures 6.5 and 6.6).

5.4.2.2 Comparison between the hydrological sub-zones in different zone types

In this last section of the results of the analyses of the water use of each of the six different land covers, comparisons are made between (a) Zones 33 and 39 in the LPWS zone type and (b) Zones 16 and 26 in the MPWS zone type. The only land cover tested for Zone 5 in the HPWS zone type was alien invasive riparian vegetation. Therefore, the comparisons made between Zones 5 and 10 are restricted to this land cover.

Comparison between Zones 33 and 39 in the LPWS zone type

Commercial forestry has similar impacts on the selected flow parameters which contribute to the hydrological indices of high information of the streamflow regimes in both Zones 33 and 39 of the LPWS zone type (*c.f.* Figures 6.17a and 6.20a). Overall, Zone 39 experiences less alteration in the flow parameters than Zone 33 as a result of this practice, particularly with regard to the average monthly flows at the end of the low flow season (October), baseflow (as measured by the IHA "baseflow" parameter) and the rate of rising streamflow level. However, as mentioned in Section 5.3.2 of this Chapter, the degree of alteration within any of the three categories (below, above and within the RVA target range) has to be considered simultaneously with the alteration within each of the other categories. For example, Figure 6.20a indicates that the IHA baseflow parameter receives no alteration to the frequency of annual values falling below the RVA target window under post-impact conditions, a small increase in the frequency of annual values falling within the RVA target window and a moderate reduction in the frequency of annual values falling above the RVA target window. In essence, this translates into a dampening of baseflows (as measured by the IHA "baseflow" parameter) in Zone 39, with the RVA target window being attained in more years than under reference hydrological conditions (also see Footnote 2, Page 6-91). Nonetheless, in terms of protecting the hydrological status of this LPWS zone type, stakeholders would be justified in choosing Zone 39 in preference to Zone 33 for any further development of commercial forestry in this zone type.

In general, rain-fed agriculture has less impact on the streamflow regime of Zone 39 than Zone 33, although there are exceptions to this distinction between the two zones (*c.f.* Figures 6.17b and 6.20b). Certainly, Zone 33 experiences less alteration in the magnitude of high flows (*i.e.* the 30-day and 90-day maxima events as well as the duration of the high flow pulses) as a result of this practice. However, the alteration in most monthly flows, baseflows (also see Footnote 2, Page 6-91), short-duration maxima events and rate of rising streamflow level is lower in Zone 39. Thus, it can be deduced that it would be prudent, at least in hydrological terms, to choose Zone 39 in preference to Zone 33 for any further development of this agricultural practice.

Comparing Figure 6.17d with Figure 6.20c indicates that the distinctions between Zones 33 and 39 as result of the presence of alien invasive riparian vegetation are much less ambiguous. This opportunistic vegetation has a greater influence on the streamflow regimes of Zone 33 than it does in Zone 39. Priority should be given to clearing this vegetation from Zone 33, particularly since in this zone it has a similar impact on the streamflow regime to that of the economically valuable practice of commercial forestry.

The distinctions between Zones 33 and 39 as result of encroaching thicket and bushland are also relatively clear (*c.f.* Figures 6.17e and 6.20e), with greater alteration to the streamflow regime occurring in Zone 33. This encroaching vegetation has dispersed beyond its natural habitat range to Zone 33 where its impacts on the streamflow regime are nearly as great as those of alien invasive riparian vegetation (*c.f.* Figure 6.17d). While vegetation succession is a natural landscape process, it may be that encroachment in Zone 33 has been facilitated by societal activity. Therefore, while this "natural" vegetation succession process may be considered to be less detrimental than the invasion of alien vegetation, it would be prudent for stakeholders to evaluate the environmental costs and benefits of controlling any further encroachment of this vegetation. Conversely, the impacts of encroaching thicket and bushland on the streamflow regime in Zone 39 are greater than those of alien invasive riparian vegetation. Thus, stakeholders in Zone 39 would benefit from allocating as many resources to controlling this encroachment as they do to clearing alien invasive riparian species.

Additional Comments on Comparison of Zones 33 and 39

According to the CSIR's 1996 LANDSAT TM image for the region, neither commercial irrigation nor degraded grassland are present in both Zones 33 and 39 and no comparisons were drawn between the two zones. Nonetheless, recommendations can still be made regarding their presence in their respective zones and the likely impact on downstream zones. Zone 33 was identified in Section 4.2.2 as having the greatest existing alteration in the LPWS zone type. The results presented in this Section concur with that finding, *viz.*

that societal activities and influences result in greater alteration in Zone 33 than they do in Zone 39. In addition to the water uses discussed, commercial irrigation is practised in Zone 33, with deleterious impacts on the streamflow regime. It would be unwise for stakeholders to consider any further expansion of this practice in this water source zone. Despite the heavy water use incurred by Zone 33, at present Zone 34 downstream experiences a much lower degree of hydrological alteration (*c.f.* Figure 6.13), *because it does not operate commercial irrigation*. Thus, development in Zone 34 could be disadvantaged by the upstream water use of Zone 33.

Degraded grassland has a similar, though lesser, impact on the streamflow regime in Zone 39 to that of rain-fed agriculture. Clearly, stakeholders should take steps to rehabilitate denuded land as it could be used more beneficially to society, without incurring further hydrological alteration. This could have relevance to the small-scale crop production systems which are vital to rural communities in the low rainfall regions on the tributaries of the Mkomazi Catchment. Moreover, a stewardship approach to the prevention of further degradation of the landscape would enhance the hydrological status of Zone 39. Despite Zone 39 being located in the natural habitat zone of "valley bushland" (*c.f.* Figure 6.5), encroaching thicket and bushland has a marginally greater impact on the streamflow regime of this zone than that of alien invasive riparian vegetation. As described in Section 5.3.2.1, this anomaly is a result of the modelling input to the *ACRU* simulations of reference conditions which comprised area-weighting of the two Acocks veld types (Acocks, 1988) represented in Zone 39. However, this factor does not detract from the hydrological relevance of controlling the encroachment of even "natural" vegetation species.

Zone 39 is located downstream of only Zone 38 (*c.f.* Figure 6.14). It can be deduced from this analysis that both Zones 38 and 39 have "very low" existing hydrological alteration (*c.f.* Figure 6.13) because they do not operate commercial irrigation.

Comparison between Zones 16 and 26 in the MPWS zone type

While the patterns in the hydrological alteration as a result of commercial forestry between Zones 16 and 26 are very similar, this practice has a greater impact on all the selected streamflow parameters of Zone 16 than those of Zone 26 (*c.f.* Figures 6.23a and 6.25a). In addition, the degree of hydrological alteration in the low flow season would be detrimental to any further development of commercial forestry in Zone 16. While there is some similarity between the two zones regarding the hydrological alteration of the rate of rising streamflow level, this can be attributed to similarities in the climatic regime of the zones of the moderate rainfall region. Therefore, it can be deduced that Zone 26 is preferable to Zone 16 for further development of commercial forestry.

The differences between Zones 16 and 26 are even more pronounced when one examines the impacts of rain-fed agriculture on the selected streamflow parameters, with greater hydrological alteration being experienced in Zone 16 (*c.f.* Figures 6.22b and 6.24b). Stakeholders would be advised to consider further development of this food production system for Zone 26 in preference to Zone 16. Again, the rate of rising streamflow level is the only parameter for which there is any similarity of the degree of hydrological alteration between the two zones. This confirms the finding above, *viz.* that this streamflow parameter is strongly driven by the rainfall regime of the region.

Clearing alien invasive riparian vegetation from Zone 16 would yield greater hydrological benefits than could be achieved from clearing Zone 26 (*c.f.* Figures 6.23d and 6.25c). Likewise, priority should be given to the control of encroaching thicket and bushland in Zone 16 rather than Zone 26 (*c.f.* Figures 6.23e and 6.25e) where the impacts of both these opportunistic species are less severe.

Additional comments on the comparison of Zones 16 and 26

The analysis of existing alteration among the different zone types identified Zone 16 as having the least alteration within the MPWS zone type and Zone 26 as having the greatest alteration (*c.f.* Section 4.2.2 and Figure 6.13). Therefore, it is unexpected that Zone 26 should be identified as being preferable to Zone 16 when considering the potential for further development of the "streamflow reduction activities" of commercial forestry and rain-fed agriculture. This is compounded by the finding that any conservation or stewardship approaches are needed more in Zone 16 than Zone 26. The only remaining difference in the water useage between these zones lies in the commercial irrigation practised in Zone 26 and the degraded grassland present in Zone 16. Figure 6.23c indicates substantial hydrological alteration in Zone 26 as a result of commercial irrigation, whereas, Figure 6.25d indicates minimal hydrological alteration in Zone 16 as a result of degraded grassland. The analysis performed in this Section excluded upstream contributions. Therefore, the only plausible explanation for the paradox of Zone 26 having greater existing hydrological alteration than Zone 16 lies in the operation of commercial irrigation in Zone 26 *and* in the upstream Zone 25 (*c.f.* Figure 6.13, where Zone 25 has "high" hydrological alteration). This emphasises the relevance of upstream-downstream hydrological connectivity to the hydrological status of the tributaries of the Mkomazi Catchment.

From another perspective, there are plans to construct a rural water supply scheme in Zone 15 upstream from Zone 16 (*c.f.* Figure 6.3). The impoundment of upstream contributions for water abstraction for off-site allocations can be expected to have further restrictions on any further development within Zone 16.

Comparison between Zones 10 and 5 in the HPWS zone type

In general, there is a higher degree of alteration in the selected streamflow parameters of Zone 5 than in Zone 10 as a result of alien invasive riparian vegetation (*c.f.* Figures 6.28d and 6.31). The most notable exceptions to this feature are that average flows in low flow months experience less alteration in Zone 5 than occurs in Zone 10, whereas the timing of the 30-day minimum flow event experiences greater alteration in Zone 5 than it does in Zone 10. Again there is little difference between these high rainfall zones in the rate of rising streamflow level. Given the high hydrological status of Zone 5 (*i.e.* it is the most minimally impacted zone of the Mkomazi Catchment) this zone should be managed with the highest level of conservation and stewardship objectives. Thus, while priority should be given to the control of encroaching thicket and bushland and rehabilitation of the denuded landscape in Zone 10, it is important that the hydrological integrity of Zone 5 be restored to its "natural" state, through the clearing of alien invasive riparian vegetation.

5.5 Summary

The catchment to river network system is largely overlooked in environmental flow assessments. The capacity of the hydrological regime to deliver the ecosystem goods and services associated with a particular ERC depends entirely on the societal activities operating in the catchment. Hydronomic sub-zoning, based on the way in which societal activities disrupt the natural hydrological processes, both off-stream and instream, can be applied at a relatively fine organisational scale to assess the incompatibilities between societal and ecological freshwater needs. Management targets, based on the statistical analysis of pre-development streamflow regimes, can be defined to assess the degree of hydrological alteration from reference hydrological conditions as a result of each societal activity. The RVA analysis applied in this Section provides a valuable tool for investigating which societal activities cause the greatest disruption to the hydrological regime in different water source zones of the Mkomazi Catchment. Because the RVA is based on ecologically relevant hydrological indices, it represents an ecosystem-based approach to guide in the formulation of stakeholder-based catchment management plans.

6 DISCUSSION

The Case Study presented in this Chapter provides a good example of the benefits of revisiting and building on previously researched material. Together, the work presented in the paper titled "The Application of the Indicators of Hydrological Alteration method to the Mkomazi River, KwaZulu-Natal, South Africa" (Taylor *et al.*, 2003) and the work presented in Sections 3, 4 and 5 of this Chapter provided an ecosystem-based approach to maximising the generation of ecosystem goods and services of the Mkomazi Catchment. In addition, the Mkomazi 2001 Study has been complemented by the proposed framework for ecologically sustainable water resources management outlined in Chapter 4 of this thesis, as well as the findings of the study on the high information hydrological indices of variable rivers and the sensitivity of those indices to record length, comprising Chapter 5 of this thesis. It is believed that the Case Study, particularly the work presented in Section 5, presents a new approach to assessing the water development of the tributaries and water source zones of a relatively unregulated catchment.

Knowledge of reference conditions is critical to any assessment of ecological and societal needs for freshwater, since they inherently incorporate ecosystem "resilience", and in a more societal context "sustainability" and "acceptability". Stakeholders can only anticipate any change in these ecosystem functions if the natural streamflow regime is known. The nature, or, degree of societal alteration to hydrological processes and the environmental response can vary greatly from upstream to downstream or across a catchment. This Case Study emphasises the benefits of hydrological modelling in providing long-term reference time series of daily streamflows at a fine spatial or organisational scale.

Environmental flow assessment methods which focus solely on aquatic and riparian ecosystems tend to disaggregate the hydrological functioning of catchment-wide processes from the ecological functioning of the river system. Indeed, stakeholders can only begin to answer the question posed in Chapter 4 on how the hydrological regime delivers the ecosystem goods and services attributable to a particular ERC by investigating how society disrupts the natural hydrological processes by the appropriation of liquid water and water vapour, particularly in the water source zones of the catchment. This requires delineation of the catchment, into *hydronomic* zones which match the spatial and organisational scale of hydrological processes and the hydrological response to societal alteration with management practices.

Richter *et al.* (1998) recommended spatial mapping of the degree of hydrological alteration at a river network for highlighting where alterations to the streamflow regime interrupt or could potentially interrupt the overall system connectivity. Spatial mapping of the hydrological alteration of the Mkomazi River system found that the greatest disruptions to the upstream-downstream hydrological connectivity of the river network arose in the lower tributaries of the Mkomazi. However, the main benefit of assessing the loss of natural hydrological variability at the relatively fine organisational scale applied in this Study is that the information can identify where and why hydrological connectivity is being disrupted. The analysis of the water use of each of the different societal activities and impacts, for different zones types, highlights how stakeholders can be better equipped to make prudent decisions concerning the ecosystem goods and services they consider most appropriate to their short-term and long-term needs. For example, this Case Study demonstrates and concurs with the emerging philosophy, that opportunities exist to produce more food per drop if the focus is changed from the downstream management of in-stream and stored water to the upstream position (*i.e.* the moderate or high precipitation water source zones in this Case Study), where rainfall enters the soil-plant system (Molden *et al.*, 2001).

It is highly relevant that the Case Study applied in this Chapter focuses on the tributaries of the Mkomazi Catchment, where, apart from some farm irrigation dams, there is no storage available for the management of downstream releases. Incompatibilities between ecological freshwater needs and societal freshwater needs on the tributaries of the Mkomazi Catchment (and many other South African catchments) can be more acute than those experienced on the mainstream river (where there also may not be any impoundments from which to operate releases to meet ecological functioning). Thus any compromise between the two systems (ecological and societal) will have to be met from changes to the ways in which water is used, with particular regard to the disruptions to the hydrological regime as a result of various land uses.

The essence of the RVA approach (Richter *et al.*, 1998) is that where there is a paucity of ecological data (as there frequently is for South African river systems) the approach guides the implementation of management target ranges which can be altered in accordance with biotic response, when monitoring information and ecological observations become available (Richter *et al.*, 1997). While the RVA presents an attractive method to address the incompatibilities of meeting societal and ecological freshwater needs, there is a shortcoming to the approach. While the objectives of ecosystem approaches to freshwater management clearly should be based around the concept that *it is not necessary to have the river attain the target range every year,* attaining a "middle" target range at the same frequency as occurred in pre-development conditions may not be sufficient to ensure that the natural variability of the streamflow regime is maintained. It is also prudent to ensure that a *similar* frequency as occurred in pre-development conditions is attained both above and below the middle target range (*i.e.* typically the expectancy of a flow parameter in 50% of the years). Too great a divergence from the frequency occurring in pre-development conditions in high flow years or low flow years could lead to reduction in the natural range of variability. Reducing streamflows to lower than their natural annual range of variation is likely to result in vulnerability of the ecosystem to any climatic or further anthropogenic disturbance.

This Case Study applied the concept proposed in Chapter 4 that "*precipitation is a better starting point for assessing the aquatic resource base than instream flows*". It is possible to match the spatial, organisational and temporal scales of both ecological and societal freshwater needs by focussing on the ways in which society disrupts the hydrological processes and energy pathways of water across a catchment, from the basic resource of precipitation to instream channel flows. Certainly, the work presented in Section 5 of this

Chapter recommends that protection of the hydrological function in aquatic ecosystems should also focus on land care management and society's dependence on water vapour flows rather than solely assessing either "how much must be left in the channel?" or "how much can be taken out of the channel?"

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STRUCTURE OF APPENDIX 6A APPLICATION OF THE INDICATORS OF HYDROLOGICAL ALTERATION METHOD TO THE MKOMAZI RIVER, KWAZULU-NATAL

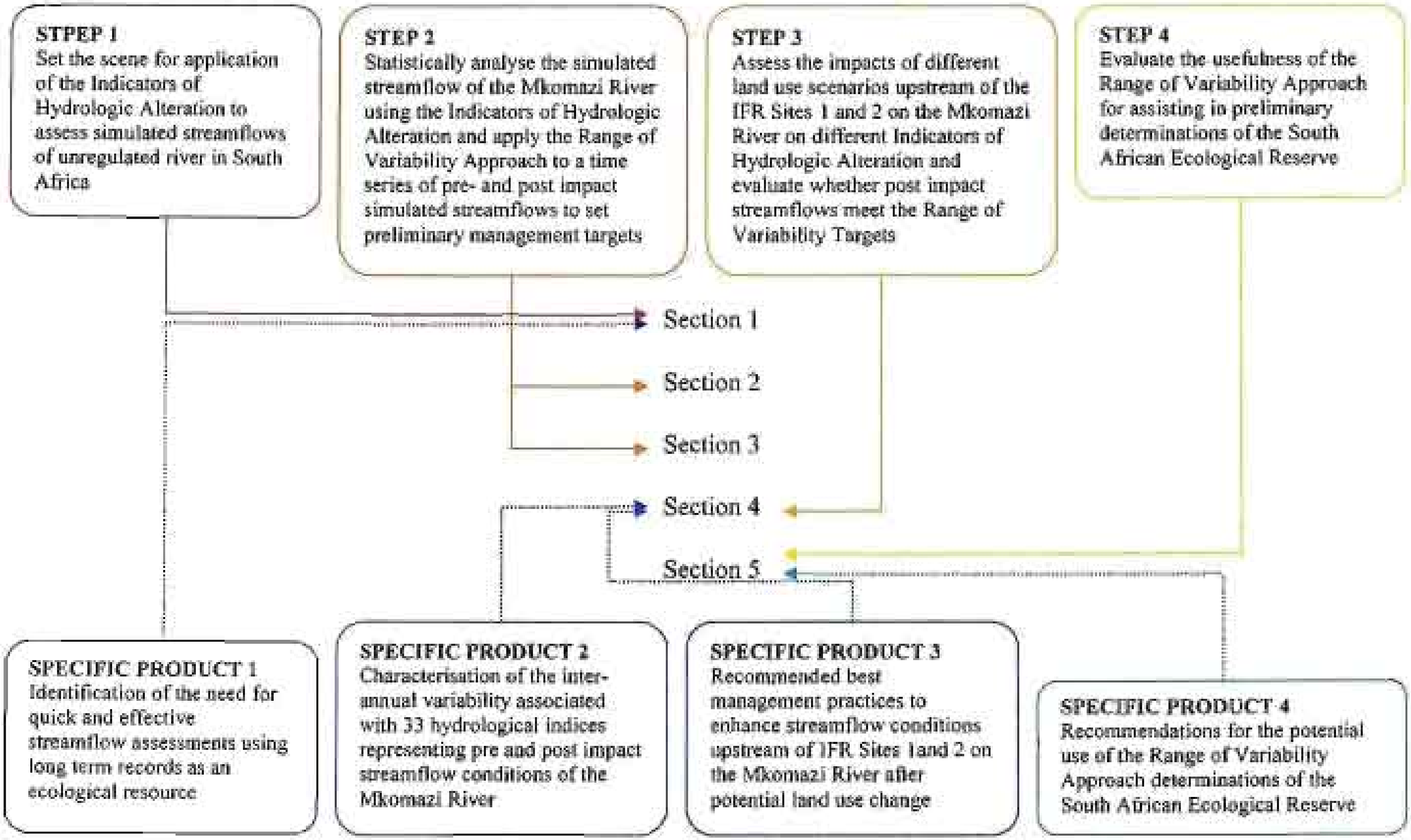

Figure 6.34 Structure of Appendix 6A: Application of the Indicators of Hydrological Alteration method to the Mkomazi River, KwaZulu-Natal

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Application of the Indicators of Hydrological Alteration method to the Mkomazi River, KwaZulu-Natal, South Africa

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Hydrological regimes play a major role in structuring the biotic diversity within river ecosystems and hydrological variation is recognised as a primary driving force within those ecosystems. The US Nature Conservancy developed a method known as the Indicators of Hydrological Alteration, IHA (Richter *et al.* 1996, 1997, 1998), for assessing the degree of hydrological alteration attributable to human induced changes within an ecosystem. The method is based on the statistical analyses of 33 hydrological parameters representing five streamflow characteristics that can be attributed to playing major roles in determining the nature of aquatic and riparian ecosystems (Richter *et al.* 1996, 1997, 1998). The Range of Variability Approach, RVA, is an application of the IHA, incorporating the concepts of hydrological variability and river ecosystem integrity, and was developed to enable river managers to define and adopt preliminary flow management targets before conclusive, long-term ecosystem research results are available. This paper presents an application of the RVA to simulated streamflows at two of the four Instream Flow Requirements (IFR) Sites on the Mkomazi River in KwaZulu-Natal, South Africa, in order to assess the extent of alteration caused by human induced changes to the hydrological regime. The assessment was achieved by comparing the range of variation of the hydrological regime simulated under natural catchment conditions (pre-impact) with the variation resulting from catchment development (post-impact). The 25th and 75th percentile values of each of the 33 parameters were selected as the lower and upper thresholds within which streamflow management targets could be set. By setting preliminary streamflow management thresholds that can be modified and refined when ecological data and information become available, the RVA incorporates flexibility and adaptability.

Keywords: hydrological regime, indicators of variation, aquatic and riparian ecosystems, flow management targets

Introduction

Scientists are challenged by the question of how much water rivers need to maintain aquatic health and to sustain the integrity of aquatic and riparian ecosystems. This has become all the more pressing in South Africa since, under the provisions of the South African National Water Act (NWA), Number 36 of 1998 (NWA 1998), some water must be set aside as an ecological Reserve to protect the ecological functioning of rivers and the resource base itself. In terms of the NWA (NWA 1998), the Minister of Water Affairs and Forestry is responsible, after consultation with stakeholders, for the classification of South Africa's water resources and for determining the quantity and quality of streamflows required to sustain each water resource in a particular classification. This classification is guided by descriptions of ecological management targets for water resources developed by the Department of Water Affairs and Forestry (DWAF 1999). The classification system comprises four ecological management classes, where A represents largely unmodified, natural conditions and D represents highly modified conditions (see Table 1). However, the process for determining the ecological Reserve is not due for publication until 2003, and in the interim, the DWAF has initiated the setting of a preliminary ecological Reserve for water resources planning projects. Expertise to assess the environmental flow requirements of South African freshwater systems is based on knowledge of the influence of flow on stream processes (King and Tharme 1994) and the most sophisticated techniques acknowledge that the natural variability of flow regimes is an important determinant in the ecological functioning of river systems. Hydrological variation is recognised as the major driving force within river ecosystems (Sparks 1995, Stanford *et al.* 1996), since it influences biotic diversity and controls key environmental conditions within aquatic ecosystems (Poff and Ward 1989).

Recognising that hydrological variation plays a major role in structuring the biotic diversity within river ecosystems, the US Nature Conservancy developed the Indicators of Hydrologic Alteration, IHA (Richter *et al.* 1996, 1997, 1998), to characterise the natural range of streamflow variation, based on the analysis of daily streamflow data. The IHA provides a statistical analysis of 33 ecologically relevant hydrological attributes that characterise intra-annual variation in the streamflow regime. The IHA method comprises the statistical analysis of computations of the central tendency (mean or median) and dispersion (standard deviation or coeficient of variation) for each of the 33 hydrological parame-

Table 1: Descriptions of ecological management classes (after DWAF 1999, King *et al.* 2000)

Class	Description of ecological management classes
A	Negligible modification from natural flow regime. Negligible modification from natural conditions with regard to water quality; instream habitat; riparian habitat; diversity and distribution of biota.
B	Slight modification from natural flow regime. Slight risk to intolerant biota with regard to water quality. Few modifications of instream and riparian habitats from natural conditions. Intolerant biota may be reduced in abundance and distribution.
C	Moderately modified flow regime. Moderate risk to intolerant biota with regard to water quality. Moderate modification of instream and riparian habitats from natural conditions. Intolerant biota may be absent from some locations.
D	Largely modified flow regime. High risk to intolerant biota with regard to water quality. High degree of modification of instream and riparian habitats from natural conditions. Intolerant biota unlikely to be present.

ters for each year of record to characterise the inter-annual streamflow variation (Richter *et al.* 1996). The development of the Range of Variability Approach, RVA (Richter *et al.* 1997), arose from the application of the IHA and is intended for use in setting streamflow based aquatic management targets, based on the statistical analysis of the 33 hydrological parameters. The RVA was developed to enable river managers to define and adopt preliminary flow management targets before conclusive, long-term ecosystem research results are available. In this paper the RVA is presented as an approach for setting streamflow based aquatic ecosystem management targets for the Mkomazi catchment in KwaZulu-Natal, South Africa.

Range of Variability Approach

The fundamental premise of the RVA is to guide efforts to restore, or maintain, the natural streamflow regime of a river using a range of inter-annual variation in 33 ecologically relevant flow parameters (Table 2) as the basis for setting streamflow management targets. The RVA recognises the relationship between the characteristics of river flow and river habitat condition and addresses the critical role of hydrological variability in the natural flow regime. To this end the approach considers the magnitude, timing, frequency, duration and rate of change of streamflows (Table 2) assumed to sustain aquatic ecosystems. The developers consider that the approach is most appropriate when protection of the natural aquatic biodiversity and aquatic ecosystem are the primary management objectives (Richter *et al.* 1997). To this extent the approach addresses issues identified by DWAF in its assessment of the preliminary ecological Reserve.

The RVA method

The developers of the RVA identify six fundamental steps for setting, implementing and refining management targets and rules for specific rivers, or river reaches (Richter *et al.* 1997). The following paragraphs summarise the salient points of each step described by Richter *et al.* (1997).

The ***first step*** is to characterise the natural range of streamflow variation using the IHA method described by Richter *et al.* (1996). Daily streamflow records are used to define natural streamflow variability. Where daily streamflow records representing natural conditions are inadequate or incomplete, existing records may be infilled or extended using regression relationships between the site of interest and other less perturbed streamflow-gauging sites (Richter *et al.* 1997). Where no streamflow records exist, the records of reference catchments with adequate record lengths and with similar climate and geology as well as minimal anthropogenic effects can be used. This would require adjustment to the streamflow data, or other statistical characteristics, to account for differences in catchment area and driving variables such as rainfall. Alternatively, a simulation model such as the *ACRU* agrohydrological model (Schulze 1995) could be applied to generate a daily time series of flows to represent defined baseline natural conditions. However, it is imperative that adequate verification of the simulation output is performed to validate the use of the generated time series. The developers of the IHA suggest that a minimum of 20 years of data is required to minimise the effects of inter-annual climatic variation on the IHA parameter statistics (Richter *et al.* 1997). This is based on research by the developers which showed that the range of estimates of the mean annual 1-day maximum from three different stream types in the United States of America begins to narrow substantially when based on at least 20 years of record. However, for southern African conditions, where statistical analyses can be influenced by a particularly wet or dry sequence of years, especially where periodic fluctuations with approximately 20-year oscillations have been identified (Tyson 1987), longer record lengths may be required (Schulze *et al.* 1995).

The ***second step*** comprises the selection of management targets that fall within the natural range of each of the 33 IHA parameters, based on the inter-annual measure(s) of dispersion used in Step 1. Ideally, the management targets should be based on available ecological information. There is, however, a paucity of such information in southern Africa and in such instances the developers recommend that ±1 standard deviation from the mean, or the 25th to 75th percentile range of each of the 33 IHA parameters are selected for preliminary upper and lower flow-based management targets (Richter *et al.* 1997). These recommendations are based on cognisance that targets set at either the maximum or minimum limits of the range of any of the 33 IHA parameters would lead to environmental stress if maintained over a prolonged period of time. Additionally, where humans and their water needs contend with environmental water needs, flow based management targets set close to the mean, or median, values of the 33 IHA parameters would not be considered practical, since human requirements would only be met in half of the years. Although the values selected may

Table 2: Summary of hydrological parameters used in the Indicators of Hydrologic Alteration (IHA) and their characteristics (after Richter et al. 1996)

IHA statistics group	Regime characteristics	Hydrological parameters
Group 1: Magnitude of monthly water conditions	Magnitude Timing	Mean value for each calendar month
Group 2: Magnitude and duration of annual extreme water conditions	Magnitude Duration	Annual minima 1-day means Annual maxima 1-day means Annual minima 3-day means Annual maxima 3-day means Annual minima 7-day means Annual maxima 7-day means Annual minima 30-day means Annual maxima 30-day means Annual minima 90-day means Annual maxima 90-day means Number of zero days 7-day minimum flow divided by mean daily flow for each year ('base flow')
Group 3: Timing of annual extreme water conditions	Timing	Julian date of each annual 1-day minimum Julian date of each annual 1-day maximum
Group 4: Frequency and duration of high and low pulses	Magnitude Frequency Duration	Number of high pulses each year ('high pulse' being defined as those periods within a year when the daily streamflow rises above the 75th percentile of all daily values) Number of low pulses each year ('low pulse' being defined as those periods within a year when the daily streamflow falls below the 25th percentile of all daily values) Mean duration of high pulses within each year Mean duration of low pulses within each year
Group 5: Rate and frequency of water condition changes	Frequency Rate of change	Means of all positive differences between consecutive daily values Means of all negative differences between consecutive daily values Number of rises Number of falls

be considered somewhat arbitrary, the recommendations of *initial* flow based management targets (RVA targets) allow for adjustment of the target range, either on the basis of the results of any monitoring programme or in the event of additional information becoming available (Richter *et al.* 1997) and as such are in keeping with preliminary assessments as recommended by DWAF prior to its publication of the Reserve determination method.

Using the RVA targets as guidelines, the ***third step*** requires that river managers design a management system comprising a set of rules that allow the targets to be met. A management system could include a viable set of reservoir release operating rules, including restrictions on abstractions, or restorative land use practices.

The ***fourth step*** involves the application of a monitoring and ecological research programme to assess the response of ecosystems to the management system described in Step 3. Catchment management strategies in the form of restorative land use programmes (e.g. the removal of alien invasive riparian vegetation) and modifications to reservoir operating rules can be assessed at this stage.

The ***fifth step*** is to apply the IHA method in order to characterise the actual (post-impact) streamflow alteration. Comparisons are made of the values of each of the 33 parameters with the RVA targets to identify whether targets resulting from the implementation of any management system (Step 3) are met. The developers recommend that this be performed on an annual basis to assess the previous year's management system.

The ***final step*** is a reiteration of Steps 2 to 5, incorporating and adapting to the results of the preceding years' management system and any additional ecological research, or monitoring information, required to refine the management system or the RVA targets.

Mkomazi River case study

Catchment description

The Mkomazi catchment comprises the 12 DWAF Quaternary catchments (QCs) numbered U10A to U10M and covers an area of 4 383km². A Quaternary catchment is the smallest areal drainage unit used by the South African Department of Water Affairs and Forestry. The QCs are numbered alphanumerically in downstream order. The areal extent of a QC varies according to the hydrological complexity of the catchment and runoff. The greater the runoff volume, the smaller the QC and *vice versa* (Midgley *et al.* 1994). The catchment is situated around 29°17'24"E and 29°35'24"S (Figure 1), stretches 170km from 3 300m altitude in the northwest to sea level in the southeast and has a Mean Annual Precipitation (MAP) ranging from 1 283mm to 752mm. The MAP is higher in the upper, higher altitude

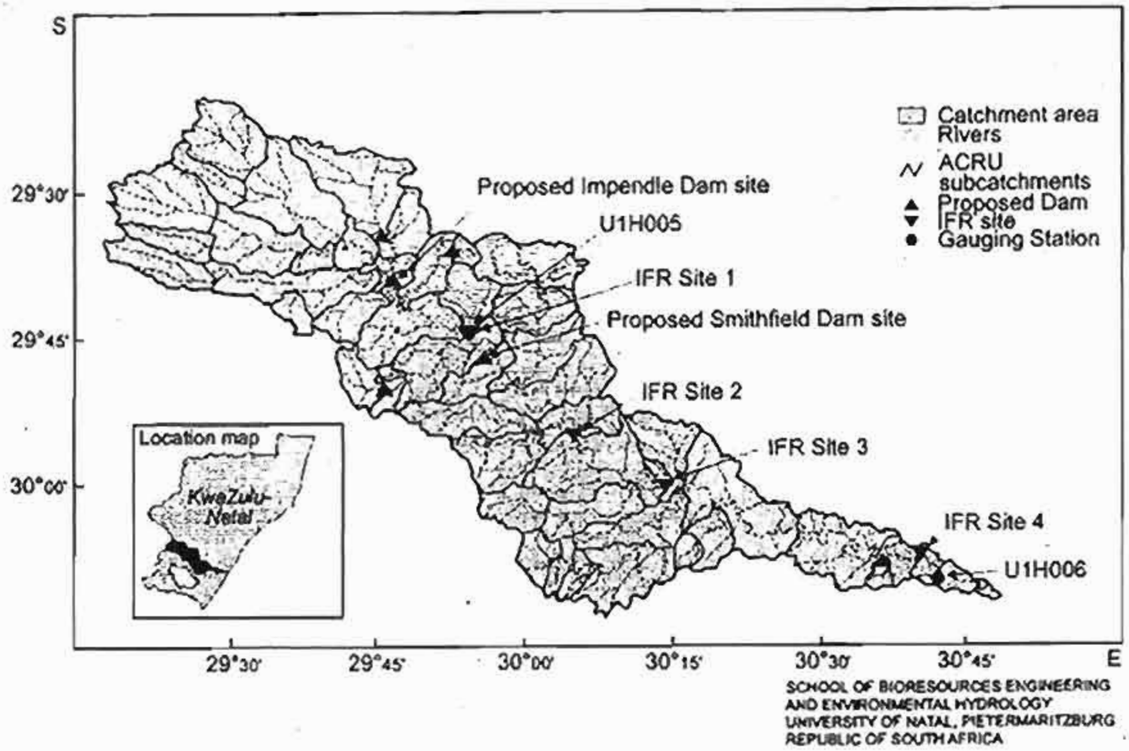

Figure 1: Mkomazi catchment: location and sites of interest

reaches of the Mkomazi catchment (950mm to 1 283mm) and, consequently, most of the catchment runoff is generated there (DWAF 1998a). The Mkomazi catchment is characterised by steep gradients of altitude and rainfall, highly variable land uses as well as highly variable intra- and inter-seasonal streamflows. The annual water yield of the Mkomazi System under present land use conditions and consumption rates has been estimated to be 905 *million* m³ (DWAF 1998a).

Currently (2002) the catchment supports commercial afforestation, extensive agriculture (principally livestock grazing and sugarcane), intensive agriculture (citrus and vegetables), subsistence agriculture as well as tourism and leisure activities. With the exception of a SAPPI paper mill at the estuary mouth and the coastal town of Umkomaas, there are no major towns or industry in the Mkomazi catchment. The distribution of rural population ranges from moderate to sparse, with a greater concentration in the lower catchment (DWAF 1998a). Use of available water resources is, therefore, considered by DWAF (1998a) to be conservative. Whilst at present there are no major reservoirs within the catchment, six potential impoundment sites have been identified by DWAF. Of these, four have been identified as potential development sites for the supply of piped water to rural communities within the catchment and two have been identified as potential sites for storage for inter basin transfer out of the Mkomazi catchment.

Despite the variability of the streamflows, the Mkomazi River flows throughout the year. It has been projected that by 2008 some form of impoundment of the Mkomazi River will be required to augment the water supply in the neighbouring Mgeni catchment to meet the water demand of the Durban-Pietermaritzburg region, one of South Africa's key industrial and commercial areas (DWAF 1998a). Two sites on the Mkomazi River have been identified as potential major impoundment sites: the proposed Impendle and Smithfield Dam Sites (Figure 1). This so-called Mkomazi-Mgeni Transfer Scheme will impact on the streamflow regime of the Mkomazi.

Hydrological modelling

The hydrological dynamics of the Mkomazi catchment were modelled with the ACRU model to assess the impacts of land use changes and proposed developments on the availability of water resources. ACRU is a physical-conceptual model that uses physically realistic and observationally derived variables on climate, soils, vegetation, catchment characteristics, irrigation and dams. Structurally and conceptually ACRU has, therefore, been designed to simulate scenarios of, *inter alia*, land use change. In this study, the Mkomazi catchment was configured to represent 52 major inter-linked sub-catchments. This configuration is based essentially on a division of the 12 DWAF QCs and focuses on different land uses and management practices as well as proposed development concerns. The final model configuration includes, *inter alia*, two major proposed dam sites (Impendle and Smithfield) and four instream flow requirement (IFR) sites, all on the Mkomazi River (Figure 1). Impact of land use and development scenario studies, using the ACRU model, were performed to simulate the impacts of (a) baseline land cover, (b) present land use and (c) present land use, but with the first phase of the proposed MMTS in place. The first phase of the MMTS comprises inter-basin transfer from the proposed Smithfield Dam via a tunnel to an existing dam in the Mgeni catchment. For the purposes of this study:

(a) Baseline land cover conditions were defined as Acocks' Veld Types (Acocks 1988), which are assumed to represent natural land cover conditions in South Africa.
(b) Present land use conditions were defined in accordance with Thompson's (1996) land cover classification and the interpretation of the 1996 LANDSAT TM image for South Africa. In the upper Mkomazi catchment (i.e. the area upstream of the proposed Smithfield Dam and, in particular, upstream of gauging station U1H005; Figure 1), present land use conditions do not vary substantially from indigenous land cover. The upper Mkomazi catchment is sparsely populated, however, the greatest change in land use from baseline conditions is semi-commercial subsistence farming and areas of degraded grassland. Changes from indigenous land cover become more pronounced further downstream with the operation of commercial forestry and irrigated agriculture. Unimproved grassland (Thompson (1996), based on Acocks' term for unaltered indigenous grassland) represents 82% of the land cover upstream of IFR Site 1, but only 70% of the land cover upstream of IFR Site 2.
(c) Daily streamflows were generated with the ACRU model on the assumption that under Phase 1 of the MMTS, the total dam storage capacity is 137 x 10^6m^3 with a daily draft of 604.8 x 10^3m^3 out of the Mkomazi system (DWAF 1998a). This inter-basin transfer represents 30% of the mean annual flows resulting from present land use at the proposed dam site. In the absence of defined reservoir operating rules, legal flow releases as required by the old Water Act of 1956 were simulated for downstream riparian and other users. Currently (2002), legal flow releases are routinely simulated in the ACRU model at 1/1500 x the full storage capacity, for each day, on two conditions incorporated in the ACRU model, viz.
   - if the total streamflow into the reservoir on a given day is less than the legal flow releases, the releases are reduced to equal those of the total inflows and
   - if the storage volumes are below the dams dead storage level (usually 10% of full supply capacity), no legal flow releases are made.

The legal flow releases from the proposed Smithfield Dam for downstream use were, therefore, assumed to be 91.33 x 10^3m^3 each day. Within the ACRU model, overflow from the dam occurs when full storage capacity is reached. Consequently, overflows in periods of high flows supplement the legal releases made for downstream use.

Verification studies were performed to compare the ACRU-simulated, present land use streamflows with the observed streamflow records at the sub-catchments representing the DWAF gauging stations U1H005 (Camden) and U1H006 (Goodenough) on the Mkomazi River (Figure 1). The difference between the median annual ACRU-simulated, present land use streamflows (631 million m^3) and the observed streamflows (594 million m^3) at U1H005 is only 6%. At U1H006, the difference between the median annual ACRU-simulated, present land use streamflows (1 035 million m^3) and the observed streamflows (747 million m^3) is 39%. However, the DWAF gauging weir at U1H006 experiences over-topping and produces unreliable high flow measurements. Furthermore, other sources estimate the present MAR at U1H006 to be 956 million m^3 (a BKS study, cited in DWAF 1998b) and 973 million m^3 (Water Resources 90, also cited in DWAF 1998b) for the period 1920 to 1995. This confirms that the ACRU simulated streamflows at Goodenough deviate by only 8% and 6% respectively from those estimates.

The instream flow requirements for the Mkomazi River were assessed in March 1998 (DWAF 1998b) by an IFR workshop using a technique known as the Building Block Methodology, BBM (Tharme and King 1998, King *et al.* 2000). The BBM workshop participants defined and determined the ecological management class with regard to the 'present ecological state' as well as a 'desired future state' of representative reaches of the Mkomazi River. They focused on four unique sites at which streamflow characteristics relating to flow regime, water quality, instream habitat, riparian habitat and biota required to maintain the integrity of the aquatic ecosystem were evaluated.

IFR Site 1 is upstream of the proposed Smithfield Dam, whereas IFR Sites 2, 3 and 4 are all downstream of the proposed Dam site (Figure 1). In this paper, the hydrological alteration and preliminary RVA management targets of streamflows at the upper two sites will be discussed, since these sites represent alteration resulting from present land use (IFR Site 1) and from inter-basin transfer (IFR Site 2 being the uppermost of the three downstream IFR Sites).

The upper part of the Mkomazi catchment is largely undeveloped, and the upper reach of the river is in relatively good condition. Therefore, the BBM workshop participants attributed a category C/B (Table 1) to the present ecological state of the river reaches in which both IFR Sites 1 and 2 are situated. However, the workshop participants considered it important that the habitat, instream and riparian integrity of these reaches be protected from degradation. Therefore, the workshop set an ecological management class of category B for the reaches upstream of both IFR Sites 1 and 2, indicating that the river upstream of both these sites should be managed to allow only slight modifications (Table 1) from natural conditions. The BBM workshop (DWAF 1998b) recommended the magnitude and duration of flows required, on a month-by-month basis, at each IFR site to meet the predetermined ecological management class B in a maintenance year (i.e. a year with average flows) and a drought year (i.e. a year in which ecosystems are stressed).

Application of the RVA for the Mkomazi streamflows

In this study, undertaken independently of the Mkomazi BBM Workshop, the RVA was applied as a preliminary assessment of management targets for the Mkomazi catchment streamflows at IFR Sites 1 and 2. This comprised the comparison of the hydrological regime at both IFR sites under baseline land cover conditions (pre-impact conditions) with different land use and development scenarios (post-impact conditions). The hydrological regimes for both pre- and post-impact conditions, at both sites, were simulated using the ACRU model to generate time series of daily flows. Schulze *et al.* (1995) show a map for southern Africa indicating the minimum rainfall record lengths required to ensure that the means of annual rainfall estimates are within 10% of the long term mean 90% of the time. For the region comprising the Mkomazi catchment, they suggest a minimum MAP record length of 20 years. Furthermore, Schulze *et al.* (1995) surmise that for a daily model such as ACRU, the ideal minimum record lengths require to be double those for MAP as given in the map. A 51-year time series of daily rainfall was applied using the ACRU model to simulate daily streamflows resulting from different land use and development scenarios.

For IFR Site 1 the comparison was made between ACRU simulated daily streamflows from baseline conditions and from present land use. For IFR Site 2 the comparison was made between ACRU simulated daily streamflows from baseline conditions and the streamflow regime after the construction of Phase 1 of the Mkomazi-Mgeni Transfer Scheme. The pre- and post-impact streamflows in both comparisons were simulated using the same climatic data and information. Each RVA analysis was performed using non-parametric statistics (percentiles) and the programme output was provided in the form of a summary scorecard, percentile statistics, an annual summary and graphs. Typically, the RVA scorecards provide the following information (Richter *et al.* 1997):

(a) Columns 1–4 show the median, coefficient of variance, and low and high extreme values for each parameter during the pre-impact period.

(b) Columns 5–8 show the same information for the post-impact period.

(c) Columns 9–10 show the low and high RVA targets. These are by default the 25th and 75th percentile values. These percentiles are used to set preliminary ecosystem management targets. The user may define different RVA targets, based on appropriateness to local management plans, or as additional hydrological and ecological information becomes available. If the target falls outside the range of the pre-impact data, it is replaced by the pre-impact range limit.

(d) Column 11 shows the hydrological alteration, defined as

$$\frac{\text{Observed} - \text{Expected}}{\text{Expected}}$$

where:

Expected = the frequency (i.e. number of years) with which annual statistics fall within the RVA limits in the pre-impact period, and

Observed = the frequency (i.e. number of years) with which annual statistics fall within the RVA limits in the post-impact period.

The second panel of the table provides a comparison of the data within, above and below the RVA range for the pre- and post-impact periods. Expected and observed frequencies and the hydrologic alteration factor are shown for the values above the RVA limits, below and within the limits. Where a yearly value is equal to either threshold limit, the RVA analysis places the occurrence *within* the range limits. Where this occurs, warnings are printed at the bottom of the table.

Based on the recommendations outlined in Step 2 of the RVA procedure described above, preliminary upper and lower thresholds of streamflow-based management targets for the Mkomazi River were set at the 75th and 25th percentiles of the range of natural variation for each of the 33 IHA parameters.

Results of RVA application at Mkomazi IFR Site 1

The statistics in Table 3 show that management plans to enhance the hydrological regime at IFR Site 1 should focus on attempts to increase flows in low flow months. Of these months, streamflows resulting from present land use in the winter months of July through to September show the highest alteration from the set RVA range with the greatest number of *below* range years (i.e. monthly mean of daily flows less than the lower RVA target of the 25th percentile). This is illustrated in Figure 2a for July. Correspondingly, the alteration from the RVA range for extreme flows, shown in Table 3, is greatest for the shorter minimum day and multi-day flows respectively, with a substantial increase in the number of years in which the 1-, 3- and 7-day minimum flows do not meet the lower threshold. Table 3 shows that there is also some depression of baseflows. However, most of the sea-

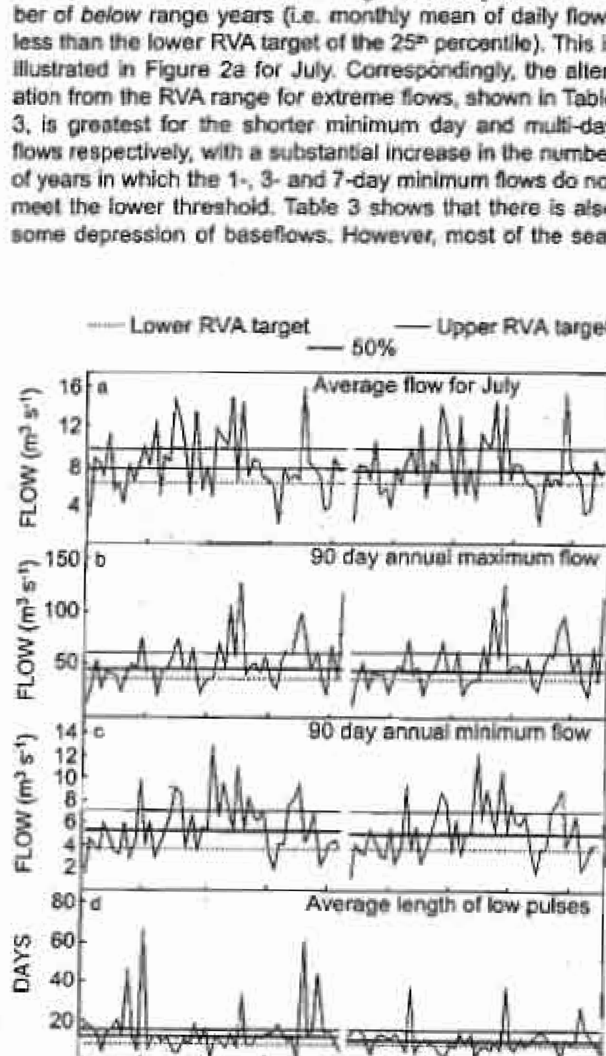

Figure 2: Examples of changes in the hydrological regime, with potential upper (75th percentile) and lower (25th percentile) RVA management targets, at IFR Site 1 on the Mkomazi River

Table 3: RVA score card: Mkomazi site 1

	Pre-impact period October 1894 - September 1944 (51 years)				Post-impact period October 1945 - September 1996 (51 years)				RVA TARGETS		HYDROLOGIC ALTERATION
			Range Limits				Range Limits				
Parameter	Medians	Coefficient of Variance	Low	High	Medians	Coefficient of Variance	Low	High	Low	High	
Group 1											
October	6.50	1.29	1.50	81.60	6.00	1.34	1.20	79.50	3.45	11.75	-0.04
November	13.70	0.88	3.00	43.70	13.30	0.89	2.70	42.80	8.67	20.71	-0.08
December	22.40	0.64	4.30	89.10	21.80	0.83	4.00	86.50	14.76	33.52	-0.08
January	37.00	0.70	6.80	118.20	35.90	0.71	6.30	115.90	25.26	51.29	-0.08
February	44.30	0.64	8.40	120.60	42.90	0.67	8.20	119.30	30.48	58.95	-0.12
March	42.00	0.52	13.90	153.60	40.40	0.53	13.10	151.60	31.27	53.06	-0.04
April	27.60	0.53	6.50	72.00	27.20	0.54	5.40	71.70	19.84	34.55	0.00
May	16.40	0.57	3.40	95.00	16.00	0.56	3.20	93.40	12.67	21.98	0.04
June	11.10	0.45	2.30	25.20	10.60	0.46	2.10	24.80	8.55	13.58	0.00
July	7.70	0.42	2.10	44.50	7.40	0.43	1.90	43.20	6.30	9.57	-0.15
August	5.80	0.68	1.80	29.40	5.40	0.69	1.60	28.30	4.52	8.45	-0.08
September	5.20	1.04	1.40	113.70	4.80	1.06	1.20	111.90	3.57	8.94	-0.12
Group 2											
1-day minimum	2.40	0.51	0.70	7.30	1.80	0.60	0.50	6.60	1.96	3.18	-0.58
3-day minimum	2.40	0.54	0.80	7.30	2.00	0.61	0.60	6.80	1.99	3.29	-0.38
7-day minimum	2.60	0.52	0.80	7.50	2.30	0.61	0.60	7.00	2.18	3.55	-0.38
30-day minimum	3.10	0.83	1.10	9.00	2.60	0.92	0.80	8.40	2.25	4.78	-0.04
90-day minimum	5.30	0.65	1.40	12.90	6.00	0.67	1.20	20.70	3.67	7.08	0.04
1-day maximum	134.20	0.81	48.20	1166.90	130.60	0.81	45.80	1166.90	102.15	211.18	0.00
3-day maximum	114.10	0.77	43.50	882.10	109.50	0.78	41.10	873.10	85.76	173.45	-0.19
7-day maximum	85.70	0.79	38.20	568.40	82.10	0.80	37.50	560.90	65.60	133.12	0.04
30-day maximum	60.60	0.58	16.90	182.20	58.90	0.58	16.00	176.10	45.75	81.04	-0.08
90-day maximum	45.20	0.53	12.00	126.60	43.50	0.53	11.40	124.40	36.04	59.90	-0.04
Number of zero days	0	0	0	0	0	0	0	0	0	0	0.00
Base flow	0.13	0.45	0.05	0.31	0.12	0.44	0.04	0.30	0.10	0.16	-0.04
Group 3											
Date of minimum	274.00	0.06	227.00	340.00	274.00	0.07	224.00	339.00	268.75	297.50	0.00
Date of maximum	39.50	0.15	2.00	361.00	39.50	0.15	2.00	361.00	33.76	163.50	0.00
Group 4											
Low Pulse Count	5.50	0.73	0.00	17.00	7.00	0.57	1.00	18.00	4.00	8.00	-0.17
Low Pulse Duration	11.60	0.58	0.00	66.00	10.30	0.56	1.00	38.00	8.65	15.46	0.15
High Pulse Count	7.50	0.53	2.00	24.00	8.00	0.66	2.00	23.00	6.00	10.00	-0.19
High Pulse Duration	9.80	1.31	2.20	46.50	9.50	1.13	2.00	46.00	5.42	18.19	0.06
Group 5											
Rise rate	8.30	0.51	3.40	25.70	5.40	0.47	2.20	17.10	6.17	10.38	-0.66
Fall rate	-2.30	-0.50	-6.50	-1.00	-2.70	-0.55	-7.40	-1.30	-2.95	-1.79	-0.08
Number of reversals	103.00	0.12	84.00	128.00	164.50	0.27	118.00	211.00	96.75	109.50	-1.00

Comparison of Statistics Within, Above and Below RVA Range

	Within RVA Range		Hydrologic	Above RVA Range		Hydrologic	Below RVA Range		Hydrologic
Parameter	Expected	Observed	Alteration	Expected	Observed	Alteration	Expected	Observed	Alteration
Group 1									
October	26	25	-0.04	13	12	-0.08	12	14	0.17
November	26	24	-0.08	13	13	0.00	12	14	0.17
December	26	24	-0.08	13	13	0.00	12	14	0.17
January	26	24	-0.08	13	13	0.00	12	14	0.17
February	26	23	-0.12	13	13	0.00	12	15	0.25
March	26	25	-0.04	13	13	0.00	12	13	0.08
April	26	26	0.00	13	12	-0.08	12	13	0.08
May	26	27	0.04	13	11	-0.15	12	13	0.08
June	26	26	0.00	13	12	-0.08	12	13	0.08
July	26	22	-0.15	13	12	-0.08	12	17	0.42
August	26	24	-0.08	13	11	-0.15	12	16	0.33
September	26	23	-0.12	13	10	-0.23	12	18	0.50
Group 2									
1-day minimum	26	11	-0.58	13	12	-0.08	12	28	1.33
3-day minimum	26	16	-0.38	13	12	-0.08	12	23	0.92
7-day minimum	26	16	-0.38	13	12	-0.08	12	23	0.92
30-day minimum	26	25	-0.04	13	12	-0.08	12	14	0.17
90-day minimum	26	27	0.04	13	10	-0.23	12	14	0.17
1-day maximum	26	26	0.00	13	12	-0.08	12	13	0.08
3-day maximum	26	21	-0.19	13	13	0.00	12	17	0.42
7-day maximum	26	27	0.04	13	11	-0.15	12	13	0.08
30-day maximum	26	24	-0.08	13	12	-0.08	12	15	0.25
90-day maximum	26	25	-0.04	13	12	-0.08	12	14	0.17
Number of zero days	51	51	0.00	0	0	0.00	0	0	0.00
Base flow	26	25	-0.04	13	10	-0.23	12	16	0.33
Group 3									
Date of minimum	26	26	0.00	13	11	-0.15	12	14	0.17
Date of maximum	26	26	0.00	13	13	0.00	12	12	0.00
Group 4									
Low Pulse Count*	30	25	-0.17	11	21	0.91	10	5	-0.50
Low Pulse Duration	26	30	0.15	13	4	-0.69	12	17	0.42
High Pulse Count*	31	25	-0.19	9	13	0.44	11	13	0.18
High Pulse Duration	26	28	0.08	13	10	-0.23	12	13	0.08
Group 5									
Rise rate	26	11	-0.58	13	5	-0.62	12	35	1.92
Fall rate*	26	24	-0.08	13	5	-0.62	12	22	0.83
Number of reversals	26	0	-1.00	13	51	2.92	12	0	-1.00

*WARNING	Use caution in interpreting expected and observed compliance rates
Low Pulse Counts	7 yearly values are equal to the upper RVA limit
Low Pulse Counts	9 yearly values are equal to the lower RVA limit
High Pulse Counts	7 yearly values are equal to the upper RVA limit
High Pulse Counts	11 yearly values are equal to the lower RVA limit
Number of Falls	6 yearly values are equal to the lower RVA limit

sonal flows remain *within* the target range, as shown in Figures 2b and 2c. Table 3 shows that the frequency of low pulses increases as a result of present land use, with 21 yearly values of counts (compared with the pre-impact of 11) being *above* the upper RVA target. Although the frequency of low pulses increases, there is less variability associated with the duration of these pulses as a result of present land use, fewer of which (four post-impact compared to 13 pre-impact) now occur *above* the upper RVA target set (Figure 2d and Table 3), further substantiating the need for management plans to address the performance of present low flows.

High pulse counts and their durations are less impacted by present land use, but together with the alteration in low pulse counts and their durations, result in alteration in the hydrograph rise and fall rates. This results in all yearly numbers of hydrograph reversals occurring *above* the upper RVA target, indicating an increase in intra- and inter-annual environmental variation. Summer high flow months show less alteration than winter low flow months, with only slight increases in the occurrence of flows *below* the target range. Moreover, the daily and multi-day maximum extreme flows are, generally, very similar to those under natural conditions.

The RVA analysis of hydrological variation generated warnings regarding the yearly values of the low and high pulse counts and number of hydrograph falls equal to either the upper or lower RVA limits (Table 3). These occurrences, for both pre- and post-impact analysis, have been included as being *within* the target range limits. For example, over the entire record period, seven of the yearly values of low pulse counts are equal to the upper target of eight, and nine yearly values are equal to the lower target of four. The statistical relevance associated with this warning is that three of the yearly values of low pulse counts under natural flow conditions were equal to the upper RVA limit and eight were equal to the lower RVA limit. While *thresholds* for statistical analysis have to be set, the RVA table results should be viewed with caution where the warnings are generated by the calculation. However, the results do not detract from the general trend of the hydrological alteration and are, therefore, still valuable for assessing whether the management targets can be achieved.

Based on the RVA analysis, it can be recommended that catchment management plans to enhance the conditions of the river reaches upstream of IFR site 1 on the Mkomazi River to a class B ecological management category from a present state class C/B should include objectives to:

(a) maintain historical winter low flows,
(b) elevate the baseflow regime,
(c) decrease the frequency of low pulses and increase their duration,
(d) decrease the frequency of hydrograph reversals resulting from the increased number of shifts between rising and falling flow levels and
(e) adjust the rate at which daily flows rise or fall

The following Best Management Practices (BMPs) could be adopted in catchment management strategies to meet these objectives for the upper Mkomazi:

(1) The removal of alien riparian vegetation to restore the baseflow regime: alien riparian vegetation has been shown to use more water than the indigenous vegetation with which it competes. In a study on the impacts of the removal of alien riparian vegetation in the Pongola catchment, KwaZulu-Natal, Jewitt *et al.* (2002) found that the most significant improvements in streamflow generation were obtained in the drier winter months;
(2) The initiation of more water use efficient agriculture in periods of low flow, including improved irrigation scheduling systems (e.g. Schulze *et al.* 1999);
(3) The rehabilitation of degraded land to increase infiltrability of water into the soil, thereby allowing the timing and duration of surface and subsurface flow reaching the stream channel to become more natural.

If the ecological management category of the Mkomazi River is to be enhanced, careful consideration should be given to the implications of issuing additional licences for irrigation or afforestation. The adoption of any management strategy would benefit greatly from the initiation of a monitoring and research programme to determine the biotic responses to the implementation of the management system (Richter *et al.* 1997).

Results of RVA application at Mkomazi IFR Site 2

Given the assumptions of the legal flow releases described above, Table 4 indicates that the hydrological alteration from the RVA target range set for post-dam streamflows at IFR Site 2 for the winter low flow months is greatest from June through to October. However, the impact of catchment development with present land use, and Phase 1 of the MMTS, is such that the low flow season is extended to include May and November, together with a substantial alteration of December flows. Figure 3a shows that the decrease in July streamflows shifts the majority of yearly values to *below* the lower target (25^{th} percentile), from 12 occurrences under natural land cover conditions to 44 in post-dam conditions (cf. Table 4). For all winter low flow months there are substantial reductions of occurrences of streamflows *within* the RVA target range, yet only slight reductions in summer high flow months, e.g. January and February both decrease from 26 occurrences to 24 and 22 occurrences respectively, whereas the number for March remains the same at 26 occurrences (Table 4).

The alteration of the magnitude of minimum flows as a result of catchment development with present land use and Phase 1 of the MMTS is considerable, with all annual occurrences of the 1-, 3-, 7- and 30-day minimum flows falling *below* the lower RVA target. Figure 3b illustrates this for the 30-day minimum flow. The 1-day and multi-day maximum flows are far less impacted and most still fall *within* the RVA target range. Table 4 also indicates the extent of suppression of the baseflow regime after the construction of the dam, with most occurrences *below* the lower RVA threshold.

With present land use and the operation of Phase 1 of the MMTS, the number of years in which low pulse counts are *within* the RVA range is increased (change from 28 to 33, Table 4) at the expense of those years *below* the lower target (change from 11 to 4). Furthermore, the average length of low pulses is much longer, with most durations being *above* the upper RVA threshold (Figure 3c), (viz. 38 occur-

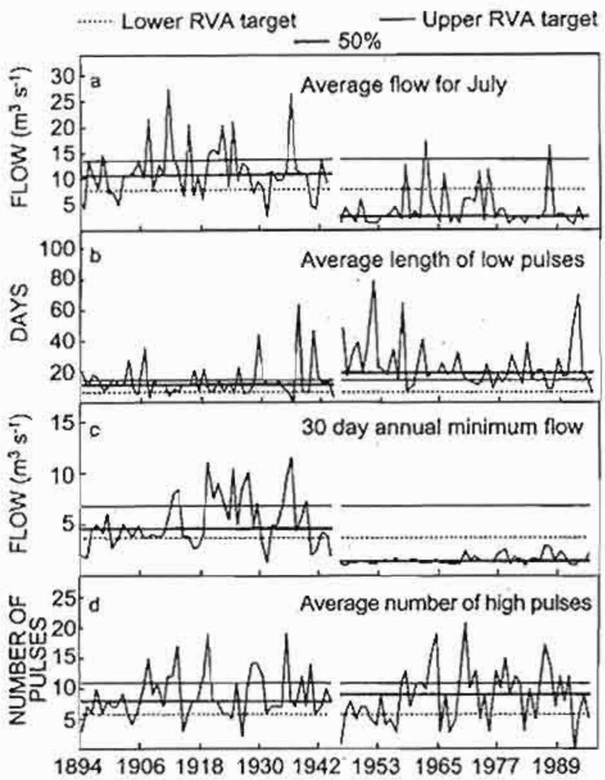

Figure 3: Examples of changes in the hydrological regime, with potential upper (75th percentile) and lower (25th percentile) RVA management targets, at IFR Site 2 on the Mkomazi River

rences, as shown in Table 4). This concurs with the findings described above, that the low flow season is considerably extended under post-dam conditions. The reverse can be shown in Table 4 for high pulses and their durations and in Figure 3d, where the number of years in which pulse counts are *below* the lower RVA target increases from five to 16 after the construction of the dam. The number of years in which high pulse durations are *below* the lower threshold increases from 12 to 21 (Table 4). The decline in high pulses and their durations can be attributed principally to the attenuation of high flows by the dam, the inter-basin transfer and to the restrictions imposed by the legal flow releases for downstream use assumed in the ACRU model simulation as described above. The extent of alteration of the low and high pulses and their durations under post-dam conditions results in high alteration in the hydrograph rise and fall rates. Under such conditions, all yearly numbers of hydrograph reversals occur *above* the upper RVA target, indicating that the modified regime is characterised by substantial changes.

The RVA analysis generated warnings regarding the interpretation of the yearly values of both the high and low pulses, duration of low pulse counts, number of hydrograph falls and reversals as well as the date of the maximum daily flow (cf. Table 4). For example, of the 34 yearly values of pre-impact high pulse counts calculated to be *within* the target range of six to 11, nine were equal to the lower threshold (6). However, as discussed above, this factor does not detract from the general trend of reductions in high pulses following the construction of the Smithfield Dam for inter-basin transfer.

Similarly to Mkomazi IFR Site 1, the BBM Workshop recommended that river management plans for the river reach upstream of the Mkomazi IFR Site 2 require the ecological management category to be enhanced from a present state class C/B to class B. The impact of present land use with the proposed Smithfield Dam clearly exerts more influence on downstream flows than present land use conditions on flows at IFR Site 1. Notwithstanding the influence of land use practices on flows upstream of IFR Site 2, management plans for this river reach should look to efficient reservoir operating rules for the Smithfield Dam in order to ameliorate the impacts of the inter-basin transfer on the streamflow regime. Based on the RVA analysis, operating rules should particularly address:

(a) maintenance of the historical winter low flows,
(b) increased releases at the start and end of the low season, to limit the low flow season to a more natural duration,
(c) elevation of the baseflow regime,
(d) decreasing the frequency of low pulses and their duration,
(e) increasing the frequency of high pulses and their duration,
(f) decreasing the frequency of hydrograph reversals attributable to the current operating rule of legal flow releases, and
(g) adjustment of the rate at which legal flows are released.

Despite the fact that this scenario is somewhat artificial, or restrictive, with regard to the IFR requirements of the NWA, it does illustrate some important features of the impacts of potential impoundment on the Mkomazi streamflow regime. In particular, curtailment of releases for inter-basin transfer may be required at pre-determined levels in the winter low flow season. Lower releases for transfer should also be considered in periods *before* low flow periods and *after* water stress months to ensure greater semblance to the natural flow regime. Additionally, restrictions to river abstractions for irrigation in low flow months could be applied. A number of short releases from the Smithfield Dam made in March (when there is no hydrological alteration within the target range from pre-dam conditions) for off-channel irrigation storage downstream may compensate for abstraction losses in periods of low flows and increase the high pulses and durations to *within* the RVA target range. However, this would be effective downstream only as far as to the point of off-take for off-channel storage.

Conclusions

The major benefit of the RVA is that, in the absence of extensive biological data or ecological expertise, preliminary targets designed to protect natural aquatic biodiversity and aquatic ecosystems, can be set using either historical hydrological data or simulated hydrological information. The absence of adequate aquatic ecosystem and climatic data, as well as of observed runoff, is common for South African catchments and rivers. Consequently, there is great scope

Table 4: RVA score card: Mkomazi IFR site 2

	October 1934 - September 1944 (31 years)				October 1945 - September 1966 (31 years)				RVA TARGETS		HYDROLOGIC ALTERATION
			Range Limits				Range Limits				
Parameter	Medians	Coefficient of Variance	Low	High	Medians	Coefficient of Variance	Low	High	Low	High	
Group 1											
October	16.10	0.90	2.40	143.60	2.90	1.10	1.20	127.50	5.68	14.81	-0.77
November	21.70	0.70	5.40	87.90	5.70	2.02	1.30	84.00	13.24	28.44	-0.62
December	30.90	0.84	6.40	145.10	19.80	1.28	1.60	123.40	22.22	48.25	-0.42
January	52.80	0.62	8.00	161.20	42.00	1.06	1.60	147.40	31.97	75.56	-0.08
February	62.40	0.84	10.80	174.60	47.70	0.80	2.30	182.40	38.68	78.73	-0.15
March	65.60	0.65	20.80	218.50	48.40	0.78	3.20	263.80	41.28	80.20	0.00
April	31.00	0.52	7.00	100.50	27.60	0.72	2.00	96.10	26.60	43.68	-0.27
May	22.20	0.47	3.50	219.60	12.80	0.79	1.40	306.30	16.04	28.37	-0.69
June	15.20	0.47	2.60	33.70	8.10	0.88	1.10	24.00	11.45	18.56	-0.81
July	10.60	0.52	2.50	78.90	3.60	1.01	1.10	66.00	7.97	13.32	-0.85
August	8.40	0.62	2.30	45.00	3.10	1.42	1.10	25.40	6.14	12.08	-0.85
September	7.70	1.03	2.30	220.10	2.00	1.17	1.20	208.80	6.13	13.07	-0.85
Group 2											
1-day minimum	3.30	0.72	1.00	10.70	1.20	0.15	0.80	2.30	2.45	4.79	-1.00
3-day minimum	3.50	0.74	1.00	10.70	1.20	0.15	0.80	2.40	2.48	4.87	-1.00
7-day minimum	3.80	0.56	1.10	10.40	1.30	0.17	0.90	2.50	3.01	5.15	-1.00
30-day minimum	4.60	0.67	1.40	11.60	1.50	0.24	1.00	3.00	3.78	6.68	-1.00
90-day minimum	7.10	0.71	1.80	15.80	1.80	0.87	1.10	24.50	6.26	10.29	-0.88
1-day maximum	222.30	0.75	81.10	2982.70	201.80	0.75	15.30	3435.70	146.11	312.58	-0.08
3-day maximum	181.80	0.87	36.90	1787.30	184.90	0.73	11.80	1731.00	117.03	273.51	-0.04
7-day maximum	129.00	0.85	31.20	1128.30	113.60	0.76	7.80	1100.30	82.94	200.60	-0.15
30-day maximum	87.30	0.73	26.60	340.50	74.20	0.63	4.20	323.60	62.00	125.13	-0.15
90-day maximum	65.60	0.64	21.90	180.10	52.80	0.80	7.00	185.30	48.74	81.52	-0.19
Number of zero days	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Base flow	0.13	0.46	0.05	0.28	0.07	0.87	0.02	0.58	0.10	0.18	-0.65
Group 3											
Date of minimum	278.50	0.08	227.00	338.00	274.00	0.08	227.00	340.00	268.75	297.50	0.27
Date of maximum	28.50	0.21	2.00	366.00	36.00	0.18	8.00	366.00	26.00	272.75	0.11
Group 4											
Low Pulse Count	6.00	0.63	0.00	20.00	7.00	0.71	2.00	16.00	4.00	9.00	0.18
Low Pulse Duration	11.80	0.66	0.00	64.00	19.80	0.75	8.60	78.60	7.00	14.88	-0.56
High Pulse Count	8.00	0.63	2.00	19.00	9.00	0.66	0.00	21.00	6.00	11.00	-0.33
High Pulse Duration	6.60	1.10	2.00	63.50	5.70	1.21	0.00	54.00	4.96	10.33	-0.08
Group 5											
Rise rate	12.30	0.48	5.50	38.30	8.80	2.61	0.70	24.00	9.66	15.08	-0.85
Fall rate	-3.70	-0.45	-10.10	-1.60	-4.00	-0.65	-13.40	-6.40	-4.49	-2.83	-0.31
Number of reversals	104.50	0.14	84.00	132.00	169.50	0.12	136.00	207.00	96.00	114.00	-1.00

Comparison of Statistics Within, Above and Below RVA Range

Parameter	Within RVA Range Expected	Within RVA Range Observed	Hydrologic Alteration	Above RVA Range Expected	Above RVA Range Observed	Hydrologic Alteration	Below RVA Range Expected	Below RVA Range Observed	Hydrologic Alteration
Group 1									
October	26	6	-0.77	13	5	-0.62	12	40	2.33
November	26	10	-0.62	13	7	-0.46	12	34	1.83
December	26	15	-0.42	13	8	-0.38	12	28	1.33
January	26	24	-0.08	13	8	-0.21	12	18	0.50
February	26	22	-0.15	13	11	-0.15	12	18	0.50
March	26	26	0.00	13	6	-0.54	12	19	0.58
April	26	19	-0.27	13	8	-0.38	12	24	1.00
May	26	8	-0.69	13	8	-0.34	12	37	2.08
June	26	5	-0.81	13	5	-0.62	12	41	2.42
July	26	4	-0.85	13	3	-0.77	12	44	2.67
August	26	4	-0.85	13	2	-0.85	12	45	2.75
September	26	4	-0.85	13	3	-0.77	12	44	2.67
Group 2									
1-day minimum	26	0	-1.00	13	0	-1.00	12	51	3.25
3-day minimum	26	0	-1.00	13	0	-1.00	12	51	3.25
7-day minimum	26	0	-1.00	13	0	-1.00	12	51	3.25
30-day minimum	26	0	-1.00	13	0	-1.00	12	51	3.25
90-day minimum	26	3	-0.88	13	1	-0.92	12	47	2.92
1-day maximum	26	24	-0.08	13	9	-0.31	12	18	0.50
3-day maximum	26	25	-0.04	13	7	-0.46	12	19	0.58
7-day maximum	26	22	-0.15	13	10	-0.23	12	19	0.58
30-day maximum	26	22	-0.15	13	9	-0.31	12	20	0.67
90-day maximum	26	21	-0.19	13	6	-0.54	12	24	1.00
Number of zero days	51	51	0.00	0	0	0.00	0	0	0.00
Base flow	26	9	-0.65	13	3	-0.77	12	39	2.25
Group 3									
Date of minimum	26	33	0.27	13	9	-0.31	12	9	-0.25
Date of maximum*	27	30	0.11	13	10	-0.23	11	11	0.00
Group 4									
Low Pulse Count*	28	33	0.18	12	14	0.17	11	4	-0.64
Low Pulse Duration*	27	12	-0.56	13	36	1.82	11	1	-0.91
High Pulse Count*	34	22	-0.35	12	13	0.08	5	16	2.20
High Pulse Duration	26	24	-0.08	13	6	-0.54	12	21	0.75
Group 5									
Rise rate	26	4	-0.85	13	6	-0.54	12	41	2.42
Fall rate*	26	18	-0.31	13	15	0.15	12	18	0.50
Number of reversals*	29	0	-1.00	12	51	3.25	10	0	-1.00

*WARNING	Use caution in interpreting expected and observed compliance rates
Low Pulse Counts	9 yearly values are equal to the upper RVA limit
Low Pulse Counts	5 yearly values are equal to the lower RVA limit
High Pulse Counts	7 yearly values are equal to the upper RVA limit
High Pulse Counts	11 yearly values are equal to the lower RVA limit
Low Pulse Duration	2 yearly values are equal to the lower RVA limit
Number of Falls	8 yearly values are equal to the lower RVA limit
Number of Reversals	3 yearly values are equal to the upper RVA limit
Number of Reversals	3 yearly values are equal to the lower RVA limit
Date of Maximum	4 yearly values are equal to the lower RVA limit

for the application of the RVA in South African catchments where river management objectives for the ecological Reserve have yet to be ascertained. By setting preliminary flow management thresholds that can be modified and refined when ecological data and information become available, the approach incorporates flexibility and adaptability. These attributes could prove to be instrumental in resolving water resource management issues.

The approach embraces the theory that the full range of natural variation of a hydrological regime is required to sustain the full natural biodiversity and integrity of aquatic ecosystems. The RVA addresses this concept by focussing on ecologically relevant hydrological parameters that characterise natural streamflow regimes. However, the developers of the RVA acknowledge that the reliance of natural aquatic biota on the 25th to 75th percentile range of the hydrological parameters used in the analysis has not been widely tested for statistical soundness. Furthermore, any statistical analysis of the causal link between flow and the organisms dependent on it is inherently limited. This could well be construed as being a shortcoming of the approach. However, as a link between river flow and river condition, by virtue of identifying critical variations in the magnitude, timing, frequency, duration and rate of change of flows, it represents a feasible and practical methodology for the preliminary assessment of the ecological Reserve for the Mkomazi River.

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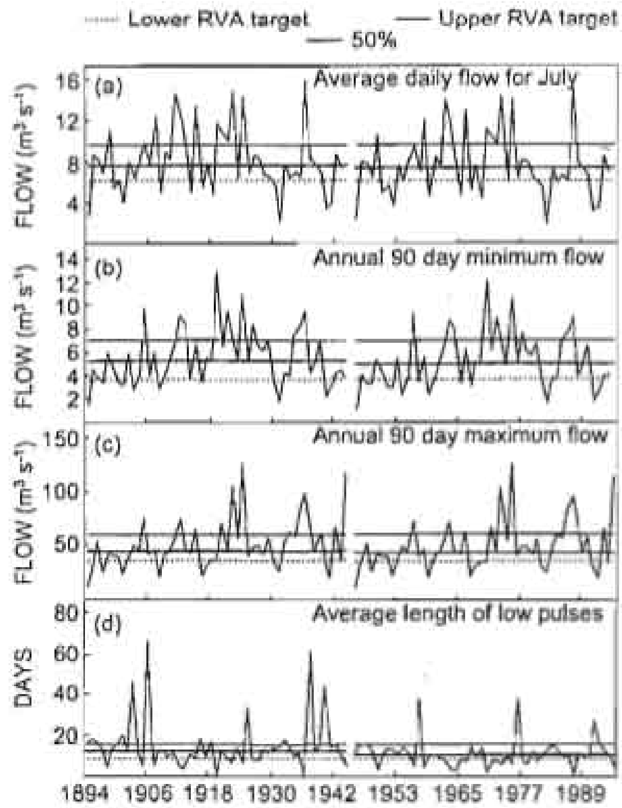

Figure 2: Examples of changes in the hydrological regime, with potential upper (75th percentile) and lower (25th percentile) RVA management targets, at IFR Site 1 on the Mkomazi River

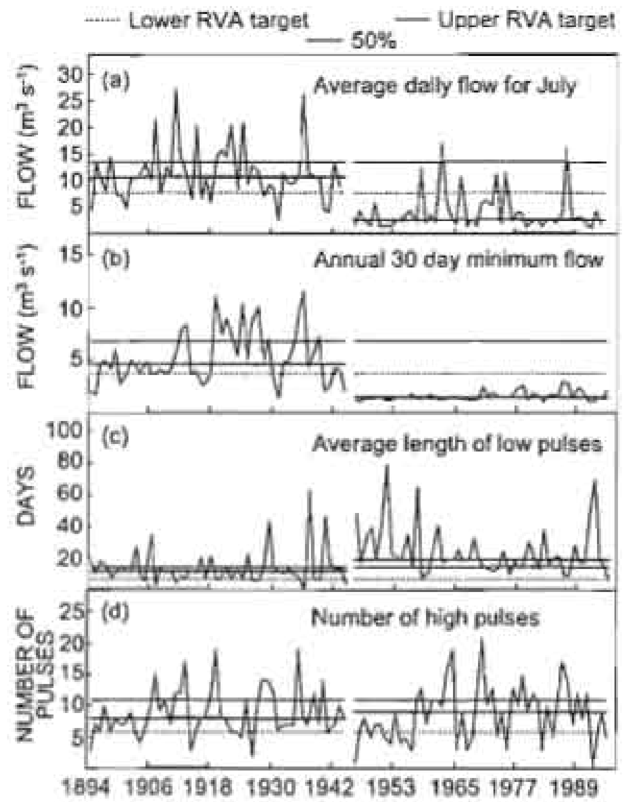

Figure 3: Examples of changes in the hydrological regime, with potential upper (75th percentile) and lower (25th percentile) RVA management targets, at IFR Site 2 on the Mkomazi River

Appendix 6B Hydrological indices used in the Study, their derivation and source of reference; symbol notation for indices as in Olden and Poff (2003)

Symbol	Unit	Definition	Reference
Magnitude of flow events			
Average flow conditions			
M_A1	$m^3.s^{-1}$	Mean monthly flow for October	Richter *et al.* (1996; 1997)
M_A2	$m^3.s^{-1}$	Mean monthly flow for November	
M_A3	$m^3.s^{-1}$	Mean monthly flow for December	
M_A4	$m^3.s^{-1}$	Mean monthly flow for January	
M_A5	$m^3.s^{-1}$	Mean monthly flow for February	
M_A6	$m^3.s^{-1}$	Mean monthly flow for March	
M_A7	$m^3.s^{-1}$	Mean monthly flow for April	
M_A8	$m^3.s^{-1}$	Mean monthly flow for May	
M_A9	$m^3.s^{-1}$	Mean monthly flow for June	
M_A10	$m^3.s^{-1}$	Mean monthly flow for July	
M_A11	$m^3.s^{-1}$	Mean monthly flow for August	
M_A12	$m^3.s^{-1}$	Mean monthly flow for September	
M_A13	-	Coefficient of Dispersion of M_A1, (*i.e.* difference between 75th percentile of values and 25th percentile of values divided by the median of values across all years of record)	
M_A14	-	Coefficient of Dispersion of M_A2	
M_A15	-	Coefficient of Dispersion of M_A3	
M_A16	-	Coefficient of Dispersion of M_A4	
M_A17	-	Coefficient of Dispersion of M_A5	
M_A18	-	Coefficient of Dispersion of M_A6	
M_A19	-	Coefficient of Dispersion of M_A7	
M_A20	-	Coefficient of Dispersion of M_A8	
M_A21	-	Coefficient of Dispersion of M_A9	
M_A22	-	Coefficient of Dispersion of M_A10	
M_A23	-	Coefficient of Dispersion of M_A11	
M_A24	-	Coefficient of Dispersion of M_A12	
M_A25	-	Ratio of seasonal variability to baseflow (CDB)	Hughes and Hannart (2003)
Low flow conditions			
M_L1	-	7-day annual minimum flow divided by mean daily flow for year	Richter *et al*, (1998)
M_L2	-	Coefficient of Dispersion in M_L1	
M_L3	-	Ratio of baseflow volume to total volume (Alt BFI)	Hughes and Hannart (2003)
M_L4	$m^3.s^{-1}$	Q75, the 25th percentile of flow values across the record	
High flow conditions			
M_H1	-	Median of annual maximum flows (HFI)	Olden and Poff (2003)
Frequency of flow events			
Low flow conditions			
F_L1	$year^{1-}$	Low pulse count (*i.e.* number of annual occurrences during which the magnitude of	Richter *et al.* (1996; 1997)

		flows is below a lower threshold. Low flow pulses are those periods within a year when flow is less than the 25th percentile of all daily values across the record)	
F_L2	-	Coefficient of Dispersion in F_L1	
High flow conditions			
F_H1	year^{-1}	High pulse count (*i.e.* number of annual occurrences during which the magnitude of flows is above a higher threshold. High flow pulses are those periods within a year when flow is greater than the 75th percentile of all daily values across the record)	Richter *et al.* (1996; 1997)
F_H2	-	Coefficient of Dispersion in F_H1	
Duration of flow events			
Low flow conditions			
D_L1	$m^3.s^{-1}$	Annual minimum 1-day average flow	Richter *et al.* (1996; 1997)
D_L2	$m^3.s^{-1}$	Annual minimum 3-day average flow	
D_L3	$m^3.s^{-1}$	Annual minimum 7-day average flow	
D_L4	$m^3.s^{-1}$	Annual minimum 30-day average flow	
D_L5	$m^3.s^{-1}$	Annual minimum 90-day average flow	
D_L6	Year^{-1}	Average annual number of days having zero daily flow	
D_L7	days	Average duration of F_L1	
D_L8	-	Coefficient of Dispersion in D_L1	
D_L9	-	Coefficient of Dispersion in D_L2	
D_L10	-	Coefficient of Dispersion in D_L3	
D_L11	-	Coefficient of Dispersion in D_L4	
D_L12	-	Coefficient of Dispersion in D_L5	
D_L13	-	Coefficient of Dispersion in D_L6	
D_L14	-	Coefficient of Dispersion in D_L7	
High flow conditions			
D_H1	$m^3.s^{-1}$	Annual maximum 1-day average flow	Richter *et al.* (1996; 1997)
D_H2	$m^3.s^{-1}$	Annual maximum 3-day average flow	
D_H3	$m^3.s^{-1}$	Annual maximum 7-day average flow	
D_H4	$m^3.s^{-1}$	Annual maximum 30-day average flow	
D_H5	$m^3.s^{-1}$	Annual maximum 90-day average flow	
D_H6	days	Average duration of F_H1	
D_H7	-	Coefficient of Dispersion in D_H1	
D_H8	-	Coefficient of Dispersion in D_H2	
D_H9	-	Coefficient of Dispersion in D_H3	
D_H10	-	Coefficient of Dispersion in D_H4	
D_H11	-	Coefficient of Dispersion in D_H5	
D_H12	-	Coefficient of Dispersion in D_H6	
D_H13	days	Average annual maximum number of days in a water year during which no floods occur across the period of record	Poff and Ward, 1989; Poff (1996)
Timing of flow events			
Average flow conditions			

T_A1	-	Predictability of flow, comprising two independent components, *viz*. constancy (i.e. a measure of temporal invariance) and contingency (i.e. a measure of periodicity)	Richter *et al.* (1996; 1997)
T_A2	-	Proportion of Predictability due to Constancy	
Low flow conditions			
T_L1	-	Average Julian date of the 1-day minimum flow over period of record	Richter *et al.* (1996; 1997)
T_L2	-	Coefficient of Dispersion in T_L1	
High flow conditions			
T_H1	-	Average Julian date of the 1-day maximum flow over period of record	Richter *et al.* (1996; 1997)
T_H2	-	Coefficient of Dispersion in T_H1	
T_H3	-	Maximum proportion of all floods over the period of record that fall within any 60-day period	Poff and Ward, 1989; Poff (1996)
Rate of change in flow events			
Average flow conditions			
R_A1	$m^3.s^{-1}.d^{-1}$	Average rate of positive changes in flow from one day to the next	Richter *et al.* (1996; 1997)
R_A2	$m^3.s^{-1}.d^{-1}$	Average rate of negative changes in flow from one day to the next	
R_A3	-	Average number of negative and positive changes in water conditions from one day to the next	
R_A4	-	Coefficient of Dispersion in R_A1	
R_A5	-	Coefficient of Dispersion in R_A2	
R_A6	-	Coefficient of Dispersion in R_A3	

Appendix 6C Month-by-month input variables for the land use categories used in the Mkomazi study

ACRU category	LANDSAT TM Classification	Variable	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Plantation: (Pines: intermediate age, pitted)	Forest Plantation	Water use coefficient	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85	0.85
		Interception loss	3.30	3.30	3.30	3.30	3.30	3.30	3.30	3.30	3.30	3.30	3.30	3.30
		Roots in topsoil	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60
		Coefficient of I_a	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35
Valley Bushveld	Thicket & Bushland	Water use coefficient	0.65	0.65	0.60	0.55	0.45	0.40	0.35	0.40	0.45	0.55	0.60	0.65
		Interception loss	1.60	1.60	1.60	1.40	1.20	1.10	1.10	1.10	1.20	1.45	1.55	1.60
		Roots in topsoil	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75
		Coefficient of I_a	0.20	0.20	0.25	0.30	0.30	0.30	0.30	0.30	0.30	0.25	0.20	0.20
Dryland	Cultivated: permanent - commercial sugarcane	Water use coefficient	0.79	0.80	0.88	0.95	0.98	0.94	0.89	0.85	0.82	0.81	0.81	0.81
		Interception loss	1.80	1.80	1.80	1.80	1.80	1.80	1.80	1.80	1.80	1.80	1.80	1.80
		Roots in topsoil	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60	0.60
		Coefficient of I_a	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.35	0.25
Dryland	Cultivated: temporary - commercial dryland	Water use coefficient	0.89	1.10	0.96	0.46	0.20	0.20	0.20	0.20	0.20	0.20	0.35	0.60
		Interception loss	1.00	1.50	1.40	1.30	1.20	0.50	0.50	0.50	0.50	0.50	0.50	0.50
		Roots in topsoil	0.79	0.74	0.78	0.91	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.92
		Coefficient of I_a	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.35	0.25
Dryland	Cultivated: temporary - semi-commercial / subsistence dryland	Water use coefficient	0.80	0.70	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.35	0.60
		Interception loss	1.00	1.00	0.60	0.50	0.50	0.50	0.50	0.50	0.50	0.50	0.50	0.80
		Roots in topsoil	0.74	0.78	0.91	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.92	0.79
		Coefficient of I_a	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.35	0.25
Urban	Degraded: Unimproved Grassland	Water use coefficient	0.55	0.55	0.55	0.45	0.20	0.20	0.20	0.20	0.30	0.40	0.50	0.55
		Interception loss	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80
		Roots in topsoil	0.90	0.90	0.90	0.94	0.94	0.94	0.80	0.80	0.80	0.80	0.80	0.80
		Coefficient of I_a	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.20	0.20
Valley Bushveld	Thicket & Bushland	Water use coefficient	0.65	0.65	0.60	0.55	0.45	0.40	0.35	0.40	0.45	0.55	0.60	0.65
		Interception loss	1.60	1.60	1.60	1.40	1.20	1.10	1.10	1.10	1.20	1.45	1.55	1.60
		Roots in topsoil	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75
		Coefficient of Ia	0.20	0.20	0.25	0.30	0.30	0.30	0.30	0.30	0.30	0.25	0.20	0.20
Grassland (Coastal forest and thornveld)	Unimproved grassland	Water use coefficient	0.85	0.85	0.85	0.85	0.75	0.65	0.60	0.65	0.75	0.85	0.85	0.85
		Interception loss	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00
		Roots in topsoil	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75	0.75
		Coefficient of Ia	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30
Grassland (Highland sourveld and dohne sourveld)	Unimproved grassland	Water use coefficient	0.60	0.60	0.60	0.45	0.20	0.20	0.20	0.20	0.30	0.50	0.60	0.60
		Interception loss	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
		Roots in topsoil	0.90	0.90	0.90	0.95	1.00	1.00	1.00	1.00	1.00	0.95	0.90	0.90
		Coefficient of Ia	0.15	0.15	0.15	0.20	0.25	0.25	0.25	0.25	0.25	0.20	0.20	0.15

Grassland (Ngongoni veld)	Unimproved grassland	Water use coefficient	0.65	0.65	0.65	0.55	0.50	0.30	0.30	0.30	0.45	0.55	0.60	0.65
		Interception loss	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20	1.20
		Roots in topsoil	0.90	0.90	0.90	0.94	0.97	1.00	1.00	1.00	0.97	0.94	0.90	0.90
		Coefficient of Ia	0.15	0.15	0.20	0.25	0.25	0.25	0.25	0.25	0.25	0.20	0.20	0.15
Grassland (Ngongoni veld of natal mist belt)	Unimproved grassland	Water use coefficient	0.63	0.63	0.63	0.50	0.35	0.25	0.25	0.25	0.40	0.53	0.63	0.63
		Interception loss	1.11	1.11	1.11	1.11	1.11	1.11	1.11	1.11	1.11	1.11	1.11	1.11
		Roots in topsoil	0.90	0.90	0.90	0.94	1.00	1.00	1.00	1.00	1.00	0.94	0.90	0.90
		Coefficient of Ia	0.15	0.15	0.15	0.20	0.25	0.25	0.25	0.25	0.25	0.20	0.2 0	0.15
Grassland (Southern tall grassland)	Unimproved grassland	Water use coefficient	0.55	0.55	0.50	0.45	0.40	0.30	0.30	0.30	0.35	0.45	0.50	0.55
		Interception loss	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00	2.00
		Roots in topsoil	0.80	0.80	0.80	0.80	0.90	0.95	0.95	0.95	0.95	0.80	0.80	0.80
		Coefficient of Ia	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.20	0.20
Grassland (Themeda-festuca alpine veld)	Unimproved grassland	Water use coefficient	0.65	0.55	0.50	0.40	0.30	0.20	0.20	0.20	0.20	0.30	0.50	0.55
		Interception loss	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80	0.80
		Roots in topsoil	0.80	0.80	0.80	1.00	1.00	1.00	1.00	1.00	1.00	0.97	0.94	0.90
		Coefficient of Ia	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.20	0.20
Channel (riparian)	None identified	Water use coefficient	0.78	0.78	0.78	0.72	0.58	0.53	0.55	0.55	0.61	0.71	0.73	0.75
		Interception loss	1.60	1.60	1.60	1.60	1.50	1.40	1.40	1.40	1.50	1.60	1.60	1.60
		Roots in topsoil	0.87	0.87	0.87	0.89	0.91	0.92	0.92	0.92	0.92	0.89	0.87	0.87
		Coefficient of Ia	0.23	0.23	0.23	0.23	0.30	0.30	0.30	0.30	0.30	0.30	0.23	0.23
Dams & Irrigation	Water Bodies	Water use coefficient	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70	0.70
		Interception loss	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
		Roots in topsoil	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
		Coefficient of Ia	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.20	0.20
Dams & Irrigation	Improved Grassland	Water use coefficient	0.80	0.80	0.80	0.70	0.60	0.5 0	0.50	0.50	0.60	0.70	0.80	0.80
		Interception loss	1.40	1.40	1.40	1.40	1.20	1.00	1.00	1.20	1.30	1.40	1.40	1.40
		Roots in topsoil	0.80	0.80	0.80	0.90	1.00	1.00	1.00	1.00	0.90	0.90	0.80	0.80
		Coefficient of Ia	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.30	0.20
Dams & Irrigation (winter cabbages & spring potatoes)	Cultivated: temporary - commercial irrigated	Water use coefficient	0.20	0.20	0.50	0.70	0.70	0.40	0.60	0.80	0.20	0.20	0.20	0.20
		Interception loss	1.00	1.00	1.00	1.00	1.00	1.50	1.50	1.50	1.00	1.00	1.00	1.00
		Roots in topsoil	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
		Coefficient of Ia	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.20	0.20
Dams & Irrigation (citrus)	Cultivated: temporary - commercial irrigated	Water use coefficient	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67
		Interception loss	1.70	1.70	1.70	1.70	1.70	1.70	1.70	1.70	1.70	1.70	1.70	1.70
		Roots in topsoil	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40
		Coefficient of Ia	0.20	0.20	0.20	0.20	0.30	0.30	0.30	0.30	0.30	0.30	0.20	0.20

CHAPTER 7 SYNTHESIS AND GENERAL CONCLUSIONS

CONTENTS

7 SYNTHESIS AND GENERAL CONCLUSIONS

1.1 Introduction

The concepts, information and, where appropriate, results pertaining to each of the preceding Chapters were discussed separately in the respective Chapter and, where appropriate, Appendices. The purpose of this Chapter is to:

(a) integrate the research findings;

(b) assess whether the overall aims of the thesis outlined in Chapter 1 have been met;

(c) present an overview of the implications of the research;

(d) identify the shortcomings as well as the benefits of the study, and

(e) recommend further research to facilitate the understanding of the hydrological basis for the protection of water resources to meet ecological and societal needs.

1.2 Integrating the Research Findings: Achievements of the Thesis

Each of Chapters 2 to 6 and Appendix 6A were preceded by a roadmap detailing the steps contained therein and products thereof. It is not the intention of this Section to discuss the details and findings of each Chapter, but rather to focus on those that pertain to the overall aims of this thesis. The overall aims of this thesis were described in Chapter 1, Section 1.6, together with the objectives formulated to meet those aims (*c.f.* Figure 1.3). Each of the objectives identified in Figure 1.3 is linked to the anticipated products of the thesis (also shown in Figure 1.3). This Section integrates the research findings and assesses the achievements of the thesis by examining whether the aims described in Section 1.6 were met. The main thrust of this thesis relates to the management of water resources so that society can continue to receive the ecological benefits provided by freshwater flows and functional aquatic ecosystems. This thread interlinks the Chapters comprising the thesis and, consequently, the aims, objectives and products are also cross-cutting. As a result not all of the objectives and associated products identified in Chapter 1 can be united discretely by Chapter. This factor reflects the complexity of the subject of the thesis.

Aim (a) in Chapter 1, Section 1.6, *"to identify the interrelationships between the hydrological cycle, the ecosystem goods and services it supports and the societal* *mechanisms that influence ecological integrity"* delivered a literature review (**Product 1** in Figure 1.3), in Chapter 2 which highlighted a renewed interest in the role of the hydrological cycle as a mediator of biodiversity and ecosystem services instream, at the riparian terrace, on the floodplain and the greater catchment region (**Objective 1** of Figure 1.3). As identified in Chapter 2, different streamflow components are associated with different ecological states and this theme is revisited in subsequent Chapters, most notably Chapters 4, 5, 6 and Appendix 6A. However, Chapter 2 initiated a major focus of the thesis, *viz.* that ecologically sustainable management of water resources is a key driver of environmental security and human health. The findings of Chapter 2 identified that the development of robust, holistic, ecologically sustainable water resource management was required to match the perspectives of different water users. Chapter 2 set the scene for defining the spatial, temporal and organisational scales which integrate ecological processes and societal mechanisms. More pertinently, two possible water resource management approaches for matching ecological systems with societal systems were highlighted, *viz.* the hydronomic zoning approach described by Molden *et al.* (2001) and the nested hierarchical approach described by Marchand *et al.* (2002).

These findings led directly to meeting **Aim (b)** in Section 1.7, *"to identify an approach to link the societal mechanisms that influence the ecological integrity of freshwater ecosystems at different spatial, temporal and organisational scales"*. One of the most reiterated perspectives in this thesis is that ecosystems and humans depend on the same water (Moberg and Galaz, 2005). Chapter 3 promoted the philosophy that humans and their societies are integral parts of ecosystems (**Objective 2** in Figure 1.3). There is a need for this understanding to be emphasised in Environmental Flow Assessments (EFAs). EFA is applied in many countries, seemingly for the purpose of defining ecological requirements for water. However, as highlighted in Chapters 3 and 4, EFAs are increasingly progressing towards scenario-based assessments of the socio-economic impacts of streamflow responses to different water-management activities on the biophysical functioning of the aquatic resource. Chapter 3 described the evolution of EFA, from its traditional approach of piece-meal determinations of the ecological streamflow requirements of individual ecosystem components to encompassing a more holistic and anthropogenic focus. However, even where this is applied (*i.e.* the "social importance" associated with the use of the river system is considered in comprehensive South African Reserve determinations), the societal needs for freshwater tend to be assessed in the same piece-meal manner as any "other" ecological component and with the focus on short-term, ecosystem goods. Piece-meal approaches to flow assessments are unlikely to result in integrated water resources management. Moreover, while the SA Reserve classification system that eventually determines the level of protection allocated to the water resource is intended to address both ecological and societal functioning, societal needs for freshwater at the catchment scale tend to be addressed in system yield analysis and scenario approaches, with the licensing of "spare" water. The goal of EFA should be to set "provisional" streamflow determinations which are as flexible as the ecosystems and societal systems being "managed". This type of adaptive management falls under the umbrella of "ecosystem management" rather than "streamflow regime management" and accounts for the societal value of ecosystem goods and services, while acknowledging the complexity and unpredictability of aquatic ecosystems. Moreover, adaptive management presents opportunities for strengthening resilience in both ecological and societal systems, thereby sustaining the long-term services that both ecological and societal systems need (**Product 2** in Figure 1.3).

In recognition of these factors, a framework for ecologically sustainable water resources management (**Objective 4** in Figure 1.3) was proposed in Chapter 4 (*c.f.* Figure 4.9). The proposed framework presented in Chapter 4 builds on the findings of Chapter 2, which identified a hydronomic zoning approach, based on managing societally influenced activities on the catchment, as being suitable for delineating water resources management units. Therefore, hydronomic zones (and sub-zones) represent an appropriate spatial and organisational scale for integrating hydrological landscape zones with societal activities. While the proposed framework is intended to be generic, it was presented in Chapter 4, since it complements the South African DWAF framework for water resources management shown in Figure 4.2. Moreover, the framework proposed in this thesis can be applied to link the South African Reserve (both the Ecological Reserve with the Basic Human Needs Reserve) with other water users (**Product 3** in Figure 1.3). Given the problems inherent with the lack of ecological data and of uncertainty of knowledge regarding ecological response to hydrological change, the benefit of the adaptive management approach presented in the framework for ecologically sustainable water resource management (Figure 4.9) is the formulation of stakeholder-based catchment management plans (Step 7) and hydronomic re-zoning and trade-offs (Step 8). Conversely, current methods for "managing flow regimes" (albeit for both water users and the

Reserves) focus on operating rules, assurance of supply and supply curtailment for systems with major dams (in accordance with the yield analysis conducted during the IFR design stage), or on the assessment of new abstraction or land use licences for catchments with no major dams and "determining whether these can be permitted without the Reserve being violated" (Hughes, 2005). Current methods propose that decisions regarding water use in systems with no water storage can be made by utilising flow duration curves to compare different flow regimes (natural, present and future) "since these are compatible with the assurance rule table methods used to define the Reserve requirements by the RDM Office in DWAF" (Hughes, 2005). Nonetheless, the non-availability of readily available information on the flow characteristics of sub-Quaternary Catchments has been highlighted as being problematic for some small-scale systems (Hughes, 2005). Moreover, there is concern that managing the Reserve by limiting other water allocation through the control of water use licences does not constitute an ecosystem approach (Rogers *et al.*, 2000). The emphasis in the study in Chapter 6 is that management of flow regimes on the tributaries of catchments cannot be achieved with rules for reservoir releases, or river abstractions and that other innovative methods are required to deal with the incompatibilities between social and ecological needs. Rather, the thesis proposes linking ecological and societal needs for freshwater in a catchment-based ecosystem goods and services approach which focuses on societal disruption of the hydrological cycle, compromise, resolution and adaptive management with hydronomic rezoning and trade-offs.

Matching the differences between the temporal scales of aquatic ecosystems and societal systems is understandably more challenging. Different indicators of ecological integrity were discussed in Chapter 2, where hydrological indicators were identified as being the most suitable for describing the temporal scale of both societal and ecological systems. Moreover, as the premise of this thesis is "the hydrological basis for the protection of water resources to meet environmental and societal requirements", hydrological indicators are the most logical indicators of choice. Indices representing the intra- and inter-annual hydrological variability of the streamflow regime were selected as indicators of the ecological integrity of freshwater ecosystems, since stakeholders generally have an understanding of the functions of different streamflow components.

Aim (c) in Section 1.6 *"to complement the Resource Directed Measures in the South African water resources management process"* (also **Objective 3** in Figure 1.3) was addressed directly in several different Chapters, *viz*, Chapters 4, 5 and 6. However, it is suggested that there is potential for the entire thesis to contribute to the Resource Directed Measures (RDMs) in the South African water resources management process. In particular, the proposed framework for ecologically sustainable water resources management presented in Chapter 4 could prove to be useful to the South Africa's water allocation reform process. The material presented in Chapter 2 highlighted the critical role of the hydrological cycle in supporting aquatic and terrestrial ecosystem goods and services, whereas Chapter 3 focussed on EFAs as a management tool for environmental security and societal well-being. However, a major emphasis of Chapters 4, 5 and 6 is the role of the hydrological cycle in providing connectivity among the major biomes and their different and diverse ecosystems. There is great scope for affirmation of the relevance of the hydrological component in the determination of the ecostatus of a river system in the South African RDMs. This is all the more relevant when considering the (often) lack of data or information relating to the other ecological components (*e.g.* riparian vegetation and aquatic invertebrates). Returning the focus of EFAs in South African water resources management to the hydrological cycle could simultaneously address the general misconception that the Ecological Reserve (ER) is for "nature", while reconnecting EFAs with the catchment beyond the river channel. Thus, hydrological connectivity is a major touchstone for holistic approaches to EFAs.

As mentioned above, hydrological indicators were highlighted in this study as being most appropriate for linking the temporal scale of both societal and ecological systems. The major portion of this thesis (*i.e.* Chapters 4, 5 and 6) also investigated the usefulness of hydrological indicators, or more pertinently hydrological indices, of streamflow characteristics for linking the spatial scales of both societal and ecological systems. Several hydrological indices are already used in the desktop or rapid South African Reserve determinations, based on research of extrapolations of more comprehensive Reserve methods (Hughes *et al.*, 1998). In addition, Joubert and Hurly (1994) characterised the different streamflow regimes found in South Africa based on differences among eight hydrological indicators on the understanding that the information relating to their study would be beneficial for ecological studies.

Nonetheless, the study in Chapter 5 of this thesis is considered to represent the most up-to-date and comprehensive study of the choice of hydrological indices for characterising the streamflow regimes found across South Africa (**Product 4** in Figure 1.3). The majority of the indices investigated fall within the suite of the Indicators of Hydrological Alteration developed by Richter *et al.* (1996). While some of these indices are uncommon in other studies (*i.e.* the number of hydrograph reversals, numbers of both low and high pulses based on specific percentiles of streamflow values across the entire record), the majority of the IHA indices are used routinely in eco-hydrological studies (*i.e.* mean monthly flow, 1-day maximum and minimum flow events). The IHA suite of indices comprises a fairly comprehensive representation of the magnitude, duration, timing, frequency and rate of change of streamflow conditions, all of which have been identified as being ecologically relevant (Heeg and Breen, 1994; Poff, 1996). However, the results in Chapter 5 indicate that some of the IHA indices have limited use for characterising South African river conditions, depending on the spatial scale investigated, which in turn influences the choice of high information indices. Conversely, the high information indices for characterising streamflow regimes in different hydro-geographical regions of South Africa could be prove useful in desktop or rapid South African Reserve determinations, since they explain a high proportion of the variation in a fairly comprehensive suite of indices. *This feature is a major finding of this thesis.*

Indices applied in the desktop or rapid South African Reserve determinations were also included in the studies comprising both Chapters 5 and 6. The results in Chapter 5 indicate that at a broad spatial scale, and even at a finer regional scale, the indices based on the Desktop Reserve Model baseflow index (M_L3) and index of overall variability (M_A25) are extremely useful for characterising the streamflow regimes found in South Africa. However, these indices were shown to be less appropriate in the study in Chapter 6 for distinguishing among the streamflow regimes at the catchment scale for the Mkomazi Catchment in KwaZulu-Natal. This factor may be unique to the Mkomazi Catchment, reflecting the relative homogeneity of these characteristics among the streamflow regimes of the 52 hydrological-landscape units. However, it may be a feature of catchments in the higher rainfall regions of South Africa, requiring further research to determine whether these indices are indeed useful for eco-hydrological studies at a local scale (*i.e.* smaller than Quaternary Catchment).

Another way in which the South African RDM process can benefit from the research in Chapter 5 relates to the study of the length of record required for reliable estimates of the high information hydrological indices of different streamflow components. Knowledge of the streamflow type, based on the results of the PCA analysis, either at the broad spatial scale of "All-Streams" or at the regional scale of the streamflow type, could be used to identify a "preferred" or optimal record length for analysis. This could be translated into an additional level of confidence in the usefulness of different high information hydrological indices since, in practice, the usefulness of any index may be constrained by the length of the available record.

The study of the "preferred" record length raises a dilemma which all ecologists and hydrologists face, *viz.*, that too little observed data in any of the spatial, temporal or organisational scales necessitates increased reliance on modelled information, whether the paucity relates to species counts or streamflow events. While a record of observed streamflow data is preferable, simulated records have a place, and are used routinely, in eco-hydrological studies, including the South African Reserve determinations. Reliable simulated streamflows which have been verified against a relatively long observed record are extremely useful as substitutes for observed data, since simulated streamflow records can be generated for longer time periods (provided that a suitably long and reliable rainfall record is available) and for sites where there is no observed record. The merits of simulated records of daily streamflows at the scale of the 52 hydrological landscape units of the Mkomazi Catchment, using the record generated with the *ACRU* agrohydrological model are described in more detail in the paragraphs relating the **Aim (e)** below. However, the relevance of simulated records to **Aim (c)** is that hydrological models can be used to generate records of sufficient length to derive the ecologically relevant hydrological indices that could be useful for complementing the South African RDMs.

Aim (d) in Section 1.6 *"to assess the ecological significance of variable streamflow regimes and the societal consequences of their management as a resource for ecosystem goods and services"* was the most cross-cutting among the aims of this thesis. Consequently it addresses **Objectives 1, 2, 3, 4 and 5** of Figure 1.3. Given that the linkages between streamflow and ecological response are still relatively poorly understood, assessing the ecological significance of variable streamflow regimes to society is a real challenge. Chapter 2 briefly highlighted the ecological relevance associated with both low flow and high flow events since they provide the variability of streamflow conditions which influences the provision of the ecosystem goods and services required by society.

The key message in Chapter 3 was that EFAs, including the natural patterns of streamflow variability, are in effect anthropogenic-centred, since society and ecological systems share aquatic ecosystems. Chapter 4 highlighted the necessity of incorporating natural variability in streamflow regimes for maintaining aquatic and landscape biodiversity.

While variable streamflow regimes present major difficulties for water resources managers and stakeholders alike, the importance of the natural range of streamflow variability for maintaining the integrity and dynamic potential of aquatic ecosystems is widely recognised (Poff, 1996; Clausen and Biggs, 1998; Baron *et al.*, 2002). A key message of this thesis is that societal well-being and biodiversity are interlinked (**Products 1, 2, 3, 4 and 5** in Figure 1.3). However, "ecologists still struggle to predict and quantify biotic response to altered streamflow regimes" (Bunn and Arthington, 2002) and the challenge of identifying which streamflow volumes and patterns are required to provide selected ecosystem goods and services that people and society need or desire is even more difficult to ascertain. This thesis argues that the best alternative to this dilemma is to focus on statistical measures of ecologically relevant hydrological indices (as discussed above) as surrogates for expert-designed streamflow regimes until improved information from monitoring programmes (see below) is available. Chapter 5 highlighted the benefits of selecting high information hydrological indices which characterise variable river systems and which could be combined with ecological knowledge, whereas in Chapter 6 the importance of assessing natural streamflow variability across the entire river system network was emphasised.

Finally, **Aim (e)**, *"to apply the approach identified in Aim (b) to typical socio-ecological systems and environmental concerns in water resources management policies of South Africa"* (also **Objective 5** in Figure 1.3) was addressed through the Case Study on the Mkomazi Catchment in Chapter 6 and in Appendix 6A (**Product 5** in Figure 1.3). The assignment for the Mkomazi Catchment was to investigate the potential of this catchment to maximise the generation of ecosystem goods and services for its stakeholders. The potential of the Mkomazi Catchment for water resource development is substantial, since the Mkomazi River is one of the few remaining unregulated rivers in South Africa. This situation alone is extremely relevant to South African water resources and any proposals to impound the river have been resisted by the DWAF in the past. Moreover, despite any proposals from the DWAF that the water development of the Mkomazi River should be reserved for the inter-basin transfer of its water to the neighbouring Mgeni Catchment

(DWAF, 2004), the recognised South African water resources principle is that the needs of the donor catchment should be met first (Gillham and Hayes, 2000). The Case Study in Appendix 6A was initiated before this thesis was formulated. Consequently it did not benefit from the PCA carried out in Chapter 6 to identify high information hydrological indices of the streamflow regimes of the Mkomazi Catchment. However, the study in Appendix 6A, conducted on the streamflow regime at the two uppermost IFR sites of the Mkomazi River, complements the Case Study of the tributaries of the Mkomazi Catchment in Chapter 6 as well as Aim (c).

As far as was considered practical, given the theoretical nature inherent in the study, the Case Study in Chapter 6 followed the steps outlined in the framework for ecologically sustainable water resources management identified in Figure 4.9 of Chapter 4. Given the benefits of the study conducted in Chapter 5, it would have been advantageous to have had (additionally) the opportunity to apply the framework to another catchment in a different hydro-geographical region. However, the Mkomazi Catchment does possess fairly typical socio-ecological systems and environmental concerns and the overall results of the study are not expected to be unique to the Mkomazi Catchment.

The Case Study in Chapter 6 indicated that it is essential that the tributaries of the nation's major river systems are given as much attention in Reserve determinations as the mainstream river. Societal systems tend to experience greater environmental austerity on the tributaries and the Mkomazi Catchment Case Study indicates that there are a variety of hydrological and socio-economic reasons for this. The Mkomazi Catchment Case Study highlighted the advantages of regarding precipitation as a better starting point for assessing the aquatic resource base than instream flows, to both ecological and societal systems. This philosophy concurs with the perspectives of several of the most respected international research groups in freshwater management (*e.g.* Stockholm International Water Institute, SIWI and the Department of Systems Ecology in Stockholm) in that they consider precipitation as the primary resource. The benefit of this approach to the Case Study was recognition of societal activities which disrupt hydrological connectivity and linkages of land use, freshwater flows and ecosystem goods and services.

Spatial mapping of the hydrological alteration of the hydronomic zones of the Mkomazi Catchment identified *where* change was taking place. However, the analysis of different water uses at the finer, sub-zone scale helped to identify *why* hydrological alteration was occurring. Despite any misgivings regarding the confidence attached to the time series of the *ACRU* simulated streamflows associated with low flow components, the study of the extent of hydrological alteration at a sub-Quaternary scale would not have been possible without hydrological modelling. To the best of this author's knowledge, the study in Chapter 6 represents a novel study of hydrological alteration at the organisational scale of hydronomic sub-zones, thereby facilitating direct comparisons of the impacts of a variety of different water uses on high information hydrological indices of streamflow variability. The study in Chapter 6 relating to how societal activities interact with, and disrupt, hydrological connectivity provides a compelling argument for intensive hydrological modelling at a relatively fine temporal and organisational scale to precede South African Reserve determination workshops.

The main critics of the RVA method applied in Chapter 6 cite the weakness of the links between statistical measures of the streamflow regime and ecological response, since freshwater systems do not respond in a linear manner to either natural disturbance or to anthropogenic alteration. However, this inherent complexity in freshwater systems also means that any linkages between streamflow and ecological response are still relatively poorly understood, particularly when compared to the linkages between streamflow and geomorphological response. This is the main reason for the focus of the SA Reserve determinations on maintaining the ecological / geomorphological habitat rather than meeting biotic freshwater requirements. However, the growing consensus in freshwater management is that dealing with complexity and uncertainty in both ecological and societal systems requires approaches which adopt adaptive management for ecologically sustainable water resources management (Rogers and Biggs, 1999; Moberg and Galaz, 2005). Monitoring ecological response to any prescribed modification of the streamflow regime is critical to the credibility of any determination of ecological freshwater needs. In the absence of any tested theories of the ecological response to statistical measures of the streamflow regime, the RVA method represents an holistic approach to setting *provisional* water management targets which can be adjusted following appropriate monitoring programmes (Richter *et al.*, 2003). Thus the application of the RVA to assess the degree of hydrological alteration in the Mkomazi Catchment is a suitable tool for adaptive water resources management.

1.3 Value of the Study: Benefits and Implications

From an international context, the main value of this thesis lies in the emphasis and application of the basic ecological principles of energy flows and nutrient cycling to describe hydrological connectivity among aquatic and terrestrial ecosystems. In addition, as the proposed framework for ecologically sustainable water resources management in Figure 4.9 incorporates the core ecosystem management principles of holistic environmental flow assessments, stakeholder participation and adaptive management, and is based on an ecosystem goods and services approach, it could be applied to any water resource development planning. However, its optimum use is anticipated to be in situations where there is potential, or perceived, conflict between ecological and societal systems, since it is based on the premise that societal well-being and biodiversity are interlinked.

The study relating to the choice of hydrological indices for characterising the streamflow regimes of South Africa also has implications for international research. While there may well be differences among the river size and patterns of streamflow of different hydrogeographic regions throughout the world, the environmental gradient, ranging from "harsh" to "moderate" is likely to be feature of any PCA of a comprehensive database of hydrological indices. This assumption is confirmed by the identification of this gradient for the streamflow regimes found across the Mkomazi Catchment which, by South African standards, is in a relatively high rainfall area. Moreover, the PCA conducted in Chapter 5 indicated similar results to the PCA conducted by Olden and Poff (2003) conducted for many more indices (171) of many more streamflow regimes (420) in the United States of America. While the statistically significant principal components identified by the PCA applied in the Olden and Poff Study explained greater percentages of the variation in the indices than was found to be the case for the indices applied to the PCA in Chapter 5, both studies identified two separate clusters of highly inter-related indices, one of central tendency and the other of dispersion.

The methods applied to the research comprising this thesis benefited from several international applications and concepts in water research. These applications include the following,

(a) the hydronomic approach developed by Molden *et al.* (2001);

(b) the proposed framework for ecologically sustainable water resources management by Richter *et al.* (2003);

(c) the relationship among precipitation, terrestrial ecosystems and aquatic ecosystems as envisaged by researchers at the Department of Systems Ecology in Stockholm, Sweden (Falkenmark, 2003);

(d) the IHA suite of hydrological indices (Richter *et al.*, 1996) for statistically analysing daily streamflow values for pre- and post water resource developments;

(e) the PCA and research conducted by Olden and Poff (2003);

(f) the RVA (Richter *et al.*, 1997; 1998) for setting provisional water resources management targets and spatially mapping any hydrological alteration;

(g) the study of record length required for New Zealand's rivers by Clausen and Biggs (2000), and

(h) various papers, articles and books all duly cited and referenced in the thesis.

However, in the spirit of scientific corroboration, it is anticipated that there is value to the water research community, internationally, in the relevance of these applications in a South African context. This is all the more pertinent since the international water research community has a genuine interest in the progress and success of the Water Allocation Reform programme of South African Water Law (DWAF, 2005).

From the perspective of South African water resources management there are several valuable components in this thesis in addition to those already mentioned above. However, the framework for ecologically sustainable water resources management presented in Figure 4.9 is highly relevant to contemporary water related issues. It is anticipated that the framework could provide a valuable touchstone for ecologically sustainable management of the nation's water resources, which is the main aim of the Water Allocation Reform programme (DWAF, 2005). The framework embraces the DWAF guidelines for water allocation and sets new benchmarks for *stakeholder participation*, linkages between *livelihoods use of water* and *productive use of water* and encourages a "*sense of catchment*" through the emphasis on *hydrological connectivity* among aquatic and terrestrial ecosystems. All of these attributes are pertinent to Guidelines 2, 5, 6 and 7 of Water Allocation Reform programme (DWAF, 2005) which refer respectively to *capacity development programmes, equitable as well as efficient* *water allocation, ecologically sustainable water resources management* and the *mechanisms to facilitate licensing evaluation procedures*.

Moreover, the Water Allocation Reform programme acknowledges that new approaches are required where "current licence applications may exceed the allocable water" or where "the water resource is already over-allocated" (DWAF, 2005). This situation is addressed in the proposed framework in Figure 4.9 where a hydrological zone can be described as "closed", yet stakeholders may still be in a position to re-allocate (or "trade") any water currently used for short-term benefits to meet their requirements for long-term security (*c.f.* Step 4 of Figure 4.9). While the framework in Figure 4.9 is of high value to South African water resources management, it is supported in this thesis by three equally valuable research components, *viz.*

(a) the choice of hydrological indices for characterising streamflow regimes across the country, at different spatial scales, using PCA and following the methods of Olden and Poff (2003),

(b) the application of a hydronomic zoning approach adapted from Molden *et al.* (2001), and

(c) the extent of hydrological alteration resulting from different water uses, applying the RVA (Richter *et al.*,1997; 1998)

The identification of minimum subsets of high information hydrological indices which account for the majority of the variation in the indices of different streamflow regime types is anticipated to be of value to eco-hydrological studies and to stakeholders in identifying which components of the streamflow regime they should focus on for the delivery of desirable ecosystem goods and services. However, it must be stressed that, wherever possible, the selection of hydrological indices should be used in conjunction with other ecological data and information and matched with the nature of the particular ecological question (Olden and Poff, 2003). Thus the findings of Chapter 5 serve as a major input to the framework for ecologically sustainable water resources management. Likewise, delineating reference hydrological zones into hydronomic sub-zones, based on societal activities, is an appropriate approach to ensure that hydrological processes and connectivity between upstream and downstream users are accounted for. Thereafter, assessing the extent of hydrological alteration from reference conditions, by applying the RVA approach, ensures that ecological and societal needs are considered in an optimal and sustainable way.

Although testing the framework presented in Figure 4.9 across different streamflow types in different hydro-geographical regions in South Africa was not within the scope of this Study, the variability among the reference hydrological zones in the Mkomazi Catchment (as identified by the PCA) is sufficiently diverse. The application of the framework to the Mkomazi Catchment in Chapter 6 represents a valuable exercise and springboard for the Water Allocation Reform programme, since it provides recommendations of best management practice, rezoning and trade-offs to achieve the most beneficial use of water resources in the interest of the catchment, its environment and its stakeholders.

A major advantage of the study in Chapter 5 was the use of the South African DWAF's streamflow records. Despite the inclusion of some inappropriate records in the study in Chapter 5, the merits of a national database of reliable streamflow records cannot be overstated. Not only does a national database inspire credibility in any analytical investigation, but it provides an unequivocal benchmark for researchers. However, the study in Chapter 5 called for streamflow records which were representative of "natural flow conditions" over a relatively long and common time period. This crucial requirement introduced real problems to the study, not least of which was the emergence of only 48 stations, out of an original 201, where the long-term records were representative of reasonably natural flow conditions and which could be analysed.

1.4 Shortcomings of the Study

Measuring a system's ecostatus requires control or reference conditions. Controls will always have some inherent undesirable feature. As stated above, a major shortcoming in the study comprising Chapter 5 was the erroneous assumption that the DWAF streamflow records applied in previous research were "recording reasonably natural flow". The misgivings regarding approximately 50% of the 83 DWAF streamflow records used in the study of hydrological indices for the river systems found in South Africa were discussed in Sections 6 and 7 of Chapter 5. On discovery of the anomaly, it was deliberated at length whether it may be more useful to initiate a nationwide study for South Africa, at the scale of the 1946 Quaternary Catchments, of simulated daily streamflows using the *ACRU* agrohydrological model. The decision was made that such a study could be undertaken as further research (see below). Moreover, it was decided that there was merit in proceeding with the analysis of the reduced set of 48 DWAF stations, since "good" or "better" records of observed streamflows are inherently more credible among scientific researchers than records of simulated streamflows which have not been, or cannot be, verified meticulously at either the necessary time steps or spatial resolution.

The parallel study of the choice of indices for characterising the diversity of the streamflow regimes found throughout the Mkomazi Catchment did call for simulated daily streamflows using the *ACRU* agrohydrological model. As discussed in Chapter 6, confidence in the full suite of the 35 indices of intra-annual variability was relatively low. While there was increased confidence in the selected high information hydrological indices, the performance of the *ACRU* model, particularly in simulating the low flow components of the streamflow regime, raises some valid concerns.

Nonetheless, confidence in the reference hydrological conditions of the river systems which have been assessed by comprehensive Reserve determinations often ranges from poor to zero. For example, the ER determinations for the Thukela Catchment in KwaZulu-Natal, resulted in only three of twenty-one resource units attracting "average confidence", whilst the remainder attracted lower confidence ratings (IWR Environmental, 2003). In addition, the other ecological components of water quality, geomorphology, riparian vegetation, fish and aquatic invertebrates also failed to attract very much higher confidence across the same sites (IWR Environmental, 2003). This factor emphasises a major analytical difficulty in any system study (either ecological or societal), *viz.*, that the resolution of data available seldom meets the requirements for analysis, either temporally or spatially. Fortunately, since fundamental hydrological processes are relatively well understood, it is more acceptable to simulate hydrological information than other ecological information to sites where no data or information exists.

1.5 Recommendations for Further Research

Towards the completion stage of this Study in 2005, the developers of the IHA method for statistically analysing records of average daily streamflows released a beta version of a very much more detailed model. While the original 66 indices are still incorporated in the software, 34 "new" indices, comprising five "environmental flow components" describing low flows, extreme low flows, high flow pulses, small floods and large floods are included in the up-dated version. The five environmental flow components are described in the IHA User's Guide by The Nature Conservancy (The Nature Conservancy, 2005) and need not be repeated here. In addition, the calculation of some of the indices in the updated version of the model differs from that in the version used in the studies included in this thesis. It would be an interesting and relevant future research project to redo this Study to test the sensitivity of the indices of high information (which account for the majority of streamflow variability) identified in Chapters 5 and 6 with the availability of the additional indices from The Nature Conservancy (2005).

The studies in this thesis focused on median values of streamflow parameters as statistical measures of ecologically relevant hydrological indices since it could be expected that under "normal" climatic conditions a compromise between ecological and societal systems could be reached in one out of two years. However, the dispersion around the median of the indices was also applied in the study in Chapter 5 to account for the variability required when streamflows are either restricted or more abundant. In addition, the study in Chapter 6 analysed the distribution of the occurrence of annual values of the hydrological parameters in three different target ranges (*i.e.* occurrence below the 25th percentile of values, occurrence between the 25th and 75th percentile of values and occurrence above the 75th percentile of values). Nonetheless, different statistical measures could be investigated in future studies. Moreover, as emphasised in Chapters 4 and 6, it is important that where there is a paucity of ecological information any preliminary management targets, based on statistical measures of ecologically relevant hydrological indices, are re-assessed and adjusted when monitoring results and ecological observations become available (Richter *et al.*, 2003).

One way in which the findings of this thesis could additionally complement the determination of the RDM in South African water resources management would be to apply the methods used in the study in Chapter 5 to the 1946 Quaternary Catchments of South Africa. Hydrological indices of high information could be identified using PCA analysis at a broad spatial scale and at a finer hydro-geographical regional scale using indices extracted from long-term "reference hydrological" records. This could be achieved using a hydrological model, which can be configured to simulate time series of daily streamflows resulting from the hydrological pathways of precipitation, interception, evapotranspiration infiltration, stormflow, and baseflow to runoff. Reference hydrological records could be simulated in the same way as they were for the Mkomazi Catchment Case Study in Chapter 6 of this thesis, *viz.* using the hydrological attributes of Acocks' veld types (Acocks, 1988) to represent baseline land cover conditions as input to the *ACRU* hydrological model. Where appropriate, it would be beneficial to conduct verification studies of the time series representing each of the hydrological indices derived from the *ACRU* simulated streamflows at the sites comprising the "best48" DWAF stations identified in Section 6 of Chapter 5, before embarking on the PCA analysis. In addition, the "updated" suite of IHA indices could be analysed for the extended study. However, PCA of even the 74 indices used in the study in Chapter 5 (*i.e.* a 74 by 74 matrix) would be a large undertaking for 1946 sites, and while the results would be beneficial to ecohydrological studies, the task would be extremely demanding in research time.

As mentioned above it would be advantageous to apply the framework for ecologically sustainable water resources management in Figure 4.9 and the methods applied in the Case Study in Chapter 6 to another catchment in a different hydro-geographical region. The procedure could identify very different results when applied to a catchment in a relatively arid region.

Lastly, there is a need for the terminology of the South African Reserve to be revisited. While it can be expected that the language and terminology of the Reserve should be evolving as quickly as the methods for the determination thereof, there is a need for consistency among both the terms and the researchers and practioners who use them. Many of the terms applied currently in the literature are used interchangeably and often erroneously (*e.g.* Resource Directed Measures and the Reserve are not the same yet they are often referred to as such; DWAF, 2001). This factor can serve only to further cloud the perceptions of stakeholders. In a similar vein, the discussion in Chapter 4 argued against the choice of the term "reserve" based on the concept of "set-aside" for a finite natural resource. Even the DWAF literature describing the concept of the Reserve uses an analogy of a bucket containing the nation's water whereby the Reserve is "the water that must always be left in the bucket for basic human and ecological needs" (DWAF, 2003). Despite the premise that the Reserve classification system determines the level of protection and, therefore, the ERC that is used, this type of analogy clouds the very important line of reasoning that "*humans and ecosystems share the same water*" (Moberg and Galaz, 2005). In addition, there needs to be greater recognition of the "Reserve" as a variable entity which depends on environmental, societal and economic conditions, just as any other finite natural resource. Of course, the difference between water resources and any other finite natural resource is that there is no alternative to this "bloodstream of the biosphere" (Moberg and Galaz, 2005) for humans and the environment that supports our existence.

1.6 General Conclusions

This thesis has argued that the hydrological cycle is a "circulation" system which unites all the major biomes and sustains both ecological and societal well-being. The overall aim of the research contained in this thesis was to increase current understanding of the linkages between the hydrological cycle, the diversity of the hydrological regime, ecosystem response to changes thereof and the generation of ecosystem goods and services that people rely on for basic survival, quality of life and which allow societies to prosper. Freshwater flows provide the linkage between ecological and societal systems, provided their physical state is not compromised to the extent that their functions are diminished.

Natural ecosystems are dynamic, complex and adaptive, providing the flows of energy and nutrients essential to life and are necessary for species survival. In general, societal systems are less complex than ecological systems in terms of their water requirements. Given that freshwater systems are more complex than societal systems, spatially, temporally and organisationally, it makes sense to adopt water management strategies which focus on societal adaptability to environmental change.

Worldwide, people make water-related choices which impact on future generations. Frequently, scientific journals publish articles forecasting global threats to natural capital (*i.e.* natural resources) associated with human population growth and activity. Yet among scientists there is no concurrence whether the "headcount" (human population growth) or the "ecological footprint" (human activity) presents the dominant threat to biodiversity (McKee *et al.*, 2003; Lui *et al.*, 2003). What is clear is that environmental change, and the impacts of those changes on aquatic ecosystems, compromises ecological functioning and the delivery of ecosystem goods and services at the spatial scale where there is most local societal reliance. Safeguarding the hydrological processes which generate ecosystem goods and services that are highly prized by people presents a clear and judicious approach for the management of natural resources. Ecosystem management strategies which adopt a stewardship approach have been evolving for centuries wherever society and the natural environment meet. However, while theories relating to the interactions within socio-economic ecological systems may vary, conservation efforts to sustain freshwater flows and aquatic biodiversity will be futile if population growth is ignored (McKee, 2003).

Even in the face of "ecological disasters", it is not practical for scientists to call for the suspension of societal utilisation of natural resources. Ideally, understanding society's relationship with natural systems should embrace an ecosystem approach with the integration of multidisciplinary knowledge, monitoring and (re)-assessment. Ecologists need to be more certain and concise in evaluating the water patterns required for ecological functioning so that ecosystem response to altered flow regimes can be anticipated. On the other hand, water users need to be aware of the impacts that specific activities have on ecological functioning, now and in the future.

A major component of this Study was the review of the South African water resources management framework. Environmental flows have a high profile in South African water resources management as a result of water law legislation which promotes sound environmental governance. As a consequence, EFAs, in the form of Reserve Determinations, are highly developed in South Africa, arguably more so than in many developed nations where the conflicts between ecological and social needs for freshwater are less acute. The framework for ecologically sustainable water resources management proposed in Figure 4.9 was described as an aid to clarifying certain relationships and processes relating to the South African water resources management process. Notwithstanding these guidelines, the framework has value for ecologically sustainable water resources management in the global arena.

While studies to estimate the worth (in monetary terms) of natural capital are widely acknowledged to be substantial underestimates of the value of the goods and services provided by society's basic life support ecosystems, they do emphasise the need for the "economic value" of current and potential ecosystem goods and services to be a major component in water resource planning. Yet, attaching a monetary value to intangible services such as nutrient cycling, is superficial for many researchers for a number of reasons, not least of which is that for most ecosystem services, only a small portion of natural functioning can be replaced with artificial substitutes (Daily, 1999). Economic valuations for specific ecosystems and the impact of different management options on *water yield alone* are rare in South Africa, but are essential to future planning of catchment resources. Future evaluations should account for the nation's "hidden" natural resources, particularly the regulation of soil formation and nutrient recycling. The role of the hydrological cycle in supporting these processes cannot be overstated.

The main criticism of the RVA is that dependence of aquatic biota and ecosystem functioning on the indices needs to be substantiated (Tharme, 2003). Nonetheless, the RVA represents an approach to set provisional targets (which can be assessed and reset through monitoring of the resource) using either historical hydrological data or simulated hydrological information in the absence of extensive biological data or ecological expertise. Various statistical measures of the RVA indices, assigned to represent different ecological reserve categories, could be applied to the Threshold Model developed by Godfrey and Todd (2000) as a foundation for identifying the links between the ecological reserve categories and the provision of ecosystem goods and services.

Maintaining options for future use of natural resources is purported to be the greatest reward for good water resource management. However, this author believes that the overarching goal of ecologically sustainable water resource management should be the sparing use of the resource. Measures to keep options open need to be implemented, rather than relying on stream rehabilitation projects which are poor surrogates for ensuring aquatic ecosystem health. It is clear that it is difficult to restore even a small amount of ecosystem functioning to aquatic ecosystems that have been degraded (Rutherfurd, 2003). Even for wetland ecosystems, for which researchers have collected considerable knowledge, there is just too little known of the processes that shape ecosystem composition, functioning and structure to restore ecosystem integrity. Recognition of freshwater flows and aquatic systems as ecological resources as well as water resources at the outset of catchment management planning could prevent costly or unsuccessful restoration projects.

The future of water management must be to find ways to improve the management of available water. The hydrological processes and connectivity that maintain both aquatic and terrestrial ecosystems goods and services are critical to ecologically sustainable water resources management since they are the link between ecological well-being, ecosystem goods and services and resource utilisation. Ecologically sustainable water resources management should emphasise the need for people, society and governments to recognise the environment, freshwater flows and aquatic ecosystems as much more than natural resources for human use. Applying hydrological records as an ecological resource is highly appropriate for assessing the variability that ecosystems need to maintain the biodiversity, ecological functioning and resilience that people and society desire.

2 References

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8 SUMMARY

Each Section of each Chapter in this Thesis provides a summary of the material or results presented and discussed in the main body of the thesis (*i.e.* Parts I and II). The purpose of this final summary is to present the principal findings of each of those Chapters as a series of key points.

KEY POINTS

Chapter 2 The hydrological cycle and the generation of ecosystem goods and services

- There is a need for the real costs of water utilisation to be ascertained so that stakeholders can make informed decisions about the "vision" or "desired future state" of their water resource.
- There is renewed interest in the role that the hydrological cycle provides in generating biodiversity and ecosystem services instream, at the riparian terrace, on the floodplain and the greater catchment region.
- The concept of rivers as spatial, temporal and organisational structures transporting the ecosystem processes required to maintain ecosystems is useful to the understanding of the relationship between streamflow and ecosystem goods and services.

Chapter 3 The value of aquatic ecosystems in sustainable water resources management

- The emphasis on environmental flow requirements for "nature" is misplaced, since the focus of environmental flow assessment (EFA) has always been anthropogenic-centred.

- Environmental flow assessment has evolved from the determination of species specific streamflows to having a major role in holistic water resources management.
- Empirical assessments of the ecosystems goods and services provided by aquatic resources undervalue the true, or full, value of ecological functioning to society.
- Philosophical assessments of the value of aquatic ecological functioning are equally tenuous, particularly in water stressed areas where headcounts vie with the ecological footprint as the main contenders for environmental degradation.
- Ecological resilience and sustainability are best protected by conservation measures; however, strategies of ecosystem management which incorporate proactive data collection and management, adaptive management and participatory stakeholder approaches are the next best (and practical) option.

Chapter 4 The role of the Reserve in South African water resources management

- South African water law promotes an enabling approach to the management and utilisation of the water resource, with goals of sustainability and equity to meet societal and ecological needs.
- The Ecological Reserve (ER) is intended to protect the ecological functioning of water resources so that societal use of water and water resource developments are carried out in a sustainable manner.
- The Basic Human Needs Reserve (BHNR) is intended to protect basic human functioning, but is anticipated to achieve a more flexible status so that water to sustain livelihoods is also recognised as a basic human right.
- Some of the greatest challenges to implementing the Reserve lie in understanding the relationships among the ER, the BHNR and other water allocations.
- A framework for ecologically sustainable water resources management is proposed, advocating the delineation of catchment landscape zones which account for both reference hydrological conditions (hydrological zones) and societal activities and management options (hydronomic zones and sub-zones).
- The zoning concept presented in the framework considers the disruption of the hydrological cycle through changes to key hydrological processes as a result of societal activities both instream and off-stream.

Chapter 5 Hydrological indices of ecological water requirements of highly variable rivers

- Long-term records of observed streamflow data are an ecological resource for the assessment of environmental flow requirements.
- The hydrological classification of the ecological characteristics of different flow regimes can be described by indices derived from the statistical analysis of daily means of streamflows.
- Hydrological indices relating to the magnitude (how much), duration (for how long), frequency (how often), timing (when) and rate of change (how quickly) of river conditions are meaningful to the diversity of stakeholders within catchments.
- Together, the Indicators of Hydrological Alteration suite of ecologically relevant hydrological indices and the Desktop Reserve Model indices based on comprehensive ER determinations provide sufficient information on the streamflow variability of South African rivers to distinguish reasonably distinct streamflow types.
- Principal Component Analysis, based on the correlation matrix, provides this information.
- Identifying high information hydrological indices of temporal and spatial variation among different streamflow types can assist stakeholders to determine which components of the streamflow regime to preserve for the generation of the ecosystem goods and services that they need.
- The length of record required for stable estimates of high information hydrological indices varies across different streamflow components and across different streamflow types.

Chapter 6 Maximising the generation of ecosystem goods and services for future options of the Mkomazi Catchment, in KwaZulu-Natal, South Africa

- The Mkomazi River in KwaZulu-Natal is presently unregulated and thus provides a useful case study for the assessment of the impacts of potential catchment development on the sustainability of water resources for its stakeholders.
- Despite perennial flow, there are potentially conflicting water issues within the Mkomazi Catchment.
- Despite the usefulness of the streamflow record at the DWAF gauging station U1H005, hydrological modelling is required to generate reference hydrological conditions at different points of interest for the entire Mkomazi Catchment.
- Five different reference hydrological zone types, based on hydro-geographical attributes, can be identified for the Mkomazi River and its tributaries.
- PCA provides this information and highlights minimum sub-sets of high information, non-redundant indices for each zone type.
- These indices are highly relevant for use in Environmental Flow Assessment to assess any incompatibilities between ecological water needs and societal water needs which may arise as a result of water developments.
- A catchment-wide analysis of the impacts of water development at relatively fine time steps and organisational resolution indicates that existing alteration of the high information indices of variability for the different reference hydrological zone types varies greatly over the Mkomazi Catchment and its river network.
- The catchment to river network system is largely overlooked in environmental flow assessments.
- Hydronomic sub-zoning, based on the way in which societal activities disrupt the natural hydrological processes, both off-stream and instream, can be applied at a relatively fine organisational scale to assess the incompatibilities between societal and ecological freshwater needs.
- The Range of Variability Approach method of assigning provisional river management targets embraces adaptive management and participatory approaches which are essential to ecosystem management.

Appendix 6A Application of the Indicators of Hydrological Alteration method to the Mkomazi River, KwaZulu-Natal, South Africa

- Streamflow reduction activities, and in particular inter-basin transfers, are known for their extensive impacts on the streamflow resources of donor catchments.
- Modelling the pre- and post-impact behaviour of different streamflow components provides water resource managers with a representation of different utilisation states on the resource.
- Methods which apply "desktop" methods (*i.e.* easily computed hydrological indices derived from statistical analysis of daily means of streamflows) are valuable for scenario planning and for monitoring the implementation of EFAs.
- There is value in the RVA method of assigning provisional water resource management targets for South African catchments where the Resource Quality Objectives for the Ecological Reserve have yet to be established.