ERRATUM

Original article: Day JA, Malan HL, Malijani E and Abegunde AP (2019) Water quality in non-perennial rivers. *Water SA* 45 (3) 487–500. https://doi.org/10.17159/wsa/2019.v45.i3.6746

How to cite: Erratum: Water quality in non-perennial rivers. [*Water SA* 45 (3) 487–500]. *Water SA* 46 (2) 330–344. https://doi.org/10.17159/wsa.2020.v46i2.8250

Text was erroneously inserted on Page 495 (also see strikethroughs in text of original article that follows)

Page 495, Column 2. Replace Paragraph 2 with:

Froebrich (2005) points out that it is difficult to predict WQ in temporary rivers because of the spatial heterogeneity in catchment processes, which leads to variability in contaminant build-up during the dry period, coupled with variability in the timing and extent of rainfall events. By implication, water chemistry needs to be assessed for each river and plans for managing it need to be based on the empirical data generated. Because of the overall lack of 'dilution capacity' in non-perennial systems an obvious way to protect them is to limit input of both point- and non-point sources of pollutants. An established method for reducing the input of sediments, nutrients and other contaminants into surface waters, especially N-PRs (Rosado et al. 2012) is through the preservation of well-developed buffer strips of natural vegetation (e.g. Macfarlane et al., 2014; Lee and Fisher et al., 2016). It is during floods that large loads of sediment and contaminants enter the rivers (Froebrich, 2005) and it is also under these conditions that riparian vegetation is uprooted and flushed downstream. In very arid areas, though, there may well be no natural riparian vegetation, or the vegetation may consist of well-established trees within the water course, neither of which protects the river from lateral movement of sediments and other pollutants.

Water quality in non-perennial rivers

JA Day[1]*, HL Malan[1], E Malijani[1] and AP Abegunde[1]

Institute for Water Studies, University of the Western Cape, Robert Sobukwe Avenue, 7530 Bellville, South Africa

ABSTRACT

More than half the river-lengths of rivers in southern Africa dry up occasionally or – more commonly – seasonally. Here we review the literature on water quality (WQ) in non-perennial rivers (N-PRs), with emphasis on river management and southern African systems. Hydrological regimes cover a spectrum from relatively predictable and unvarying in perennial rivers, to unpredictable and highly variable in non-perennial rivers, which are complex, continually shifting mosaics of flowing water, standing-water pools and terrestrial habitats. N-PRs are uncommonly difficult to manage because they represent a limited source of water that is renewed unpredictably and is competed for by local people as well as being required by wildlife. Groundwater, and therefore its chemical and physical features, contributes significantly to base flow and to the maintenance of pools remaining in the bed when the river is not flowing. Water chemistry reflects catchment geology except in polluted systems. Salinity varies temporally, and spatially over three dimensions, and is the variable controlling the composition of the biotas of many N-PRs. Hydrological regimes are seldom predictable with any certainty; WQ varies naturally over time and space; groundwater often determines the WQ of surface water, especially in pools; and WQ in non-perennial rivers and pools may be affected by activities far upstream in the catchment. As yet we have no more than a sketchy understanding of the extent to which data on any one system can be applied to any other. Until we have a better understanding of these systems, the following basic principles should guide the management of WQ in N-PRs: (i) Rivers need to be assessed on a case-by-case basis. (ii) Understanding of the groundwater regime, including its chemistry, is crucial. (iii) Effluents need to be controlled by conservative effluent standards set for both ground and surface waters. (iv) Flows may need to be augmented at certain times of the year.

Keywords: non-perennial, rivers, water quality, review, water management, southern Africa

INTRODUCTION

Many rivers cease to flow from time to time and some even dry up completely for a few days, a few months, or a few years at a stretch. All of these can be said to be non-perennial rivers (N-PRs), in that they do not behave like 'normal' rivers that flow all the time. Various terms are used to describe the degree of non-perenniality of rivers. Datry et al. (2014a), for instance, classify as **intermittent** all rivers in which flow is temporary, ephemeral, seasonal or episodic. These terms are – confusingly – used differently by different authors. In this paper we use the terminology of Seaman et al. (2010), who developed a conceptual framework for non-perenniality (Fig. 1). In Fig. 1 dashed lines indicate that the boundaries are not fixed: for instance, some apparently perennial rivers may cease to flow in extreme drought conditions (Rossouw et al., 2005). A proportion of N-PRs are seasonal, ceasing to flow and even drying up for some months every year; these are 'ephemeral' or 'semi-permanent'. (Note that the term 'ephemeral' is used – more correctly – in studies of temporary wetlands to refer to those in which water occurs only fleetingly after rain – see, for instance, Keddy, 2010.)

Beyond cessation of flow, a river becomes a landscape of isolated longitudinal pools (Fazi et al., 2013), a phenomenon that has profound consequences for its biota and also for humans who use its waters. Uys and O'Keeffe (1997) present a conceptual framework suggesting ways in which predictability/variability of flow, connectivity of aquatic habitats, natural disturbance, and biotic/abiotic controls change with

hydrological state. Both biotic and abiotic forces influence the species present and thus affect community structuring. As will be seen later in this review, the hydrological state, flow variability, aquatic connectivity and, to some extent, vegetation and faunal community structure, can all influence the concentration of chemical constituents and the values of physical variables in both the water column and the groundwater. These factors therefore need to be kept in mind when interpreting water quality (WQ).

Non-perennial rivers are complex, continually shifting mosaics of flowing-water (lotic), standing-water (lentic) and terrestrial habitats (Datry et al., 2014a). Such systems tend to be ecologically fragile (Rossouw et al., 2005) and yet they are often exceptionally important to people living in the vicinity. Particularly in arid areas, they support vegetation and associated animals and represent scarce resources in a dry terrestrial landscape (Jacobson, 1997). According to Rossouw et al. (2005), nearly half of the lengths of South African rivers are non-perennial and globally such rivers are far more common (Datry et al., 2014a) than is generally recognised. Furthermore, the number and geographical extent of N-PRs is increasing world-wide as a result of climate change and increased water abstraction for human use (Von Schiller et al., 2011; Jaeger et al., 2014; Kibria, 2016).

Historically, ecologists have tended to avoid working in N-PRs (Larned et al., 2010; Datry et al., 2014a) at least partly because their variable hydrology makes them difficult to study. Rossouw et al. (2005), Seaman et al. (2010) and Seaman et al. (2016 a; b) were pioneers in the field in South Africa, providing useful insights into the hydrology, geohydrology, geomorphology, water quality, ecology, and people–ecosystem interactions of N-PRs. More recently, international attention

has increasingly been focused on these systems, as evidenced by the European MIRAGE (Mediterranean Intermittent Rivers Management: Prat et al., 2014; www.mirage-project.eu) and IRBAS (Intermittent River Biodiversity Analysis and Synthesis: www.irbas.fr) projects. The current paper is part of a multi-disciplinary project aimed at improving our understanding of the relationships between river flow, groundwater, and ecosystem characteristics, in order to inform decision-making related to the ecological and social consequences of modifying the hydrology of N-PRs. Here we review water quality in these non-perennial systems, discussing the major drivers influencing the concentrations of chemical constituents and the magnitudes of physical variables in these systems, as well as the ways in which they differ from perennial systems, and in which key WQ variables interact within them. Finally, some principles for managing WQ in N-PRs are presented. In this review, we make use of the literature pertaining to WQ in wetlands as well as N-PRs, since during periods of low flow N-PRs are effectively

linear systems of wetlands with varying degrees of hydrological connectivity. More comprehensive reviews of WQ in aquatic ecosystems from a South African perspective are to be found in DWAF (1996); Davies and Day (1998); Malan and Day (2002; 2012) and Day and Dallas (2011).

Effects of interactions between water quality variables on living organisms

Individual WQ variables can influence each other, acting synergistically (e.g. the effect of two or more chemicals together having a greater effect than either alone), or antagonistically (two or more agents in combination having a lesser effect than each alone). The major interactions that can occur between WQ parameters, and the resultant effects on a benthic community, are summarized in Fig. 2. These interactions are key to understanding WQ in aquatic ecosystems and are discussed individually below.
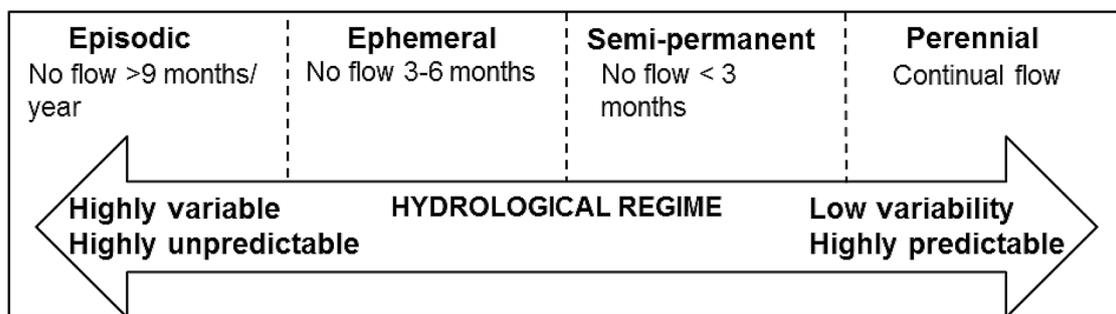

Figure 1. Conceptual framework of the hydrological continuum from perennial to non-perennial rivers (Seaman et al., 2010)
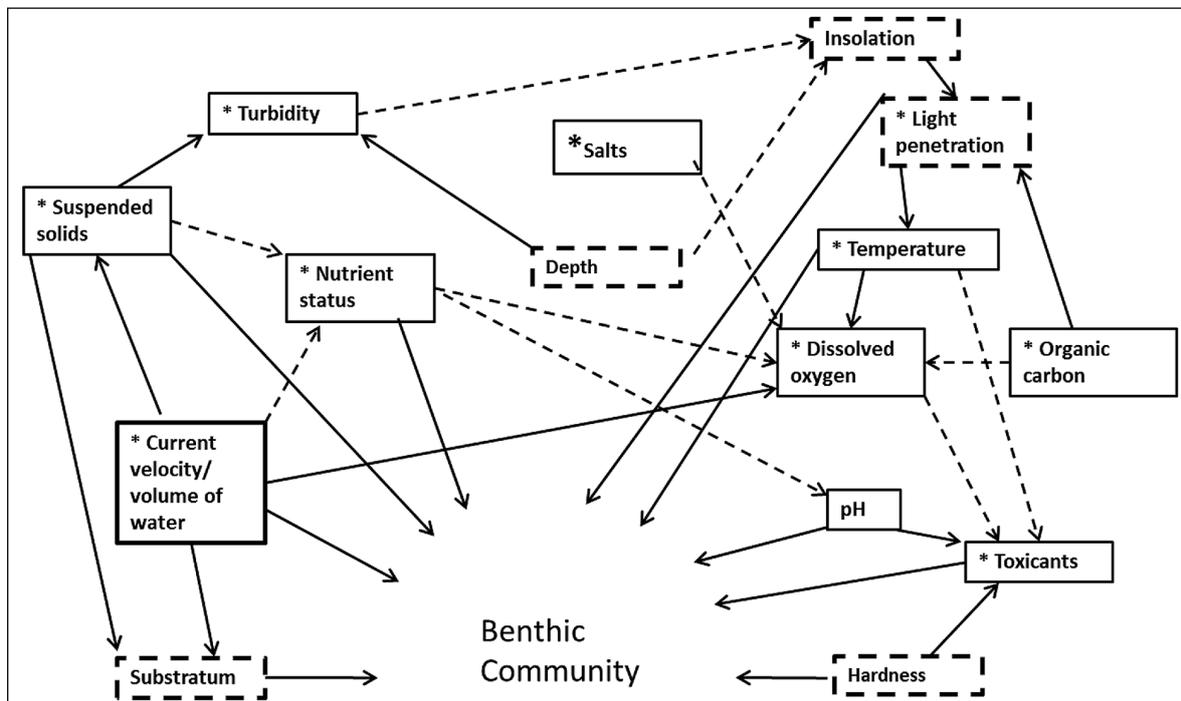

Figure 2. Interactions between water quality variables and other drivers of ecosystem functioning in rivers and wetlands. Water quality variables discussed in this review are enclosed in solid text boxes; those of peripheral importance are in dotted boxes. Direct effects are indicated by solid arrows and indirect effects by stippled arrows. Asterisks indicate aspects of particular relevance to N-PRs. Adapted from Hawkes (1979) and Dallas and Day (2004).

Surface and groundwater in non-perennial rivers

In N-PRs the single most important factor influencing water column concentrations of chemical constituents, and the magnitudes of physical variables, is the amount of water in the system (Rossouw et al., 2005). Larned et al. (2010) note that as pools become disconnected because of decreasing discharge they undergo abrupt thermal and chemical changes. The amount of water present determines the hydrological state of the river – whether it is in a state of flood, a small trickle, a longitudinal system of isolated pools, or dry river bed (with or without) hyporheic (under-bed) flow (Datry et al., 2016). The amount of water is likely to influence, amongst other factors, the concentrations of solutes (salts, toxins), nutrients (Von Schiller et al., 2011) and water temperature. As flow reduces, decreased turbulence will affect the entrainment of air within the water, and thus the concentration of DO (dissolved oxygen), and the entrainment of sediments from the bottom, and thus sediment loads – and turbidity – in the water (Mosley, 2015).

Although WQ in N-PRs is affected by the quantity and degree of movement of river water, the source of that water is also significant. Numerous studies (e.g. Rossouw et al., 2005; Seaman et al., 2010; Woelfle-Erskine et al., 2017) have emphasised the great importance of groundwater in non-perennial rivers, often contributing significantly to base flow (Rossouw et al., 2005). (It should be noted that, conversely, dryland rivers may also recharge aquifers: Rau et al., 2017). Where a connection between groundwater and surface water exists in the dry season, long-lasting pools may develop as a result of groundwater discharge (Seaman et al., 2013); shallow groundwater, rather than surface flow, is in fact the main source of 'new' water during the dry season in this type of system.

Understanding the link between ground and surface water is crucial for assessing the movement of water, and WQ, in N-PRs (Rau et al., 2017). Practically, it is fortunate that ground and surface waters in the same river often have different physical and chemical properties (e.g. Woelfle-Erskine et al. (2017), allowing natural components of WQ to be used for tracing the links between them. Various approaches, including isotopes and geochemistry, have been used for identifying and quantifying ground water–surface water interactions (Deodhar et al., 2014). The most commonly used isotopes in hydrology are Oxygen-18 (18O) and deuterium (2H), their isotopic compositions and ratios in water varying due to fractionation of isotopes during evaporation and condensation (Deodhar et al., 2014). Rau et al. (2017) have recently developed an approach that uses temperatures at different depths to establish relationships between ground- and surface water, based on the thermal diffusivity of sediments, which causes temperatures to vary with depth. From groundwater and surface water levels and the thermal signatures obtained, the different hydrological components can be identified.

Sub-surface flow in the hyporheic zone of many gravel- and sand-bed N-PRs represents a substantial proportion of the total discharge in the river, and exchange of nutrients and contaminants between hyporheic and surface water significantly affect biogeochemical processes occurring in these systems (Larsen et al., 2014). Not only may groundwater differ chemically and physically from surface water, but the variations in geohydrology and geochemistry over even a relatively small area can be complex. For example, Woelfle-Erskine et al. (2017) investigated two streams in California for several years over a period of increasing drought. Using electrical conductivity (EC) and tracers to estimate the source of base flow, these authors identified several aquifers in the study region, and showed that base flow originated from heterogeneous sources.

One of the most significant principles regarding ecosystem functioning in N-PRs is the critical importance of pools (Bernardo and Alves, 1999; Rossouw et al., 2005; Seaman et al., 2010). From observations on the Seekoei River in the Orange River catchment in South Africa (Seaman et al., 2010), the amount of water in pools during periods of no-flow seemed to depend on the balance between groundwater discharge from springs and evaporation from the surface of the pool; evaporation would vary with climatic conditions and time of year. Various conceptual flow models (e.g. Watson and Burnet, 1993; Ward and Robinson, 2000) are available in the literature to describe groundwater/surface water interactions, including a useful model developed for the Mokolo River (Seaman et al. 2013), which describes hydrological processes involved specifically in N-PRs.

In relation to the above discussion, Seaman et al. (2010) found that not only the location, timing and persistence of pools along the length of a river were highly unpredictable, but that the WQ in such pools was also variable. Even in the same geomorphic reach, pools could vary in conductivity, for example, probably related to differences in the sources and quantities of groundwater entering the system. Such variability makes it difficult to predict the nature of water chemistry along the length of the river. Choy et al. (2002), who investigated four endorheic rivers in the central arid zone of Australia, also found that water chemistry varied both spatially and temporally in complex ways.

Pools represent refugia for aquatic organisms during periods when there is no flowing water. Parsons and Wentzel (2007) cite the findings of Brown et al. (2003) on the Doring River, an N-PR in the south-western Cape, South Africa. Brown and her colleagues showed that groundwater played a critical role in maintaining pools that provided refugia for animals, including fish, during periods of no flow. Interestingly, it was noted that indigenous fish used only those pools fed by fresh groundwater sources, while alien fish were found in all pools.

A consequence of the variability of WQ in N-PRs is that the organisms living in them are exposed to significant stresses, primarily desiccation and chemical variations in terms of salinity, ionic proportions, and concentrations of nutrients (Rossouw and Vos, 2008).

WATER QUALITY IN N-PRS DURING THE DRYING PHASE

In a recent key paper, Mosley (2015) summarizes the literature on drought and its impact on WQ separately for rivers and streams, and for lakes and reservoirs (Fig. 3). This paper has some useful observations for inferring WQ in N-PRs during drying or no-flow periods, although the results need to be interpreted with care because the findings for rivers and streams do not necessarily pertain to the non-flowing state. In addition, lakes and reservoirs are usually much bigger and deeper than pools, so different processes might be at play; arid areas seldom have permanent freshwater lakes of any size so there are few opportunities to compare permanent lakes and pools in N-PRs. Nevertheless, there is some useful information in Mosley's review and Fig. 3 is used as a template as we discuss predicted changes in WQ in N-PRs during the drying-down phase of the hydrological cycle. Note that the literature is sparse, and only the few papers and reports already mentioned deal with South African conditions.
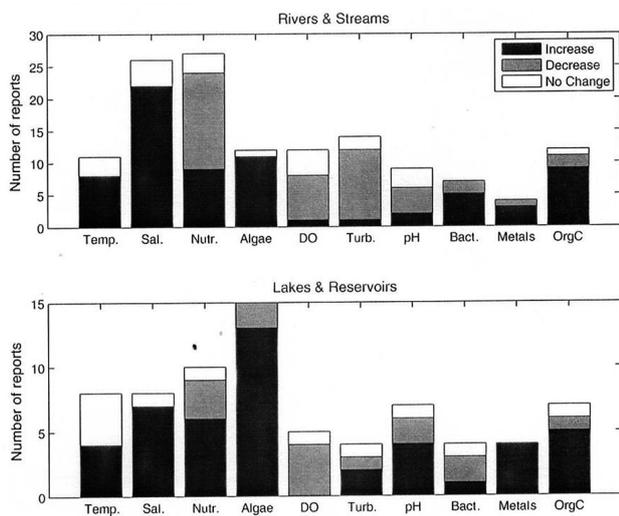
Figure 3. Summary of the effect (increase, decrease, no change) of drought on water quality variables for temperature (Temp.), salinity (Sal.), nutrients (Nutr.), algae, dissolved oxygen (DO), turbidity (Turb.), pH, bacteria (Bact.), metals and organic carbon (OrgC) in various kinds of aquatic ecosystem. Note that for most variables an increase in concentration is usually considered to be detrimental, except for pH and DO, where a decrease is usually considered detrimental. (From Mosley, 2015).

Temperature

General comments

Natural fluctuations in water temperature occur daily and seasonally, and indigenous organisms are adapted to such variations (Dallas, 2008). Organisms living in areas within their thermal ranges may not be affected by small changes in temperature, but the additional stress of even small increases in temperature may be critical when organisms are near the limit of their tolerance range (Sweeting, 1994). Furthermore, an increase in temperature enhances the toxic effects of some pollutants, including metals. A very important co-parameter is oxygen: oxygen solubility is depressed as temperature increases, leading to oxygen stress in susceptible organisms. Temperature has also been shown to affect the transport of fine sediments (e.g. Leliavsky, 1955). Below impoundments, for example, loads of suspended sediments may vary seasonally as a result of the higher viscosity of cold than of warm water (Leliavsky, 1955). Removal of riparian vegetation, and associated shading, may also result in temperature changes and, of course, in increased light penetration (Dallas, 2008).

Temperature in N-PRs

Water temperature generally increases in rivers when they are not flowing, especially if accompanied by an increase in air temperature (Lillebø et al., 2007). Larned et al. (2010) also note that as pools become disconnected by decreasing flow, diurnal temperature fluctuations increase in amplitude due to the decrease in thermal mass. According to Mosley (2015), thermal stratification in lakes may become more pronounced under conditions of drought; thermal stratification in deep pools of rivers is also likely to occur (Malan and Day, 2002) and may result in decreased DO levels with concomitant changes in the chemical environment, as discussed further on.

Salinity, electrical conductivity and total dissolved solids

General comments

The concentration of total dissolved solids (TDS) is the mass of all compounds dissolved in a sample of water, whereas electrical conductivity (EC) is a measure of the ionised substances within the water and represents a subset (usually a very large proportion) of the dissolved components. Salinity, a phenomenon used most commonly by marine chemists, also refers to the quantity of dissolved material, and is usually measured as EC. Note that it is often accepted practice to use the term 'salinity' for inland waters with high salt concentrations. An important point to note is that N-PRs are usually found in arid areas where ambient air temperatures are relatively high. As a consequence, evaporation rates are also elevated, increasing the salinity of standing water and damp soils (Rossouw et al., 2005; Gunkel et al., 2015).

Salinity in N-PRs

In isolated pools, salt concentrations are likely to increase through the process of evapo-concentration of solutes during the drying process (Rossouw et al., 2005; Woelfle-Erskine et al., 2017). A review of the literature by Mosley (2015) showed an increase in salinity to be the most likely outcome of drought in both rivers and lakes. Such increases are usually due to evaporation, but sometimes also to reduced dilution of saline groundwater, and in some cases to the inflow of effluents. In cases where the groundwater is not saline, or is fresher than surface water, salinities tend not to increase. Salinity was found to be particularly pronounced in highly seasonal, shallow systems near the point of complete dry-down.

In many arid environments salinity is elevated as a result of irrigation, clear-felling of trees, sewage return-flows, and river regulation (Walker, 2006). In this way naturally high salinities are exacerbated by human activities, sometimes leading to a loss of terrestrial and aquatic biodiversity, and of productive agricultural land. Bailey et al. (2006) contend that salt is the most under-recognised pollutant in the arid and semi-arid regions of the world, largely because salinization is seen as a problem only for agriculture. These authors provide a detailed review of 'salinization as an ecological perturbation to rivers, streams and wetlands of arid and semi-arid regions'.

Nutrients

General comments

Nutrients (often called 'limiting nutrients') are those elements required for plant growth; those of particular importance in aquatic ecosystems are nitrogen (N) and phosphorus (P). Eutrophication, or excessive accumulation of nutrients in a water body, is a serious problem world-wide, because it can result in rampant plant and algal growth (Ridge et al., 1995; Human et al., 2018). Many studies (e.g. Bouwman et al., 2017) have shown that the ratio of inorganic nitrogen to inorganic phosphorus is important for the normal functioning of aquatic ecosystems. Organic matter in unimpacted streams typically has an inorganic N:P ratio ranging between 25:1 and 40:1, whilst most impacted streams have ratios of <10:1.

Nutrients in N-PRS

Of all the variables considered in Fig. 3, nutrients respond with greatest diversity to drought in both lotic and lentic ecosystems. According to Mosley (2015), citing a range of authors, **lowered** concentrations of dissolved and total nutrients (N and P) during the drying phase of the hydrological cycle have been attributed to the following:

- Reduced input from the catchment because of reduced surface run-off
- Increased uptake by algae and macrophytes
- Increased denitrification due to longer residence times of the water
  On the other hand, if point-sources or polluted groundwater enter the system during dry conditions, concentrations of nutrients tend to **increase** due to reduced dilution capacity. Boulton et al. (2006) note that food webs in floodplains of larger desert rivers may be driven by production of zooplankton, which in turn is initiated by pulsed nutrients and carbon associated with inundation of the floodplain. The fate of nutrients in N-PRs can therefore be of major significance in determining the ecological functioning of the entire ecosystem.

Stratification in water bodies can lead to alterations in internal processing of nutrients. Mosley (2015) cites studies reporting **increases** in ammonia, organic nitrogen and total phosphorus in the hypolimnion (bottom water) of stratified systems exhibiting low DO concentrations, a consequence of desorption of nutrients from the sediments. Reduction in stream flow that allows sediments to settle may also change the nutrient status of the ecosystem and hence the composition of the biota, since nutrients can desorb into the water column (see below). Some papers, on the other hand, report the turn-over of nutrient-enriched hypolimnetic water and resuspension of sediments as lakes become shallow and are acted upon by wind, with subsequent enhancement of algal growth. Pools in non-perennial rivers probably behave in the same way.

With regard to nutrient cycling, very little information is available on the fate of nutrients entering N-PRs. Seaman et al. (2010) investigated particle-bound nutrients in pools in the Seekoei River, South Africa, and found no significant trends in concentrations, although some seasonal patterns were identified. Concentrations of dissolved organic nitrogen increased during the dry winter months, for instance, while dissolved organic phosphorus decreased, presumably due to deposition in the sediments. McLaughlin (2008) found in Sycamore Creek, Arizona, USA, that dissolved nutrients became evaporites when the river was dry, coating sandbars and sometimes being blown away by the wind, along with fine silt and organic particles.

Lillebø et al. (2007), studying nutrient dynamics in the seasonal Mediterranean Fuirosos stream in Spain, found that the hydrological sequence of drying followed by re-wetting was the most important factor controlling nutrient levels in the water. It was during the period of dis-connectivity, when the water column was shallow and temperatures high, that nutrient interactions took place at the sediment/water interface. Von Schiller et al. (2011) investigated nutrient availability in a non-perennial Mediterranean stream, finding that nitrate concentrations ranged from 0.01 to 2.0 mg N/L, peaking in winter (when the river would be flowing). On the other hand, ammonium and soluble reactive phosphorus exhibited concentrations ranging from 0.005 to 0.15 mg N/L and 0 to 0.02 mg P/L, respectively, peaking in summer (during no-flow) due to anoxic conditions. Dissolved organic nitrogen showed

no clear seasonal pattern. These same authors developed a conceptual model for the complex pattern of changes in dissolved nitrogen (N) and phosphorus (P) availability. During the drying part of the hydrological cycle, when the system consists of isolated pools, nutrient availability and speciation are dominated by processes occurring within the pools. Because of the chemically reducing conditions, ammonium concentrations are higher than the more oxidised forms of nitrogen and some nitrogen is lost due to denitrification. Organic forms of nitrogen are released due to leaching from organic material and tend to be higher in concentration than the inorganic forms. Phosphorus is released from sediments due to the anoxic conditions, but concentrations of inorganic phosphorus are higher than those of organic phosphorus. Overall, temporal changes in concentration are more pronounced for N than for P. The authors emphasise the high spatial variability in nutrient concentrations whilst the stream was in this hydrological phase.

The conceptual model of Von Schiller et al. (2011) indicates the complex, often competing, processes that are likely to control instream concentrations of the various species of N and P. The extent to which it can be extrapolated to other non-perennial rivers is unclear. It is likely that similar processes occur in other systems but that the relative magnitudes and rates may differ. For example, in the above model, groundwater sources of nutrients are reported only during the re-wetting phase, whereas in many other non-perennial systems, groundwater sustains pools during the dry period. The extent to which a river is forested (shaded), land use in the catchment, and pollutant loads could all be expected to affect nutrient dynamics in freshwater systems and serve to emphasise the site-specific nature of water quality in N-PRs.

Algae and blue-greens

General comments

Still-water conditions and high concentrations of nutrients promote the growth of algae, which occur in fresh waters in two forms: as microscopic members of the phytoplankton, and as long filaments. Elevated concentrations of nutrients in fresh waters frequently result in eutrophication, which manifests as excessive blooms of phytoplankton, causing the water to become green, and mats of filamentous algae that can cover the surface of a pool, affecting the light regime. Certain cyanobacterial forms are toxic.

Algae and blue-greens in N-PRs

Pools in N-PRs often support a large biomass of algae, particularly filamentous forms, which represent a food source and refuge for insects and fish (Rossouw et al., 2005), although they also obscure the surface and sometimes prevent free diffusion of oxygen into the water. In addition, moderate temperatures, low turbidity and, frequently, increased nutrient concentrations, all combine to promote the growth of algae in the water column (phytoplankton) and on the bottom (benthic algae) (Nouri et al., 2009). In some cases there may be shifts from relatively benign algae to toxic cyanobacterial species during drought (Mosley, 2015). An extreme example of this phenomenon was a bloom of the toxic blue-green *Anabaena circinalis,* which extended down almost a thousand kilometres of the Barwon-Darling River in Australia. According to Donnelly et al. (1997) the bloom was the result

of low river flow during hot conditions, accompanied by stratification, clarification due to settling and flocculation of sediments, and phosphorus release from anoxic sulphate-reducing sediments. Several features of harmful algal species (e.g. *Anabaena* spp.), including tolerance of high temperatures, salt tolerance and the ability to fix nitrogen, make them particularly adapted to non-flowing freshwater bodies during drought (Paerl and Paul, 2012).

Dissolved oxygen (DO)

General comments

DO concentrations vary diurnally due to the competing processes of respiration and photosynthesis, as well as due to the effect of temperature on gas solubility. As a result of the uptake of oxygen by living organisms during respiration, DO levels in natural water bodies are at a minimum at dawn. During daylight, plants photosynthesize, releasing oxygen and resulting in maximal concentrations at mid-afternoon in eutrophic systems (Dallas et al., 1998). Sudden decreases in the DO content of a water body ('anoxic events') may have a severe impact on living organisms, especially animals, leading, for example, to fish-kills (e.g. La and Cooke, 2011). As noted previously, the solubility of oxygen in water is inversely proportional to salinity and temperature (Malan and Day, 2002).

Other factors having a pronounced effect on DO levels include the presence of excess oxidizable organic matter. The source may be anthropogenic (e.g. a carbon-rich effluent such as sewage) or natural (e.g. decaying organic matter) and can lead to oxygen depletion due to blooms of heterotrophic, oxygen-consuming bacteria (Gromiec et al., 1983). Metals (e.g. iron and manganese), as well as sulphides, can appear in solution under conditions of oxygen depletion (in addition to nutrients as noted previously), a situation that may be exacerbated at high temperatures (Ure and Davidson, 1995). According to Mitsch and Gosselink (2007), transformations of N, S, Fe, Mn, C and P occur in anoxic wetland sediments as a result of chemically reducing conditions and some of those transformations, for example that of S to H_2S, can result in the formation of toxins. Other important transformations include the processes of denitrification and methanogenesis, which release N_2 and methane to the atmosphere. Many of these processes are catalysed by anaerobic microbes.

Dissolved oxygen in N-PRS

The extent to which redox-driven chemical transformations will occur in a N-PR is likely to depend on the length of time pools have been hydrologically isolated (Mosley, 2015). The diffusion rate of oxygen between air and water has a marked effect on the oxygen content of water and this in turn is strongly influenced by turbulence, which is a function of current speed and stream bed characteristics (Campbell, 1982), and wind. Slowly flowing or stagnant waters, on the other hand, are prone to anoxia. Dissolved oxygen is usually not limiting to aquatic organisms in flowing systems, but concentrations tend to drop as flow ceases. As noted previously, the decomposition of organic matter (e.g. leaves, organic pollutants) also contributes to low DO levels (Chapman and Kramer, 1991). In the case of pools formed during the drying-out period of N-PRs, the extent to which pools might become anoxic is likely to depend on site-specific features such as wind, water depth, the extent of water movement (e.g. from springs, or in the hyporheos),

the length of time that water persists in the pool, the amount of carbonaceous material in the sediments, and the type and biomass of living organisms present – and thus the rate of respiration versus photosynthesis. If algae and macrophytes are abundant, diurnal DO (and pH) may increase during the day (Williams, 2006), whereas in pools with little primary production, DO may steadily decrease due to respiration and decreased solubility.

Turbidity and suspensoids

General comments

Changes in the quantity of suspended particles can affect the concentrations of inorganic ions, including nutrients, dissolved in the water column (Sweeting, 1994; Dallas et al., 1998). Phosphate, for example, binds to sediment particles, sometimes resulting in deficiencies of this nutrient in rivers carrying high sediment loads. This ameliorating effect may be reversed if sediments are disturbed or other system variables, such as pH, change. High levels of suspended sediments can also decrease water temperature due to reflection of heat by the suspended particles, resulting in reduced heat absorption by the water molecules (Kirk, 1985). A more obvious negative correlation exists between suspended sediments in the water column and the extent to which light penetrates into a water body. Reduced light penetration limits the potential for primary production by phytoplankton and rooted macrophytes because the depth of the photic zone (the zone in which enough light is available for photosynthesis) is limited (e.g. Grobler et al., 1987), while visually-searching predators may be unable to locate prey.

Turbidity and suspensoids in N-PRs

In rivers, suspended sediment loads normally decrease with decreasing flow (Fig. 2; and Hrdinka et al., 2012) due to settling out of sediments from the water column and decreased supply of sediments from the catchment, resulting in clear water in pools. In both Australia (e.g. Bunn et al., 2006) and South Africa, however, many perennial as well as N-PRs are permanently turbid. This turbidity is the result not of suspended organic particles such as algae or their remains, but of very finely-divided clay particles that are so small (<0.45 µm or so) and highly charged that they remain permanently suspended in the water column (Kirk, 1985). Very high levels of turbidity significantly limit light penetration and thus photosynthesis in pools in many N-PRs (Bunn et al., 2006). Alexandrov et al. (2003) examined the relationship between suspended sediment concentrations and discharge in the Eshtemoa (a dryland ephemeral channel) in the northern Negev, Israel. They showed that only half of the observed variation in the concentration of suspended sediments could be linked to variations in discharge, implying that variations in sediment supply, and changes in the importance of source areas, are equally significant in this semi-arid catchment (Alexandrov et al., 2003).

pH and alkalinity

General comments

pH may vary naturally from time to time. Streams in the south-western Cape, for example, tend to be more acidic during winter (Britton et al., 1993) than summer because of release of acidic

humic substances from the soil and groundwater during winter rains (Raubenheimer and Day, 1991). In productive ecosystems, on the other hand, as a result of removal of CO_2 during the process of photosynthesis, pH may reach values of 10 or more during the day, falling to 8 or less at night (Sweeting, 1994).

The pH of water can have a marked effect on the species (chemical form) of metal ions and nutrients, and hence on bioavailability (Filella et al., 1995). Decreasing pH frequently also leads to increased levels of metal cations in the water column as a result of desorption from the surfaces of suspended particles. The metals most profoundly affected by pH shifts in the 4–7 range include Al, Cu, Hg, Pb and Fe (Sweeting, 1994). Alkaline conditions, on the other hand, can result in enhanced levels of un-ionised ammonia (NH_3), which is toxic to living organisms. Casey and Farr (1982) also noted that the equilibrium between sediment-bound and free phosphate ions was sensitive to pH: as pH increased, phosphate was displaced from the sediments into the water column.

pH in N-PRs

Values for pH seem to exhibit great variability in response to drying (Fig. 3). Mosley (2015) summarizes the conflicting literature for this topic and cites several studies where it would appear that site-specific factors (e.g. sulphide oxidation in one Finnish catchment, and decreased dilution of bicarbonate-dominated groundwater in another), are the causes of a decrease or, in the second case, an increase, in pH. Von Schiller et al. (2011) found that in a largely un-impacted, forested Mediterranean stream, pH decreased as flow decreased in summer. Boulton et al. (2000) report that in some Australian N-PRs, pH can fall as low as 4.5 due to the leaching of organic acids from leaf litter as pools start to dry-up. Diurnal changes in pH due to photosynthesis have been reported in systems in which algae and macrophytes are abundant (Williams, 2006).

Organic carbon

General comments

Dissolved organic matter (DOM) and particulate organic matter (POM) are derived largely from decaying plant and animal matter and occur naturally in streams, rivers and wetlands. These substances are critical components of the food-chain, forming the energy base for detritivores and decomposer bacteria and fungi.

Organic carbon in N-PRs

As flow decreases and disconnected pools appear, large amounts of detritus may accumulate in pools and in the dry channel (Larned et al., 2010). With further drying of the river bed, there is a shift to processes characteristic of the dry phase. The breakdown of organic matter slows down and changes from degradation arising from leaching, and the action of aquatic microbes and invertebrates, to processes controlled by photodegradation, physical abrasion and decomposition mediated by terrestrial microbes and invertebrates (Datry et al., 2014a). Once the river flows again, much of the DOM (Guarch-Ribot and Butturini, 2016) and POM are flushed downstream. In rivers, the dynamics of accumulation and decomposition of organic matter are affected by drying and hydrological contraction to form pools. Dahm et al. (2003) reported decreases in the levels of DOM due to hydrological

disconnection in semi-arid intermittent rivers and ephemeral streams, although contrasting results have been reported by various authors. For example, studies on a semi-arid intermittent stream in Spain reported no change in DOM levels in surface waters during contraction (Ylla et al., 2010; Von Schiller et al., 2015). Von Schiller et al. (2015) hypothesise that differences between N-PRs are related to the balance between autotrophy and heterotrophy in each stream. Low river flow (especially during hydrological contraction) increases the accumulation and retention of both coarse (e.g. small woody debris and leaves) and fine POM (Dewson et al., 2007). Isolated pools accumulate detritus, resulting in extreme respiration, and limited re-aeration rates lead to hypoxia (Von Schiller et al., 2011) resulting in significant changes in the concentration and forms of DOM and nutrients. Von Schiller et al. (2011) also noted chemical changes in DOM, which tended to become less aromatic with time in isolated pools, suggesting an increase in the influence of both algal and microbial processes on DOM as the pools fragment. In addition, the concentration of low-molecular and potentially labile DOM may increase in older isolated pools. Further, these authors noted rapid changes in the composition and concentration of DOM just before the stream totally desiccated at the surface. They suggest that this may be due to rapid exudation of DOM from algae and microbes under stress, and ultimately from cell lysis (Von Schiller et al., 2015). Quantitatively, however, Casas-Ruiz et al. (2016) consider degradation and leaching of accumulated leaf litter and animal remains to be the major sources of DOM in rivers, stream and wetlands.

Toxic substances

General comments

Toxic substances form a very diverse group of chemicals, from metals to organics, including pesticides, and whole effluents. Furthermore, there is an ever-growing list of emerging contaminants derived from pharmaceuticals, toiletries and other products (Richardson and Ternes, 2011). Particulate matter acts as both a source of, and a sink for, numerous anthropogenic and natural contaminants. Many contaminating organic wastes entering aquatic systems ultimately accumulate in sediments, where they may negatively affect the benthic biota, as well as entering human and pelagic food chains. Organic enrichment is probably one of the commonest and best documented forms of pollution in rivers (Day and Dallas, 2011). It is usually due to a heterogeneous range of chemical compounds, from excreta in sewage to contaminants in waste streams from industrial processes. The major consequence is usually a decrease in ambient oxygen levels due to microbial uptake of oxygen during the process of decomposition. Other potential effects of organic enrichment include increased turbidity, suspended solids and nutrients. On a more positive note, both DOM and POM can be effective in binding metal ions and reducing their toxicity (ANZECC/ARMCANZ, 2000).

Toxic substances in N-PRs

The pollution profiles of N-PR basins are becoming similar to watersheds in perennial rivers, which are characterised by point sources and non-point sources. Many non-perennial streams are becoming, or have already become, channels for raw wastewater and treated effluents as sewage treatment procedures in many semi-arid regions move from septic tanks

and cesspools to major sewage systems, which affect both the morphology and the water of the stream (Hassan and Egozi, 2001; Tal et al., 2010; Köck-Schulmeyer et al., 2011). Evaporation, which is likely to concentrate toxicants, is an important driver in N-PRs, as are interactions such as changes in pH, or increased temperature, that govern the release and adsorption of toxicants from and to sediments.

Major manipulation of river-associated wetlands may have unexpectedly serious consequences. The San Joaquin River in California is a case in point. As a result of urban runoff, spillage and drainage of irrigation water, and reduction in virgin discharges, concentrations of some trace elements have increased dramatically. For instance, the concentrations of boron have increased from roughly 30 to > 2 900 µg L^{-1} and of selenium from <1 to >6 µg L^{-1}, resulting in mass deformities and death of thousands of waterbirds and fish in the Kesterson wetlands along the river (Kingsford et al., 2006).

WATER QUALITY IN N-PRS DURING THE RE-WETTING PHASE

When flow resumes in N-PRs, depending on runoff intensity and the geomorphology of the catchment, the advancing wetted front can take the form of a flood bore (a wave), small rivulets, or expanding pools of upwelling groundwater (Jacobson et al., 2000, Jacobson and Jacobson, 2013; Datry et al., 2014a) and may be gradual or rapid (Datry et al., 2014b). Such events are unpredictable and can often, for example with flash floods occurring in deserts, be spectacular (Corti and Datry, 2012). Note that advancing and retreating fronts can occur also in the hyporheic zones of temporary rivers, as well as in river channels (Larned et al., 2010).

With the onset of rainfall and river flow, the hydrological link between the catchment and the river is renewed. Several studies (see Mosley, 2015) have found that during the re-wetting phase of the hydrological cycle nitrate is flushed from the catchment soils in the immediate post-drought period. This is likely to be due both to N having been concentrated in the catchment during the drought phase, and stimulation of mineralisation and nitrification when dried soils are rewetted. Leaching of organic matter and evaporite dissolution also cause spikes in solute concentrations in the waters of the newly flowing river (Jacobson et al., 2000; Hladyz et al., 2011; Corti and Datry, 2012). McLaughlin also found that rewetting following cessation of flow sometimes resulted in a release of significant quantities of organic nitrogen back into the river system. In contrast, Arce et al. (2015), found that in microcosm experiments, the process of rewetting sediments triggered denitrification.

In a similar way to rivers, the banks of reservoirs in semi-arid areas undergo desiccation during dry periods. Organic material on the banks becomes mineralized and after re-wetting N, P and C are dissolved into the water and released from the detritus (Gunkel et al., 2015). Thus, when flow resumes in the river, the concentrations of solutes and loads of suspended solids at the edge of the advancing front can be very high, and can exceed base flow concentrations by orders of magnitude. Organic matter, nutrients, and plant material (terrestrial plants, leaf-litter) that have accumulated on dry channel surfaces are transported downstream by advancing wetted fronts (Datry et al., 2014a). Non-perennial rivers are also likely to accumulate sediment (and in some situations associated contaminants) during periods of low-flow and lose them to flushing during periods of higher flow. In semi-arid

regions, suspended sediment loads are highest at the beginning of the wet season as flood waters scour the beds of particulate matter that has accumulated during the dry period. It has been estimated that re-suspended sediments may constitute more than 90% of the very high sediment loads in ephemeral dryland rivers in Israel (Powell et al., 1996). According to Datry et al. (2014a), advancing fronts in intermittent rivers can move many tons of organic matter to downstream receiving waters or to locations where flood attenuation results in deposition within the river channel. These deposits can be important carbon sources for heterotrophic organisms but they can also cause hypoxia (Rossouw et al., 2005) and fish and invertebrate kills (Hladyz et al., 2011). According to Datry et al. (2014a), the issue of large loads of organic matter released from intermittent rivers in pulses has not been fully studied. There may be consequences to the ecosystems of down-stream lakes, reservoirs and coastal environments in terms of eutrophication and oxygen depletion as well as greenhouse gas production and carbon sequestration.

During the re-wetting phase of a non-perennial stream, the dispersal of chemical constituents by the flowing water tends to homogenise nutrient concentrations, decreasing spatial heterogeneity (Chapman and Kramer, 1991; Von Schiller et al., 2011). The longer that flow persists in a N-PR, the more hydrological connectivity there will be between isolated pools, between groundwater and surface water, and between the river and the hyporheos and, as noted by Von Schiller et al. (2011), the greater the homogenisation of nutrients, but also of other WQ constituents. Malan and Day (2002) reviewed the literature concerning the effect of river discharge on instream WQ. They found that both catchment wash-off processes and instream processes influenced the resultant concentration of chemical constituents and values of physical variables. These findings were taken from the literature for perennial rivers; the longer water remains flowing within a non-perennial river, however, the more the trends described in Malan and Day (2002) for individual parameters will be relevant and can be used to predict WQ.

OVERARCHING PROCESSES DRIVING WATER QUALITY IN N-PRS

From his review of studies on the effect of drought on WQ in freshwater ecosystems, Mosley (2015) was able to identify three key factors or processes controlling the concentrations of chemical constituents and the magnitudes of physical variables. The first factor is – fairly obviously – mass balance. In a well-mixed water body at any point in time, the concentration of a chemical constituent is equal to the sum of all the masses from different sources (inflow, surface run-off, groundwater, point sources) divided by the total dilution volume. Processes such as biological or chemical transformation and sedimentation may also remove a constituent from the pool. Thus during drought, as volume decreases, the concentration will increase if the loading remains the same. The residence time of the remaining water also increases, causing sediments to settle out and supplies from the catchment to be cut off.

The second key factor identified by Mosley (2015) is temperature. During drought, ambient air temperatures commonly increase, accompanied by increases in water temperature. Consequently, another commonly observed effect of drought is an increased incidence and persistence of stratification and harmful algal blooms due to lower flows and increased thermal energy. Temperature increases and increased stratification result in lower dissolved oxygen concentrations.

The third key factor identified by Mosley (2015) is the increased importance of instream processes. During drought, when there is little or no flow in a river, not only is the residence time of the water in a given pool increased, but also that of chemical constituents, both dissolved and particulate; as a consequence, instream processes become more important than they are during the lotic state. For example, macrophytes and benthic algae become more prolific, possibly because of the longer residence time of nutrients and also because of the increased penetration of sunlight and decreased scouring effects. Other instream processes that Mosley (2015) identifies as important in controlling water chemistry are the movement of chemical constituents across the sediment/water interface, and sediment-associated processes such as denitrification.

In a key paper describing emerging concepts in temporary river ecology, Larned et al. (2010) consider that in temporary rivers, flow pulses trigger biogeochemical and physiological processes causing such systems to act as longitudinal, biogeochemical reactors. They describe how, because of the low moisture content in dry river beds, there are also low levels of microbial activity and consequently low rates of decomposition. With the onset of a flood event in a dry river bed, an advancing wetted front develops longitudinally in channels and laterally on floodplains. The advancing wetted front carries high loads of particulate organic matter (POM), sediments and organisms and also has high concentrations of dissolved chemical constituents. Through repeated cycles of flood pulses, material is moved downstream to a new reaction site (pool or sand bar) where it gradually dries out and water-dependent degradation processes cease.

MANAGEMENT OF WATER QUALITY IN NON-PERENNIAL RIVERS

Water quality and hydrology

In order to balance competing demands for water, many countries require water allocations (environmental flow requirements) to be specified for rivers, including N-PRs. Suitable allocations, with regard to both quality and quantity of water, have proved to be challenging (Stefanidis et al., 2016). Choy et al. (2002) recommend that, in setting water allocations for N-PRs, attention is paid to maintenance of natural periods of hydrological connectivity and of fragmentation, since periods of both are necessary for the maintenance of ecological integrity (Bishop-Taylor et al., 2017). In South Africa, the development of a method known as DRIFT-Arid is documented by Seaman et al. (2016a) and applied (Seaman et al. 2016b) to the Mokolo River in South Africa. (The DRIFT method for assessing water requirements for perennial rivers is documented in King and Brown, 2009).

Even for organisms adapted to living in unimpacted N-PRs, natural fluctuations in WQ variables such as water temperature, oxygen concentration and salinity are potentially lethal. When such rivers are additionally affected by disturbances such as water abstraction or the discharge of pollutants, the challenges to surviving in these systems are magnified. Deksissa et al. (2003) discuss management options for WQ improvement in the Crocodile River, Mpumalanga, South Africa. Whilst the river was probably perennial under natural conditions, it no longer is, so this study has some useful lessons for N-PRs in semi-arid areas that are subjected to pollution of different kinds and from different sources. These authors suggest that limiting water abstraction, setting and implementing effluent limits, and augmenting flow during low-flow periods with releases from dams, will improve WQ (mainly by decreasing salinity and nutrient concentrations in this case) during periods of low flow. Kingsford et al. (2006) further discuss the effects of dams on desert rivers, including N-PRs.

Froebrich (2005) points out that it is difficult to predict WQ in temporary rivers because of the spatial heterogeneity in catchment processes, which leads to variability in contaminant build-up during the dry period, coupled with variability in the timing and extent of rainfall events. By implication, water chemistry needs to be assessed for each river and plans for managing it need to be based on the empirical data generated. Because of the overall lack of 'dilution capacity' in non-perennial systems an obvious way to protect them is to limit input of both point- and non-point sources of pollutants. An established method for reducing the input of sediments, nutrients and other contaminants into surface waters, especially N-PRs (Rosado et al. 2012)~~expanding during the wet autumn and winter period and contracting, often to a series of large pools, during the spring and summer. There can be considerable variation in annual rainfall patterns and current knowledge about the hydrology and dynamics of temporary streams is very limited. This means the confidence in the ability to predict the likely ecological consequences of climate change and increased water demand is low. To improve the existing level of understanding and predictive capability, a hydrological modelling study was carried out on the River Pardiela catchment in southern Portugal. The main objectives were to study long-term patterns of air temperature and precipitation, to apply the Soil and Water Assessment Tool (SWAT~~, is through the preservation of well-developed buffer strips of natural vegetation (e.g. Macfarlane et al., 2014; Lee and Fisher et al., 2016). It is during floods that large loads of sediment and contaminants enter the rivers (Froebrich, 2005) and it is also under these conditions that riparian vegetation is uprooted and flushed downstream. In very arid areas, though, there may well be no natural riparian vegetation, or the vegetation may consist of well-established trees within the water course, neither of which protects the river from lateral movement of sediments and other pollutants.

Choy et al. (2002) investigated the ecosystem integrity of four N-PRs located in the arid centre of Australia. Whilst the rivers at most of the study sites were found to be little affected by human activities, damage by livestock to the physically fragile banks was commonly noted. Similar impacts have been noted in the Doring River in the Western Cape (Paxton, 2017, pers. comm.). Rosado et al. (2012) consider exclusion of livestock from temporary Mediterranean streams as being a key factor in management, since uncontrolled roaming of livestock can damage stream beds, pollute the water, and cause health hazards to other users. Note that livestock play an important role in the economy and culture of subsistence farmers in Africa, so excluding them from riparian areas of N-PRs is extremely challenging, particularly where pools in river beds are the only sources of drinking water for stock animals. Note, too, that until recently, large herbivores, including megaherbivores such as elephants, were an integral part of the landscape even in arid areas in Africa; by implication, trampling must have been a greater factor in the natural perturbation of N-PRs than it is now.

Assessing environmental condition of N-PRs

Assessing the environmental condition ('ecological integrity') of any ecosystem requires information about some base state against

which the current condition of an ecosystem can be assessed. This base state, the 'reference condition', is usually considered to be the unimpacted pristine condition and is normally assessed separately for each type of aquatic ecosystem in each ecoregion. In such assessments it is necessary to distinguish between biotic responses to natural variability in hydrology and climate (i.e. drought, drying and flooding) and those brought about by anthropogenic impacts. Several authors (Arthington et al., 2014; Cid et al., 2016) have noted, however, that it is particularly difficult to assess the reference condition of N-PRs because of their dynamic nature and the sometimes decades-long long (but unmeasured) periodicity of drought and flood. Recognising this variability, Seaman et al. (2010) used the present state as the equivalent of the reference condition when developing a method for ascertaining the environmental water requirements of N-PRs. The predicted effects of climate change in the region – decreased rainfall, increased temperatures, greater variability and wider extremes (e.g. Schulze 2011) – will add to the challenge of assessing environmental condition in N-PRs.

Various approaches to assessing the ecosystem condition of N-PRs are reported in the literature. For non-perennial systems in the arid centre of Australia, Choy et al. (2002) used four aspects: the level of human influence; habitat condition; water chemistry; and aquatic invertebrates. In contrast, given challenges in the availability of water quality data, and the variability in WQ itself, Seaman et al. (2013) proposed the use of expert knowledge based on useful catchment information, including land use, potential pollution sources, soil types and geology. (Note that a similar approach has been used to make predictions of likely WQ in South African wetlands – Malan and Day, 2012.)

It seems that no bioassessment tools have successfully been developed for estimating the WQ of non-perennial rivers and wetlands. The well-known SASS (South African Scoring System) tool, based on invertebrate families and widely used in South Africa for assessing WQ and the ecosystem condition of perennial rivers, has proved not to be effective in wetlands (Bowd et al., 2006; Bird et al., 2013). Our hypothesis is that invertebrate assemblages will be no more effective in assessing the condition of N-PRs. As part of the MIRAGE project, however, Prat et al. (2014) have developed tools for assessing the ecosystem status of temporary Mediterranean rivers. According to Cid et al. (2016), though, these tools suffer the disadvantage of needing hydrological data, which is frequently lacking. In response these authors have developed a prototype tool (Bio-AS) for assessing temporary Mediterranean streams, using macro-invertebrates. The ratio EPT/OCH (i.e. the ratio between Ephemeroptera + Plecoptera + Trichoptera, and Odonata + Coleoptera + Hemiptera) has also been used to assess flow connectivity in Mediterranean streams (Bonada et al., 2006). It should be noted, though, that although these taxa can be useful in seasonal streams, rivers in very arid areas are unlikely to support more than one or two species, if any, of these taxa.

Water quality guidelines

In most countries, WQ criteria for aquatic ecosystems are designed for the protection of surface waters, particularly rivers. It is evident from the preceding discussions, though, that WQ in non-perennial rivers is naturally highly variable. Furthermore, the concentrations of some chemical constituents (e.g. salts, and therefore salinity) can rise markedly at the same time that others (e.g. DO) decrease (Mosley et al., 2012) and become limiting to the growth and even the survival of

some organisms. Von Schiller et al. (2011), for instance, note that because of the intrinsically high temporal and spatial variability in nutrient concentrations, reference conditions used to evaluate the status of permanent streams may not be appropriate for temporary streams. In fact, uncritical use in non-perennial systems of WQ criteria developed for perennial rivers might be harmful to some members of the biota (Rossouw et al., 2005). In particular, some of the natural values recorded for DO and salinity in N-PRs would, in perennial rivers, indicate poor WQ. In this regard, Feio et al. (2014) examined the data from a large number of national reference sites for perennial and non-perennial Mediterranean rivers from seven countries in an effort to characterize and define thresholds for the Least Disturbed Condition. Based on their results they were able to recommend a lower threshold value for DO for non-perennial than for perennial rivers.

Principles of water quality management in non-perennial rivers

It is possible to identify some general features of N-PRs:
- Hydrological regimes are seldom predictable with any certainty
- WQ varies naturally over time and space
- Groundwater often determines the WQ of surface water, especially in pools
- WQ in non-perennial rivers and pools may be affected by activities far upstream in the catchment
  Furthermore, we still have no more than a sketchy understanding of the extent to which data on any one system can be applied to any other.

We therefore offer the following basic principles as guides to the management of these systems.
- Rivers need to be assessed on a case-by-case basis until such time as we can apply general principles to an understanding of WQ in N-PRs.
- Abstraction of both surface and groundwater, and storage of water in upstream dams, needs to be strictly limited and understanding of the groundwater regime is crucial if we are to avoid unsustainable 'mining' of the resource.
- Effluents need to be controlled and conservative effluent standards need to be set, sometimes on a case-by-case basis, for both ground and surface waters.
- Flows may need to be augmented at certain times of the year.
- Buffer zones need to be set, and where possible these should be designed to encourage the growth of natural vegetation.

It seems to us that the most useful step towards improving management of these systems would be the development of a much-simplified version of DRIFT-Arid for assessing water requirements for N-PRs, using additional test cases across the spectrum from episodic to semi-permanent systems and in different biomes. This should be linked to the development of a suitable monitoring programme for a number of N-PRs, particularly those for which water allocations have already been set. In seeking to deepen our knowledge of the ecological functioning of N-PRs it needs to be understood that because of the inherent variability of these systems, short-term investigations are of limited use and that study projects need to be long-term (10 years or more).

CONCLUSION

Processes taking place in N-PRs are poorly understood. Lack of knowledge, combined with the dynamic and sometimes

unpredictable nature of N-PRs, makes them challenging to manage. In consequence, intermittent rivers are particularly vulnerable in many parts of the world because of a lack of legislation, and therefore a lack of adequate management practices, protecting them and their waters (Datry et al., 2014a). Because of climate change, in southern Africa temperatures are expected to rise, rainfall to decrease, and inter-annual variability in rainfall to increase (Schulze, 2011), especially in those arid parts where temporary streams are located. Such conditions will invariably lead to an increased demand for water by humans, while the biotas of the rivers themselves will be subject to increased selective pressures; balancing conservation of N-PRs with human water demands will become more and more difficult over time (Rosado et al., 2012).

ACKNOWLEDGEMENTS

REFERENCES

ALEXANDROV Y, LARONNE JB and REID I (2003) Suspended sediment concentration and its variation with water discharge in a dryland ephemeral channel, northern Negev, Israel. *J. Arid Environ.* 53 (1) 73–84. https://doi.org/10.1006/jare.2002.1020

ANZECC/ARMCANZ (2000) *Australian and New Zealand Guidelines for Fresh and Marine Water Quality* (NWQMS4, 2000). Australian and New Zealand Environmental and Conservation Council/ Agriculture and Resource Management Council of Australia and New Zealand, Canberra, ACT. https://doi.org/10.1111/anzs.2000.42.issue-3

ARCE MI, SÁNCHEZ-MONTOYA M, DEL M and GÓMEZ R (2015) Nitrogen processing following experimental sediment rewetting in isolated pools in an agricultural stream of a semiarid region. *Ecol. Eng.* 77 (3) 233–241. https://doi.org/10.1016/j.ecoleng.2015.01.035

ARTHINGTON AH, BERNARDO JM and ILHÉU M (2014) Temporary rivers: linking ecohydrology, ecological quality and reconciliation ecology. *River Res. Applic.* 30 1209-1215. https://doi.org/10.1002/rra.2831

BAILEY PCE, BOON PI, BLINN DW and WILLIAMS WD (2006) Salinisation as an ecological perturbation to rivers, streams and wetlands of arid and semi-arid regions. Chapter 10. In: Kingsford R (ed.) *Ecology of Desert Rivers*. Cambridge University Press, Cambridge. 354 pp. ISBN: 0521818257 (hbk.), 9780521818254 (hbk.). 280–314.

BERNARDO JM and ALVES MJ (1999) New perspectives for ecological flow determination in semi-arid regions: A preliminary approach. *Reg. Rivers: Res. Manage.* 15 221–229. https://doi.org/10.1002/(sici)1099-1646(199901/06)15:1/3<221::aid-rrr537>3.3.co;2-1

BIRD MS, MLAMBO M and DAY JA (2013) Macroinvertebrates as unreliable indicators of human disturbance in temporary depression wetlands of the south-western Cape, South Africa. *Hydrobiologia* 720 19–37. https://doi.org/10.1007/s10750-013-1618-2

BISHOP-TAYLOR R, TULBURE MG and BROICH M (2017) Surface-water dynamics and land use influence landscape connectivity across a major dryland region. *Ecol. Appl.* 27(4) 1124–1137. https://doi.org/10.1002/eap.1507

BONADA N, RIERADEVALL M, PRAT N and RESH VH (2006) Benthic macroinvertebrate assemblages and macrohabitat connectivity in Mediterranean-climate streams of northern California. *J. N. Am. Benthol. Soc.* 25 (1) 32-43. https://doi.org/10.1899/0887-3593(2006)25[32:bmaamc]2.0.co;2

BOULTON AJ, SHELDON F and JENKINS KM (2006) Natural disturbance and aquatic invertebrates in desert rivers. Chapter 6 (pp.) In: Kingsford R (ed.) *Ecology of Desert Rivers*. Cambridge University Press, Cambridge. 354 pp. ISBN: 0521818257 (hbk.), 9780521818254 (hbk.). 133–153

BOULTON AJ, SHELDON F, THOMS MC and STANLEY EH (2000) Problems and constraints in managing rivers with contrasting flow regimes. In: Boon PJ, Davies BR and Petts GE (eds) *Global Perspectives in River Conservation: Science, Policy and Practice*. John Wiley and Sons, Chichester. 415–430.

BOUWMAN AF, BEUSEN AHW, LASSALETTA L, VAN APELDOORN DF, VAN GRINSVEN HJM, ZHANG J and VAN ITTERSUM MK (2017) Lessons from temporal and spatial patterns in global use of N and P fertilizer on cropland. *Sci. Rep.* 7 40366. https://doi.org/10.1038/srep40366

BOWD R, KOTZE DK, MORRIS CD AND QUINN NW (2006) Testing the applicability of the SASS5 scoring procedure for assessing wetland health: a case study in the KwaZulu-Natal midlands, South Africa. *Afr. J. Aquat. Sci.* 31 (2) 229–246. https://doi.org/10.2989/16085910609503893

BRITTON DL, DAY JA and HENSHALL-HOWARD M-P (1993) Hydrochemical responses during storm events in a South African mountain catchment: the influence of antecedent conditions. *Hydrobiologia* 250 143–157. https://doi.org/10.1007/bf00008585

BROWN C, COLVIN C, HARTNADY C, HAY R, LE MAITRE D and RIEMAN K (2003) Ecological and environmental impacts of large-scale groundwater development in the Table Mountain Group. Discussion document. Water Research Commission, Pretoria, South Africa.

BUNN SE, BALCOMBE SR, DAVIES PM, FELLOWS CS and MACKENZIE-SMITH FJ (2006) Aquatic productivity and food webs of desert river ecosystems. In: Kingsford R (ed.) *Ecology of Desert Rivers*. Cambridge University Press, Cambridge. 354 pp.

CAMPBELL IC (1982) Biological water quality monitoring: An Australian viewpoint In: Hart BT (ed.) *Water Quality Management: Monitoring Programs and Diffuse Runoff*. Water Studies Centre, Chisholm Institute of Technology and Australian Society for Limnology (joint publishers), Melbourne.

CASAS-RUIZ JP, TITTEL J, VON SCHILLER D, CATALÁN N, OBRADOR B, GÓMEZ-GENER L, ZWIRNMANN E, SABATER S and MARCÉ R (2016) Drought induced discontinuities in the source and degradation of dissolved organic matter in a Mediterranean river. *Biogeochemistry* 127 (1) 125–139. https://doi.org/10.1007/s10533-015-0173-5

CASEY H and FARR IS (1982) The influence of within-stream disturbance on dissolved nutrient levels during spates. *Hydrobiologia* 92 447–462. https://doi.org/10.1007/bf02391959

CHAPMAN LJ and KRAMER DL (1991) Limnological observations of an intermittent tropical dry forest stream. *Hydrobiologia* 226 153–166. https://doi.org/10.1007/bf00006857

CHOY SC, THOMSON CB and MARSHALL JC (2002) Ecological condition of central Australian arid-zone rivers. *Water Sci. Technol.* 45 (11) 225–232. https://doi.org/10.2166/wst.2002.0399

CID N, VERKAIK I, GARCIA-ROGER EM, RIERADEVALL M, BONADA N, SANCHEZ-MONTOYA MM, GOMEZ R, SUAREZ ML, VIDAL-ABARCA MR, DEMARTINI D and co-authors (2016) A biological tool to assess flow connectivity in reference temporary streams from the Mediterranean Basin. *Sci. Total Environ.* 540 178–190. https://doi.org/10.1016/j.scitotenv.2015.06.086

CORTI R and DATRY T (2012) Invertebrates and sestonic matter in an advancing wetted front travelling down a dry river bed (Albarine, France). *Freshwater Sci.* 31 1187–1201. https://doi.org/10.1899/12-017.1

DAHM CN, BAKER MA, MOORE DI and THIBAULT JR (2003) Coupled biogeochemical and hydrological responses of streams and rivers to drought. *Freshwater Biol.* 48 1219–1123. https://doi.org/10.1046/j.1365-2427.2003.01082.x

DALLAS H (2008) Water temperature and riverine ecosystems: an overview of knowledge and approaches for assessing biotic responses, with special reference to South Africa. *Water SA* 34 (3) 393–404.

DALLAS HF and DAY JA (2004) The effect of water quality variables on aquatic ecosystems: a review. WRC Report No. TT 224/04. Water Research Commission, Pretoria.

DALLAS HF, DAY JA, MUSIBONO DE and DAY EG (1998) Water quality for aquatic ecosystems: Tools for evaluating regional guidelines. WRC Report No. 626/1/98. Water Research Commission, Pretoria.

DATRY T, LARNED ST and TOCKNER K (2014a). Intermittent rivers: A challenge for freshwater ecology. *BioScience* **64** (3) 229–235. https://doi.org/10.1093/biosci/bit027

DATRY T, LARNED ST, FRITZ KM, BOGAN MT, WOOD PJ, MEYER EI and SANTOS AN (2014b) Broad-scale patterns of invertebrate richness and community composition in temporary rivers: effects of flow intermittence. *Ecography* **37**(1) 94–104. http://dx.doi.org/10.1111/j.1600-0587.2013.00287.x

DATRY T, PELLA H, LEIGH C, BONADA N and HUGUENY B (2016) A landscape approach to advance intermittent river ecology. *Freshwater Biol.* **61** (8) 1200–1213. https://doi.org/10.1111/fwb.12645

DAVIES B and DAY J (1998) *Vanishing Waters.* University of Cape Town Press, Cape Town, South Africa. 487 pp. https://doi.org/10.15641/ghi.v2i1.728

DAY JA and DALLAS HF (2011) Understanding the basics of water quality. In: Grafton Q (ed.) *Water Resources Planning and Management.* Cambridge University Press, Cambridge. 777 pp. https://doi.org/10.1017/cbo9780511974304.007

DEODHAR AS, ANSARI A, SHARMA S, JACOB N, KUMAR US and SINGH G (2014). *Isotope Techniques for Water Resources Management* Newsletter **337** 29–35. Bhabha Atomic Research Centre, Department of Atomic Energy, Government of India.

DEKSISSA T, ASHTON PJ and VANROLLEGHEM PA (2003) Control options for river water quality improvement: A case study of TDS and inorganic nitrogen in the Crocodile River (South Africa). *Water SA* **29** (2) 209–218. https://doi.org/10.4314/wsa.v29i2.4858

DEWSON ZS, JAMES ABW and DEATH RG (2007) Stream ecosystem functioning under reduced flow conditions. *Ecol. Appl.* **17** 1797–1808. https://doi.org/10.1890/06-1901.1

DONNELLY TH, GRACE MR and HART BT (1997) Algal blooms in the Darling–Barwon River, Australia. *Water Air Soil Pollut.* **99** 487–496. https://doi.org/10.1007/bf02406888

DWAF (Department of Water Affairs and Forestry, South Africa) (1996) South African Water Quality Guidelines. Volume 7: Aquatic ecosystems. Department of Water Affairs and Forestry, Pretoria.

FAZI S, VÁZQUEZ E, CASAMAYOR EO, AMALFITANO S and BUTTURINI A (2013) Stream hydrological fragmentation drives bacterioplankton community composition. *PloS one* **8** (5) e64109. https://doi.org/10.1371/journal.pone.0064109

FILELLA M, TOWN R and BUFFLE J (1995) Speciation in fresh waters. In: Ure AM and Davidson CM (eds.) *Chemical Speciation in the Environment.* Blackie Academic and Professional, New York.

FEIO MJ, AGUIAR FC, ALMEIDA SFP, FERREIRA J, FERREIRA MT, ELIAS C, SERRA SRQ, BUFFAGNI A, CAMBRA J, CHAUVIN C and co-authors (2014) Least Disturbed Condition for European Mediterranean rivers. *Sci. Total Environ.* **476–477** 745–756. https://doi.org/10.1016/j.scitotenv.2013.05.056

FROEBRICH J (2005) Water quality dynamics in temporary streams and implications for water management strategies in semi-arid regions. *Geophysical Research Abstracts (GRA)* **7** 01857. http://www.cosis.net/abstracts/EGU05/01857/EGU05-J-01857.pdf

GROBLER DC, TOERIEN DF and ROSSOUW JN (1987) A review of sediment/water quality interaction with particular reference to the Vaal River system. *Water SA* **13** 15–22.

GROMIEC MJ, LOUCKS DP and ORLOB GT (1983) Stream quality modelling. In: Orlobb GT (ed.) *Mathematical Modelling of Water Quality: Streams, Lakes and Reservoirs.* John Wiley and Sons, New York.

GUARCH-RIBOT A and BUTTURINI A (2016) Hydrological conditions regulate dissolved organic matter quality in an intermittent headwater stream. From drought to storm analysis. *Sci. Total Environ.* **571** 1358–1369. https://doi.org/10.1016/j.scitotenv.2016.07.060

GUNKEL G, LIMA D, SELGE F, SOBRAL M and CALADO S (2015) Aquatic ecosystem services of reservoirs in semi-arid areas: sustainability and reservoir management. *WIT Trans. Ecol. Environ.* **197** 1743–3541. https://doi.org/10.2495/rm150171

HASSAN MA and EGOZI R (2001) Impact of wastewater discharge on the channel morphology of ephemeral streams. *Earth Surf. Process. Landf.* **26** (12) 1285–1302. https://doi.org/10.1002/esp.273

HAWKES HA (1979) Invertebrates as indicators of river water quality. In: James A and Evison L (eds.) *Biological Indicators of Water Quality.* John Wiley and Sons, Chichester.

HLADYZ S, WATKINS SC, WHITWORTH KL and BALDWIN DS (2011) Flows and hypoxic blackwater events in managed ephemeral river channels. *J. Hydrol.* **401** 117–125. https://doi.org/10.1016/j.jhydrol.2011.02.014

HRDINKA T, NOVICKÝ O, HANSLÍK E and RIEDER M (2012) Possible impacts of floods and droughts on water quality. *J. Hydro-environ. Res.* **6** (2) 145–150. https://doi.org/10.1016/j.jher.2012.01.008

HUMAN LRD, MAGORO ML, DALU T, PERISSINOTTO R, WHITFIELD AK, ADAMS JB, DYSEL SHP and RISHWORTH GM (2018) Natural nutrient enrichment and algal responses in near pristine micro-estuaries and micro-outlets. *Sci. Total Environ.* **624** 945–954. https://doi.org/10.1016/j.scitotenv.2017.12.184

JACOBSON PJ (1997) An ephemeral perspective of fluvial ecosystems: viewing ephemeral rivers in the context of current lotic ecology. PhD dissertation, Faculty of the Virginia Polytechnic Institute and State University, Blacksburg, Virginia.

JACOBSON PJ and JACOBSON KM (2013) Hydrologic controls of physical and ecological processes in Namib Desert ephemeral rivers: Implications for conservation and management. *J. Arid Environ.* **93** 1–14. https://doi.org/10.1016/j.jaridenv.2012.01.010

JACOBSON PJ, JACOBSON KM, ANGERMEIER PL and CHERRY DS (2000) Variation in material transport and water chemistry along a large ephemeral river in the Namib Desert. *Freshwater Biol.* **44** 481–491. https://doi.org/10.1046/j.1365-2427.2000.00604.x

JAEGER KL, OLDEN JD and PELLAND NA (2014) Climate change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. *PNAS* **111** (38) 13894–13899. https://doi.org/10.1073/pnas.1320890111

KEDDY P (2010) *Wetland Ecology: Principles and Conservation.* (2nd edn.). Cambridge University Press, Cambridge. 495 pp.

KIBRIA G (2016) World rivers in crisis: water quality and water dependent biodiversity are at risk- threats of pollution, climate change & dams development. URL: https://www.researchgate.net/publication/263852254_World_Rivers_in_Crisis_Water_Quality_and_Water_Dependent_Biodiversity_Are_at_Risk-_Threats_of_Pollution_Climate_Change_and_Dams_Development. https://doi.org/10.24926/2471190x.1546

KING J and BROWN C (2009) Integrated basin flow assessments: concepts and method development in Africa and South-east Asia. *Freshwater Biol.* **55** (1) 127–146. https://doi.org/10.1111/j.1365-2427.2009.02316.x

KINGSFORD RT, LEMLEY AD and THOMPSON JR (2006) Impacts of dams, river management and diversion on desert rivers. Chapter 8 (pp203-248) in: Kingsford R (ed.) *Ecology of Desert Rivers.* Cambridge University Press, Cambridge. 354 pp.

KIRK JTO (1985) Effects of suspensoids (turbidity) on penetration of solar radiation in aquatic ecosystems. *Hydrobiologia* **125** (1) 195–208. https://doi.org/10.1007/bf00045935

KÖCK-SCHULMEYER M, GINEBREDA A, POSTIGO C, LÓPEZ-SERNA R, PÉREZ S, BRIX R, LLORCA M, DE ALDA ML, PETROVIĆ M, MUNNÉ A and co-authors (2011) Wastewater reuse in Mediterranean semi-arid areas: the impact of discharges of tertiary treated sewage on the load of polar micro-pollutants in the Llobregat river (NE Spain). *Chemosphere* **82** (5) 670–678. https://doi.org/10.1016/j.chemosphere.2010.11.005

LA VT and COOKE SJ (2011) Advancing the science and practice of fish kill investigations. *Rev. Fish. Sci.* **19** (1) 21–33.

LARNED ST, DATRY T, ARSCOTT DB and TOCKNER K (2010) Emerging concepts in temporary-river ecology. *Freshwater Biol.* **55** (4) 717–738. https://doi.org/10.1111/j.1365-2427.2009.02322.x

LARSEN LG, HARVEY JW and MAGLIO MM (2014) Dynamic hyporheic exchange at intermediate timescales: testing the relative importance of evapotranspiration and flood pulses. *Water Resour. Res.* **50** 318–335. https://doi.org/10.1002/2013wr014195

LEE J and FISHER C (2016) Arid green infrastructure for water control and conservation: state of the science and research needs for arid/semi-arid regions. Report prepared by The Scientific Consulting Group, Inc. for the Environmental Protection Agency, Las Vegas, NV 89193. https://doi.org/10.13031/2013.13834

LELIAVSKY S (1955) Sediment suspension. In: *Introduction to Fluvial Hydraulics.* Constable and Co., New York. 181–191.

LILLEBØ AI, MORAIS M, GUILHERME P, FONSECA R, SERAFIM A and NEVES R (2007) Nutrient dynamics in Mediterranean temporary streams: A case study in Pardiela catchment (Degebe River, Portugal). *Limnologica* **37** (4) 337–348. https://doi.org/10.1016/j.limno.2007.05.002

MACFARLANE DM, BREDIN IP, ADAMS JB, ZUNGU MM, BATE GC and DICKENS CWS (2014) Preliminary guideline for the determination of buffer zones for rivers, wetlands and estuaries: Consolidated report. WRC Report No. TT 610/14. Water Research Commission, Pretoria, South Africa.

MALAN HL and DAY JA (2002) Linking discharge, water quality and biotic response in rivers: a literature review. WRC Report No. 956/2/02. Water Research Commission, Pretoria.

MALAN HL and DAY JA (2012) Water quality and wetlands: defining Ecological Categories and links with land-use. WRC Report No. 1921/1/12. Water Research Commission, Pretoria.

MCLAUGHLIN C (2008) Evaporation as a nutrient retention mechanism at Sycamore Creek, Arizona. *Hydrobiologia* **603** 241–252. https://doi.org/10.1007/s10750-007-9275-y

MITSCH WJ and GOSSELINK JG (2007) *Wetlands* (4th edn.). John Wiley & Sons, Inc., Hoboken, New Jersey. 582 pp.

MOSLEY LM (2015) Drought impacts on the water quality of freshwater systems; review and integration. *Earth Sci. Rev.* **140** 203–214. https://doi.org/10.1016/j.earscirev.2014.11.010

MOSLEY LM, ZAMMIT B, LEYDEN E, HENEKER TM, HIPSEY MR, SKINNER D and ALDRIDGE KT (2012) The impact of extreme low flows on the water quality of the Lower Murray River and Lakes (South Australia). *Water Resour. Manage.* **26**(13) 3923–3946. https://doi.org/10.1007/s11269-012-0113-2

NOURI J, MIRBAGHERI SA, FARROKHIAN F, JAAFARZADEH N and ALESHEIKH AA (2009) Water quality variability and eutrophic state in wet and dry years in wetlands of the semiarid and arid regions. *Environ. Earth Sci.* **59** (7) 1397–1407. https://doi.org/10.1007/s12665-009-0126-1

PAERL HW and PAUL VJ (2012) Climate change: links to global expansion of harmful cyanobacteria. *Water Res.* **46** 1349-1363. https://doi.org/10.1016/j.watres.2011.08.002

PARSONS R and WENTZEL J (2007) Groundwater Resource Directed Measures Manual. WRC Report No. TT299/07. Water Research Commission, Pretoria.

PAXTON B (2017) Personal communication, April 2017. B. Paxton, Freshwater Research Centre, Imhoff's Gift, Cape Town.

POWELL DM, REID I, LARONNE JB and FROSTICK L (1996) Bed load as a component of sediment yield from a semi-arid watershed of the northern Negev. *IAHS Publications-Series of Proceedings and Reports-Intern Assoc Hydrological Sciences* **236** 389–398.

PRAT N, GALLART F, VON SCHILLER D, POLESELLO S, GARCÍA-ROGER EM, LATRON J, RIERADEVALL M, LLORENS P, BARBERÁ GG, BRITO D and co-authors (2014) The MIRAGE toolbox: An integrated assessment tool for temporary streams. *River Res. Appl.* **30** 1318–1334. https://doi.org/10.1002/rra.2757

RAU GC, HALLORAN LJS, CUTHBERT MO, ANDERSEN MS, ACWORTH R I and TELLAM JH (2017) Characterising the dynamics of surface water-groundwater interactions in intermittent and ephemeral streams using streambed thermal signatures. *Adv. Water Resour.* **107** 354–369. https://doi.org/10.1016/j.advwatres.2017.07.005

RAUBENHEIMER CM and DAY JA (1991) *In vitro* leaching of two sclerophyllous fynbos plants. *Hydrobiologia* **224** 167–174. https://doi.org/10.1007/bf00008466

RICHARDSON SD and TERNES TA (2011) Water analysis: Emerging contaminants and current issues. *Anal. Chem.* **83** 4616–4648. https://doi.org/10.1021/ac200915r

RIDGE T, PILLINGER P and WALTERS S (1995) Alleviating the problems of excessive algal growth. In: Harper DM and Ferguson AJD (eds.) *The Ecological Basis for River Management.* John Wiley & Sons, Chichester. 630 pp.

ROSADO J, MORAIS M, SERAFIM A, PEDRO A, SILVA H, POTES M, BRITO D, SALGADO R, NEVES R, LILLEBØ A and co-authors (2012) Key factors in the management and conservation of temporary Mediterranean streams: A case study of the Pardiela River, Southern Portugal. In: Boon PJ and Raven PJ (eds) *River Conservation and Management.* John Wiley & Sons, Ltd, Hoboken, New Jersey. https://doi.org/10.1002/9781119961819.ch22

ROSSOUW L, AVENANT MF, SEAMAN MT, KING JM, BARKER CH, DU PREEZ PJ, PELSER AJ, ROOS, JC, VAN STADEN JJ, VAN TONDER GJ and co-author (2005) Environmental water requirements in non-perennial systems. WRC Report No 1414/1/05. Water Research Commission, Pretoria.

ROSSOUW L and VOS T (2008) Water quality component. In: Seaman et al. (2010) Developing a method for determining the environmental water requirements for non-perennial systems. WRC Report No. TT 459/10. Water Research Commission, Pretoria.

SCHULZE RE (2011) Approaches towards practical adaptive management options for selected water-related sectors in South Africa in a context of climate change. *Water SA* **37** (5) 621–645. https://doi.org/10.4314/wsa.v37i5.1

SEAMAN MT, AVENANT MF, WATSON M, KING JM, ARMOUR J, BARKER CH, DOLLAR E, DU PREEZ PJ, HUGHES D, ROSSOUW L and co-author (2010) Developing a method for determining the environmental water requirements for non-perennial systems. WRC Report No. TT 459/10. Water Research Commission, Pretoria.

SEAMAN MT, WATSON M, AVENANT MF, JOUBERT AR, KING JM, BARKER CH, ESTERHUYSE S, GRAHAM D, KEMP ME, LE ROUX PA and co-authors (2013) Testing a methodology for environmental water requirements in non-perennial rivers. The Mokolo River case study. WRC Report No. TT 579/13. Water Research Commission, Pretoria.

SEAMAN M, WATSON M, AVENANT M, KING J, JOUBERT A, BARKER C, ESTERHUYSE S, GRAHAM D, KEMP M, LE ROUX P and co-authors (2016a) DRIFT-ARID: A method for assessing environmental water requirements (EWRs) for non-perennial rivers. *Water SA* **42** (3) 356–367. https://doi.org/10.4314/wsa.v42i3.01

SEAMAN M, WATSON M, AVENANT M, KING J, JOUBERT A, BARKER C, ESTERHUYSE S, GRAHAM D, KEMP M, LE ROUX P and co-authors (2016b) DRIFT-ARID: Application of a method for environmental water requirements (EWRs) in a non-perennial river (Mokolo River) in South Africa. *Water SA* **42** (3) 368–383. http://dx.doi.org/10.4314/wsa.v42i3.02

STEFANIDIS K, PANAGOPOULOS Y, PSOMAS A and MIMIKOU M (2016) A methodological approach for evaluating ecological flows using ecological indicators: a case study in River Pinios, Greece. In: *1st GLOBAQUA International Conference Managing The Effects Of Multiple Stressors On Aquatic Ecosystems Under Water Scarcity.* 11–12 January 2016, Freising (Germany).

SWEETING RA (1994) River pollution. In: Calow P and Petts GE (eds.) *The Rivers Handbook, Vol. 2.* Blackwell Scientific Publications, Oxford. 23–32.

TAL A, AL KHATEEB N, NAGOUKER N, AKERMAN H, DIABAT M, NASSAR A, ANGEL R, AL SADAH MA., HERSHKOVITZ Y, GASITH A and co-author (2010) Chemical and biological monitoring in ephemeral and intermittent streams: a study of two transboundary Palestinian–Israeli watersheds. *Intl. J. River Basin Manage.* **8** (2) 185–205. https://doi.org/10.1080/15715124.2010.491796

URE AM and DAVIDSON CM (1995) *Chemical Speciation in the Environment.* Chapman & Hall, Glasgow. 408 pp.

UYS MC and O'KEEFFE JH (1997) Simple words and fuzzy zones: Early directions for temporary river research in South Africa. *Environ. Manage.* **21** (4) 517–531. https://doi.org/10.1007/s002679900047

VON SCHILLER D, ACUÑA V, GRAEBER D, MARTÍ E, RIBOT M, SABATER S, TIMONER X and TOCKNER K (2011) Contraction, fragmentation and expansion dynamics determine nutrient availability in a Mediterranean forest stream. *Aquat. Sci.* **73** (4) 485–497. https://doi.org/10.1007/s00027-011-0195-6

VON SCHILLER D, GRAEBER D, RIBOT M, TIMONER X, ACUÑA V, MARTÍ E, SABATER S and TOCKNER K (2015) Hydrological transitions drive dissolved organic matter quantity and composition in a temporary Mediterranean stream. *Biogeochemistry* **123** (3) 429–446. https://doi.org/10.1007/s10533-015-0077-4

WALKER KF (2006) The lower River Murray, Australia. Chapter 9. In: Kingsford R (ed.) *Ecology of Desert Rivers*. Cambridge University Press, Cambridge. 354 pp. 248–279.

WATSON I and BURNET AD (1993) *Hydrology: An Environmental Approach*. Buchanan Books, Fort Lauderdale, USA. 702 pp.

WILLIAMS DD (2006) *The Biology of Temporary Waters*. Oxford University Press, Oxford, New York. 337 pp.

WOELFLE-ERSKINE C, LARSEN LG and CARLSON SM (2017) Abiotic habitat thresholds for salmonid over-summer survival in intermittent streams. *Ecosphere* **8** (2) 1-23. https://doi.org/10.1002/ecs2.1645

WARD RC and ROBINSON M (2000) *Principles of Hydrology* (4th edn.). McGraw-Hill, New York. 450 pp.

YLLA I, SANPERA-CALBET I, VÁZQUEZ E, ROMANÍ AM, MUÑOZ I, BUTTURINI A and SABATER S (2010) Organic matter availability during pre-and post-drought periods in a Mediterranean stream. *Hydrobiologia* **657** (1) 217–232. https://doi.org/10.1007/s10750-010-0193-z