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Drought impacts on the water quality of freshwater systems; review and integration

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ABSTRACT

Droughts are increasing in frequency and severity in many regions of the world due to climate change. The meteorological drivers of drought often cause subsequent hydrological effects such as reduced catchment runoff, river flows and lake levels. Hydrological droughts may also result in significant changes in water quality. This review provides a synthesis of past observational research on the effects of drought on the water quality of freshwater systems (rivers, streams, lakes, reservoirs). Over the last 10–20 years there has been an increasing amount of studies on the water quality effects of drought, mostly in North America, Europe, and Australia. In general droughts, and the immediate recovery period, were found to have profound water quality effects. These effects were varied, depending on the characteristics of the water body and its catchment. Key drivers of water quality change were identified and integrated across different systems using quantitative analysis where possible. Water flow and volume decreases during drought typically led to increased salinity due to reduced dilution and concentration of mass. Temperature increases and enhanced stratification occurred during drought in some systems due to air temperature increases and longer hydraulic residence times. This also enhanced algal production, promoted toxic cyanobacterial blooms, and lowered dissolved oxygen concentrations. Nutrient, turbidity and algal levels also often increased in lake systems due to reduced flushing and enhanced productivity, and resuspension in some shallow lakes. In contrast, nutrients and turbidity often decreased during droughts in rivers and streams with no significant loading from point and agricultural non-point sources. This was due to disruption of catchment inputs and increased influence of internal processes (e.g. biological uptake of nutrients, denitrification, settling). Where point sources of pollution were present, water quality generally showed deterioration due to less dilution, particularly for nutrients. Storage and buildup of material and changed geochemistry (e.g. sulfide oxidation) in catchments during drought resulted in mobilisation of large post-drought flood loadings of constituents such as major ions, nutrients and carbon. In some cases this caused severe downstream water quality effects such as deoxygenation. Key areas for further research are process-level understanding of the key drivers of water quality change in catchments and receiving water bodies during drought, development of predictive models, and studying the resilience of systems to the predicted increase in frequency of drought and floods. The maintenance of long term water quality monitoring programmes is also critical.

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1. Introduction

Droughts are increasing in frequency and severity in many regions of the world due to increased rainfall variability and heat additions with anthropogenic climate change (Dai, 2013; Trenberth et al., 2014). The predicted future changes in climate will not be uniform and regional variations in precipitation (Trenberth et al., 2014) may result in more extreme drought–flooding hydrological patterns. Natural climatic variability also has a large influence, with severe droughts occurring globally during the El Niño phase of the El Niño/Southern Oscillation (ENSO). During droughts, the climate water deficit propagates through the hydrological cycle and can subsequently reduce groundwater levels, streamflows, and lake levels. This is often termed a hydrological drought to denote that these effects can be separated spatially and temporally from the climatic drivers of drought (Tallaksen and van Lanen, 2004).

Over the next 30–50 years most of the world's major rivers are predicted to show large increases in frequency of hydrological drought conditions relative to the historic records over the last century (Hirabayashi et al., 2008). Many lake and reservoir systems are likely to be similarly affected. Increased water extraction for consumptive purposes also increases the likelihood of hydrological droughts occurring, regardless of changing climatic factors. There is also medium confidence that anthropogenic influences are also contributing to an intensification of extreme precipitation at the global scale (IPCC, 2012), with many rivers predicted to increase their flood frequency this century (Hirabayashi et al., 2008). A transition to more extreme drought–flooding hydrological patterns could have profound consequences for freshwater ecosystems (Mulholland et al., 1997), and severe social and economic impacts.

The water quality of freshwater systems is controlled by climatic variability, hydrological, biogeochemical, and anthropogenic influences. These influences operate at various temporal and spatial (e.g. global, river basin, local catchment) scales. Droughts are a perturbation in the natural climatic and hydrologic regime which can affect the determinants of water quality in multiple ways. For example, low flows and water levels observed during hydrological droughts increase the residence time and reduce the flushing rate of water bodies. Reduced water flows/levels and elevated temperatures during some droughts may change the rates of processes such as productivity, respiration, and reaeration. Droughts may also change the delivery pattern of water quality constituents, retaining them in catchments during dry conditions, releasing them during wet conditions (Worrall and Burt, 2008). At the local scale, water quality effects of drought and floods can be quite variable and specific depending on the biophysical characteristics of water bodies and their catchments (Caruso, 2002; Sprague, 2005).

There have been several observational studies at local and regional scales on the water quality impacts of drought. Given the predictions of increasing frequency and severity of droughts in freshwater systems, it was considered beneficial to undertake a review and synthesis of the relevant literature on this topic. A previous review by Delpla et al. (2009) assessed the impacts of climate change on surface water quality in relation to drinking water production. Some drought-related studies were included, but wider non-drinking water impacts were not assessed. Similarly, Murdoch et al. (2000) and Whitehead et al. (2009) reviewed the wider potential effects of climate change on surface water quality and ecology. These studies include some coverage of drought impacts on water quality, but a more specific review is warranted given the likelihood of increasing drought risks.

The aim of this paper is to review and synthesise findings on the impacts of drought on the water quality of freshwater systems (rivers, streams, lakes, reservoirs). The intent is that this information will help enable improved recognition, assessment and management of the impacts of drought on water quality.

2. Methods

Three large scientific databases (Scopus, Science Direct, Web of Science) were queried for key words e.g. “drought” and “water quality” in title, abstract, or keywords. The search of Scopus returned 2192 results while Science Direct returned 398 results. Using “drought” as a topic & “water quality” as a title in the Web of Science returned an additional 365 results. The abstracts were scanned online to check relevance. Only observational (field-based) studies conducted in rivers, streams, lakes and reservoirs were assessed with estuary, wetland and groundwater-specific studies excluded. Publications as conference proceedings and reports were also excluded. Additional studies were reviewed that were obtained from reference lists in the selected publications and/or were relevant to understanding the observed water quality changes during drought.

Due to the wide range of methods used in the different studies, it was difficult to systematically categorise different droughts on the basis of meteorological or hydrological parameters. In general there is no universally agreed method available to define drought (Tallaksen and van Lanen, 2004; Mishra and Singh, 2010; IPCC, 2012). Hence we have accepted that the water quality outcomes described in the various studies were during hydrological droughts involving anomalies from normal conditions. Where the severity or duration of a particular drought is particularly relevant, we discuss it in the context of drivers of water quality changes. Water quality in the immediate post-drought recovery phase is also discussed where relevant.

The next section (Section 3) of the review details the temporal and spatial distributions of drought water quality studies. Changes in key water quality parameters (temperature, salinity/conductivity, nutrients, algae/chlorophyll, dissolved oxygen, turbidity/suspended solids, pH, bacteria, metals and organic carbon) are then discussed separately for rivers/streams and lake/reservoirs. Section 4 synthesises and integrates the study findings, using a quantitative framework where possible, to find common themes and processes that lead to particular water quality outcomes. The final Section 5 of the review details recommendations for future research, monitoring and assessment.

3. Results

3.1. Spatial and temporal distributions of drought water quality impacts

The majority of studies on the water quality impacts of drought have been undertaken in North America, Europe and Australia (Fig. 1 and list of studies in Supplementary material S1). However, given the distribution of meteorological droughts relative to the world's river systems (see Fig. 1), it appears likely that many impacts, particularly in Asia, Africa and South America, have not been documented in the published literature.

The temporal distribution of studies documenting the water quality impacts of drought is shown in Fig. 2. The first study identified on the impacts of drought on water quality was that of Armstrong et al. (1931) in the eastern U.S.A. Studies appear sparse in the published journal literature until the 1970s, and then a substantial increase in the number of studies occurs between 1990 and 2010 (Fig. 2). This increase likely reflects an increased global awareness and focus on climatic and hydrological changes in freshwater systems, possibly also the increasing incidence of droughts (Dai, 2013).

3.2. Rivers/streams

3.2.1. Temperature

Temperature increases in rivers and streams during drought have been reported in many studies (Fig. 3, Davies, 1978; Boulton and Lake, 1992; Caruso, 2002; van Vliet and Zwolsman, 2008; Ziellnski et al., 2009; Hrdinka et al., 2012). van Vliet and Zwolsman (2008) recorded median temperature increases of 2 °C in the Meuse River, while in the same drought, temperature in some Czech Republic streams increased 1.7 °C (Hrdinka et al., 2012), and an average of 1.3 °C in some Polish streams (Ziellnski et al., 2009). Ha et al. (1999) observed a very large (7 °C) increase in water temperature in the regulated lower Nakdong River (South Korea) during a drought. In studies that also reported meteorological parameters, increased water temperature during droughts was correlated with higher air temperatures (van Vliet and Zwolsman, 2008; Ziellnski et al., 2009; Hrdinka et al., 2012).

In contrast, Wilbers et al. (2009) did not find a significant temperature increase during a drought on the Dommel River (Netherlands) which may have been due to a large proportion of flow being derived

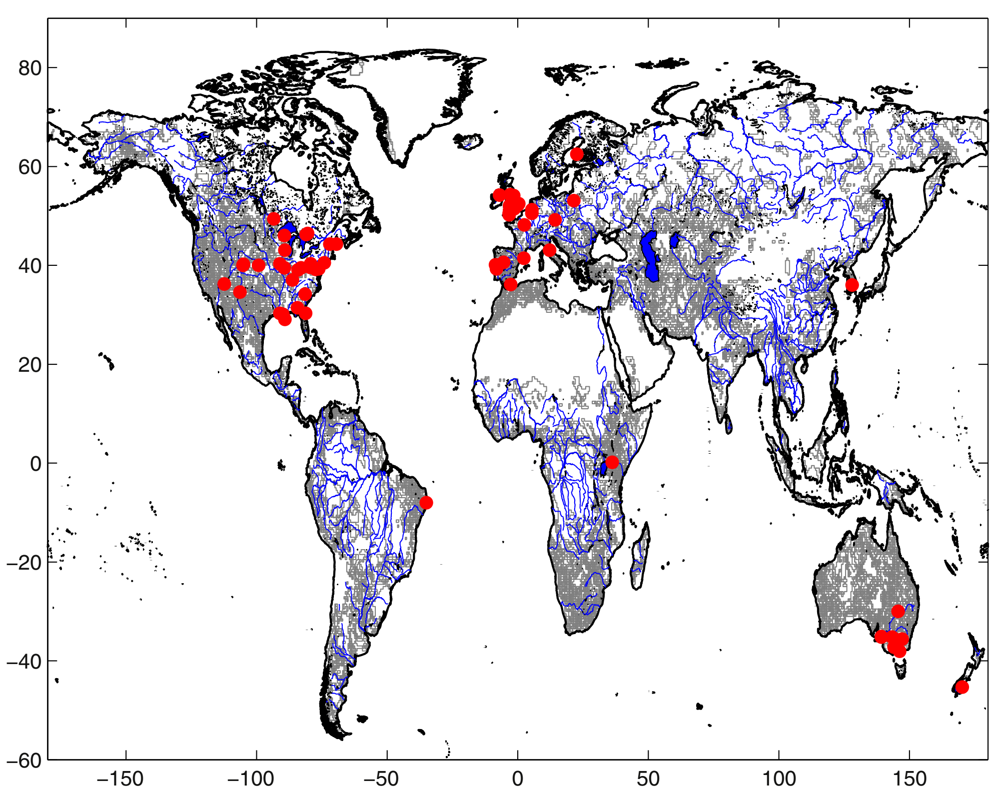

Fig. 1. Spatial distribution of studies (red dots) documenting the water quality impact of droughts, along with the occurrence of meteorological droughts (grey-shaded area) based on precipitation records (1980–2001) from the global risk data platform (see http://preview.grid.unep.ch/). The major rivers/streams and lakes/reservoirs of the world are also shown (blue lines/polygons) based on the global self-consistent, hierarchical, high-resolution geography database (see http://www.soest.hawaii.edu/pwessel/gshhg/).

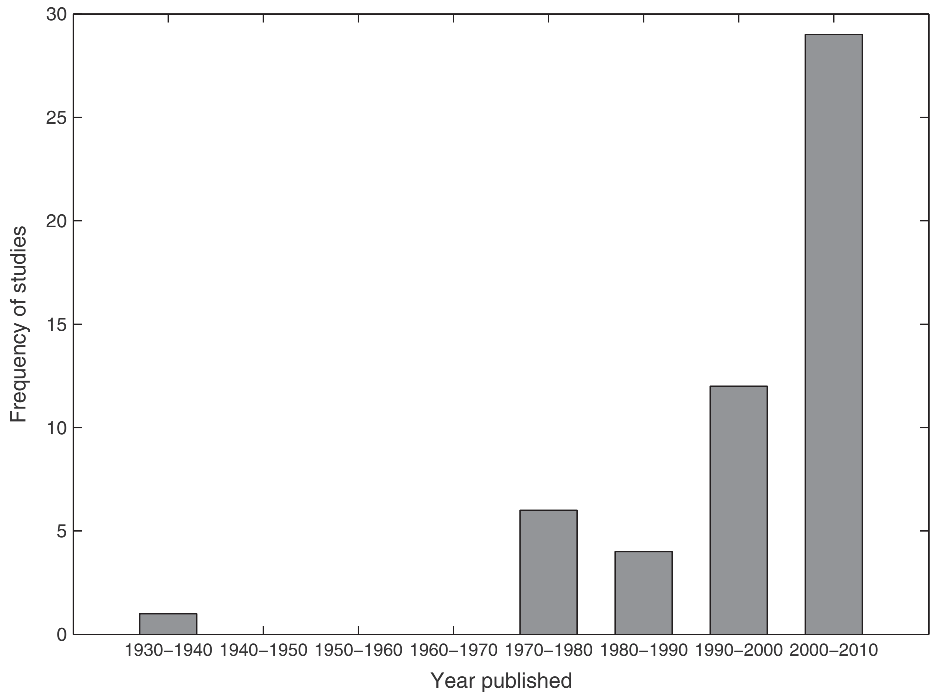

Fig. 2. Frequency of studies on the water quality impacts of droughts, 1930–2010.

from deeper groundwater sources. Mosley et al. (2012) also did not observe any significant increase of water temperature in the lower River Murray (Australia) during extreme low flows, which was attributed to local air temperature not increasing. Streams in forested catchments in the South Platte River Basin (U.S.A.) increased in temperature whereas streams in urban and agricultural catchments showed little change (Sprague, 2005). This was attributed to more efficient heating of the smaller headwater streams in the forested areas, and dilution from cooler reservoir inputs, point sources, and/or groundwater downstream.

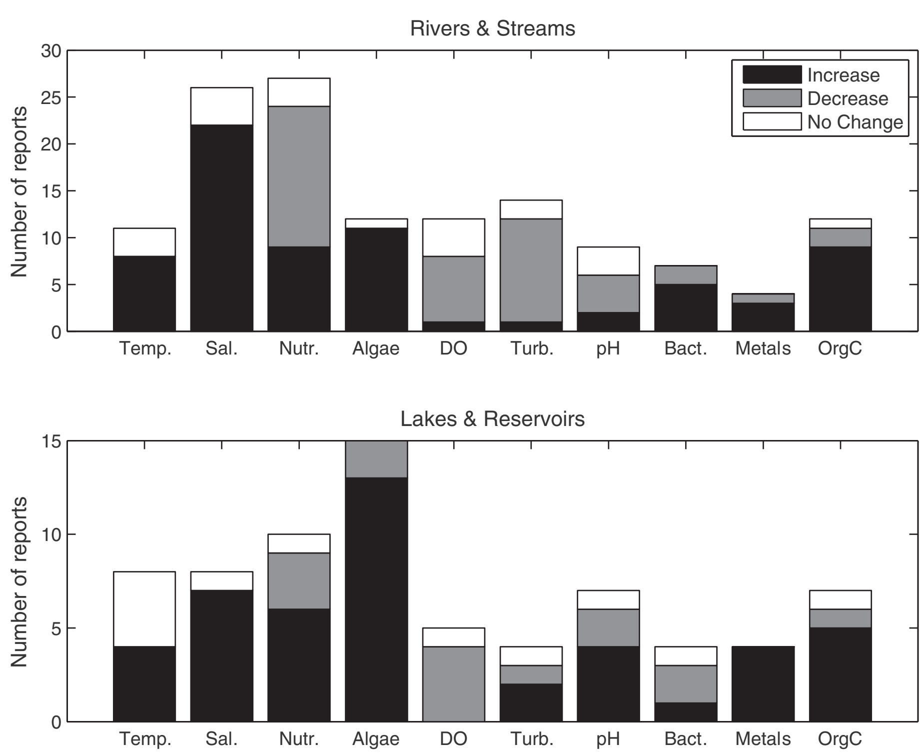

Fig. 3. Summary of drought water quality changes (increase, decrease, no notable change) for temperature (Temp.), salinity (Sal.), nutrients (Nutr.), algae, dissolved oxygen (DO), turbidity (Turb.), pH, bacteria (Bact.), metals, and organic carbon (OrgC). Note, for most parameters an increase in concentration is usually considered detrimental but the exception to this is for pH and DO where a decrease is usually considered detrimental.

3.2.2. Salinity, major ions

Salinity has been shown to increase in most streams and rivers during droughts (Fig. 3). This has been attributed to evapoconcentration and less dilution of more saline groundwater inputs (Foster and Walling, 1978; Muchmore and Dziegielewski, 1983; Caruso, 2001, 2002; van Vliet and Zwolsman, 2008; Mayer et al., 2010; Hrdinka et al., 2012; Mosley et al., 2012; Burt et al., 2014a), and in some cases point sources (Davies, 1978; Chessman and Robinson, 1987; van Vliet and Zwolsman, 2008). Penetration of saline wedges up river systems from estuaries and saline lakes during drought has also been reported (Ingram et al., 1996; McAnally and Pritchard, 1997; Mosley et al., 2012). Where groundwater and non-saline point sources are a significant source of river flow, insignificant changes in salinity have been noted during drought (Wilbers et al., 2009). Mast et al. (2013) observed increasing stream sulfate during drought at several stations that was likely due to pyrite oxidation in the catchment.

Post-drought, complex patterns of salinity and individual solute (e.g. calcium, magnesium, potassium, chloride) flushing have been observed that relate to accumulation, hydrological pathways, and sediment-solution characteristics at particular sites (Anderson and Burt, 1978; Foster and Walling, 1978; Stott and Burt, 1997; Burt et al., 2014a). These effects were short-lived in some cases (Foster and Walling, 1978).

3.2.3. Nutrients

Nutrient concentrations in river and streams during droughts show mixed responses (Fig. 3). Lower dissolved and total nutrient concentrations have been observed during droughts in many rivers and streams for nitrogen and phosphorus (Boar et al., 1995; Schindler et al., 1996; Morecroft et al., 2000; Caruso, 2001, 2002; Golladay and Battle, 2002; Oelsner et al., 2007; Ziellnski et al., 2009; Mosley et al., 2012; Baurès et al., 2013) and silica (Oborne et al., 1980; Hunt et al., 2005). This has mainly been attributed to a reduction or lack of catchment inputs, but also increased in stream retention via uptake of dissolved nutrients by aquatic algae and macrophytes (Boar et al., 1995; Andersen et al., 2004; Baurès et al., 2013), and increased denitrification due to longer water residence times (van Vliet and Zwolsman, 2008).

In contrast, increasing nutrient concentrations during droughts has occurred in rivers/streams where point sources of nutrients (industrial, domestic, agricultural wastewater discharges) are present (and remain relatively constant during drought) and a lack of dilution occurs (Anderson and Faust, 1972; Davies, 1978; Oborne et al., 1980; Boar et al., 1995; Caruso, 2002; Jarvie et al., 2003; Andersen et al., 2004; Sprague, 2005; van Vliet and Zwolsman, 2008; Macintosh et al., 2011; Hrdinka et al., 2012). Some studies have found higher nitrate concentrations in some agricultural catchments at low flows, which were linked to reduced dilution of groundwater drainage input (Foster and Walling, 1978; Golladay and Battle, 2002; Sprague, 2005; Jarvie et al., 2003), and increased influence of sediment nitrogen fluxes (van Vliet and Zwolsman, 2008).

Mixed and counteracting drought responses between different nutrients were relatively common. Muchmore and Dziegielewski (1983) found that ammonia from effluent sources increased significantly in the Sangamon River (U.S.A.) during a drought while nitrate and nitrite decreased due to reduced agricultural runoff and drainage. Sprague (2005) found that in urban and agricultural catchments, fertiliser application and point source discharges likely contributed to increased nitrate and nitrite concentrations but ammonia, and dissolved and total phosphorus were not markedly higher or lower during the drought in any land use area. Wilbers et al. (2009) found that nutrient concentrations remained similar in drought compared to non-drought conditions on the River Dommel (Netherlands). This was attributed to counteracting effects where in wet periods shallow groundwater flow from agricultural soils is the main source of nutrients to the surface water whereas during dry periods deep groundwater inflow and wastewater contributions become more significant. Ziellnski et al. (2009) found that dissolved nutrients showed little change but TP increased. Changing nutrient ratios have also been observed during droughts (Hunt et al., 2005; Mosley et al., 2012; Baurès et al., 2013).

Post-drought, several studies have found that nitrate was flushed from the catchment soils in the immediate post-drought period (Foster and Walling, 1978; Oborne et al., 1980; Morecroft et al., 2000; Jarvie et al., 2003; Burt and Worrall, 2009). This is likely both due to concentration in the catchment during the drought phase and stimulation of mineralisation and nitrification when dried soils are rewet (Morecroft et al., 2000).

3.2.4. Algae

There were several reports of increased water column and benthic algal levels in rivers and streams during drought (Fig. 3). García-Prieto et al. (2012) found much higher total algal values (chlorophytes and cyanophytes dominate) during drought in a Spanish river that was attributed to reduced water volumes and high temperatures. van Vliet and Zwolsman (2008) also found that large chlorophyll-*a* increases occurred during drought at one more stagnant site in the Meuse River (Netherlands), but another site with higher flows showed no significant increase. Boar et al. (1995) found that chlorophyll-*a* in the water column did not increase during drought in the River Nar (U.K.).

In some studies there was a shift to more toxic cyanobacterial species during drought. Donnelly et al. (1997) found that an extensive (>1000 km) algal (*Anabaena circinalis*) bloom on the Darling–Barwon River (Australia) was driven by low river flow, water column stratification and clarification, and phosphorus release from the anoxic sulfate-reducing sediments. Ha et al. (1999) reported a large and sustained cyanobacterial bloom (*Microcystis* sp.) in the lower Nakdong River (South Korea) that was attributed to increased irradiance and water temperature during drought. Bowling et al. (2013) documented an extensive (>1000 km) potentially toxic cyanobacterial (predominantly *A. circinalis*) bloom during a sustained drought on the River Murray (Australia). Temperature and nutrient conditions in the low flows were also conducive to sustaining the bloom as it was transported downstream over several weeks.

Increased benthic algal production during some droughts has also been observed. Caruso (2001, 2002) observed benthic algae blooms during drought, in many cases covering 50–80% of the stream bed, particularly in lowland agricultural catchments with low water velocities and minimal shade. Caramujo et al. (2008) also found that chlorophyll-*a* in river biofilms increased during drought.

3.2.5. Dissolved oxygen

Dissolved oxygen has showed mixed responses during droughts in river and streams (Fig. 3). Several studies found little change in dissolved oxygen levels during droughts, mostly in shallow streams with good reaeration (Hudson et al., 1978; Caruso, 2001, 2002; Hrdinka et al., 2012). Some studies have found dissolved oxygen levels increased during daytime in drought due to enhanced primary production (Ha et al., 1999; Sprague, 2005; van Vliet and Zwolsman, 2008).

Declines in dissolved oxygen have been recorded in some studies, particularly where point sources are present (Anderson and Faust, 1972; Chessman and Robinson, 1987) and where water temperatures have been noted to increase (Boulton and Lake, 1992; Sprague, 2005; van Vliet and Zwolsman, 2008; Ylla et al., 2010). Post-drought, there have also been reports of severe water deoxygenation due to the delivery of large amounts of labile carbon that had been stored in the catchment during dry conditions (Davies, 1978; Whitworth et al., 2012).

3.2.6. Turbidity/suspended sediment

Decreased turbidity has been observed during droughts in many rivers and streams (Fig. 3, Tunnicliff and Brickler, 1984; Caruso, 2001, 2002; Golladay and Battle, 2002; van Vliet and Zwolsman, 2008; Hrdinka et al., 2012; Mosley et al., 2012). This has been attributed to lack of catchment runoff and increased sedimentation due to lower water velocities. However turbidity can also increase during droughts when a significant portion of the suspended load is derived from point sources (Anderson and Faust, 1972; Caruso, 2002).

3.2.7. pH

pH has shown mixed responses during drought (Fig. 3). Caruso (2001, 2002) found that pH values were not significantly different in several streams during a drought, compared to the preceding reference period. Minor (0.2–0.4 pH unit) but statistically significant decreases were observed in Polish streams (Ziellnski et al., 2009) and the River Murray (Mosley et al., 2012) during droughts. Österholm and Åström (2008) observed a persistent decrease in stream pH for multiple years following a drought which resulted in lowered water tables and sulfide oxidation in a Finnish catchment. Increases in pH and alkalinity have been observed in some river systems during drought due to decreased dilution of bicarbonate-dominated groundwater (Sprague, 2005; Ziellnski et al., 2009).

3.2.8. Organic carbon

Increases in organic carbon have been noted in several studies in river and streams during drought (Fig. 3). The early study of Armstrong et al. (1931) highlighted many issues with organic carbon, taste and odour compounds relating to drinking water supply quality. Baurès et al. (2013) found that total organic carbon concentrations during very low flows were higher than during low–moderate flows in several French rivers, and this was attributed to aquatic biomass production.

In contrast, some studies have noted decreasing dissolved organic carbon (DOC) concentrations during droughts. For example, Ylla et al. (2010) found that biodegradable DOC and its quality as a food source decreased to low levels as a stream dried in a drought. Ziellnski et al. (2009) also found that DOC levels during a drought in some Polish streams were about 60% pre-drought levels. These findings likely reflect reduced catchment organic matter inputs during drought.

Post-drought, significant organic carbon mobilisation from catchments has also been reported. Flooding (first in over a decade of drought) of both forested and agricultural floodplains in the Murray–Darling Basin (Australia) mobilised large stores of reactive DOC leading to low oxygen levels throughout >2000 km of river system (Whitworth et al., 2012). Ylla et al. (2010) reported similar trends in reactive/biodegradable DOC export after a shorter (few months) duration drying in a small Mediterranean stream but oxygen levels remained high. A review of long term data in the U.K. by Worrall and Burt (2008) found that increasing DOC concentrations post-drought was more controlled by changes in hydrological throughput rather than additional production of DOC during drought.

3.2.9. Bacteria

Increased faecal coliform levels in some streams during droughts (Fig. 3) have occurred due to greater stock use of waterways and the lack of dilution and flushing flows (Caruso, 2001, 2002), and increased influence of point sources (Armstrong et al., 1931). In contrast, faecal coliforms were lower during drought in the Colorado (Tunnicliff and Brickler, 1984) and Sangamon (Muchmore and Dziegielewski, 1983) rivers (U.S.A.) during droughts. The reasons for this were somewhat unclear but may been have related to reduced agricultural runoff and drainage. Hrdinka et al. (2012) also found low faecal bacteria concentrations during drought in two Czech Republic rivers but in a flood at the end of the drought concentrations increased significantly. Tunnicliff and Brickler (1984) and Shehane et al. (2005) found that decreases in wet-weather inputs during droughts resulted in faecal coliforms from wildlife sources dominating compared with human sources pre-drought.

3.2.10. Metals and metalloids

There have been relatively few studies of metal behaviour in streams and rivers during droughts (Fig. 3). van Vliet and Zwolsman (2008) found that total selenium, barium and nickel increased during drought but lead, chrome, mercury, and cadmium showed significantly lower concentrations. This difference was attributed to varying adsorption capacities to suspended solids which determined the fate of the metals. The concentrations of all metals remained low during a drought in the Czech Republic, but markedly increased (1760%) due to sediment runoff in the post-drought period (Hrdinka et al., 2012). Hudson et al. (1978) also found that most metal concentrations did not increase significantly during drought, apart from manganese which approximately doubled. Muchmore and Dziegielewski (1983) also found that manganese increased significantly in the Sangamon River during a drought, likely due to an increased influence of point-source wastewater inputs.

3.3. Lakes/reservoirs

3.3.1. Temperature

Temperature has shown mixed responses in lakes during droughts (Fig. 3). Some studies have recorded temperature increases and more intensive stratification during drought (Baldwin et al., 2008; Flanagan et al., 2009). This appears to be driven by increasing air temperatures and/or temperatures of inflows during drought, and reducing water volume. In contrast, Olds et al. (2011) and Mosley et al. (2012) did not report a drought temperature increase despite lake levels in their studies reducing greatly in volume. This was presumed to be due to the local air temperature not increasing. Similarly, García-Jurado et al. (2012) did not find a temperature increase in an alpine lake during a drought characterised by low precipitation and water levels.

3.3.2. Salinity, major ions

Salinity has been observed to increase in most lakes and reservoirs during droughts (Fig. 3). This has been attributed to reduced flushing/outflows and evapoconcentration (Flanagan et al., 2009; Mosley et al., 2012; Jirsa et al., 2013). In contrast, Barros et al. (1995) found that despite significant reductions of water levels in reservoirs, salinity increases were relatively minor and not significantly correlated with reservoir volume. The reasons for this were unclear and there may have been salt export from the reservoirs during the drought, or potential changes in catchment salt inputs. Kratz et al. (1997) found that lakes high in the landscape decreased in calcium mass during drought, while lakes low in the landscape increased in mass of calcium due to a relatively higher groundwater input.

3.3.3. Nutrients

Lakes and reservoirs have shown mixed responses in their nutrient concentrations during droughts (Fig. 3). Large increases in total nutrient concentrations have been observed during droughts associated with large water level declines, although dissolved nutrients can remain at low concentrations due to rapid uptake by algae (Bouvy et al., 2003; Mosley et al., 2012; Jirsa et al., 2013). Increased atmospheric dust deposition during droughts was also observed to increase total phosphorus concentrations in an alpine lake (García-Jurado et al., 2012). In contrast, Barros et al. (1995) found no significant increase in nutrient levels despite water level decreases in Portugal reservoirs during a drought. They attributed this to reduced diffuse pollutant input from the catchments which counteracted the concentration effect. Lathrop (2007) also found that reduced phosphorus loadings during drought reduced lake total phosphorus and algal concentrations.

Changed internal processing has been identified as contributing to some of the nutrient trends during drought in lakes and reservoirs. Ammonia, organic nitrogen and total phosphorus have been observed to increase in the bottom water (hypolimnion) of systems which have stratified and shown reduced oxygen levels (Bouvy et al., 2003; Baldwin et al., 2008). Shallowing of lakes can also lead to more turnover of nutrient-laden bottom water (Baldwin et al., 2008), and resuspension (Mosley et al., 2012), potentially enhancing primary productivity. Dissolved nutrient ratios have also been observed to change during drought which could influence primary production via nutrient limitation (Hunt et al., 2005; García-Jurado et al., 2012; Mosley et al., 2012). Ruley and Rusch (2002) found that growth of filamentous algae was likely limited by light during drought due to the high amount of tannins/colour that developed in City Park Lake (U.S.A.).

3.3.4. Algae

A majority of studies reported significantly elevated chlorophyll-*a* and algal species concentrations during droughts in lake and reservoirs (Fig. 3). This has been attributed to more favourable hydrodynamic conditions for algal growth, concentration of algal biomass during water level declines, and changed nutrient dynamics (Naselli-Flores, 2003; Olds et al., 2011; Mosley et al., 2012; Reynolds et al., 2012). There have also been observations of shifts to more toxic cyanobacterial species during droughts. Davies (1978) reported large algal blooms (*Aphanizomenon*, *Stephanodiscus*, *Microcystis* sp.) from several water storage reservoirs in the Anglian region of the U.K. in 1975–1976. Baldwin et al. (2008; 2010) detected an increase in algae and shift to more toxic cyanobacterial species (principally *A. circinalis*) under an extreme drawdown situation in Lake Hume reservoir (Australia). The toxic algae were exported from the reservoir a substantial distance downstream in a river. Bouvy et al. (2003) found that the decrease in the reservoir storage volume led to a significant increase in chlorophyll-*a* concentrations and a shift in population structure to toxic cyanobacterial species (*Cylindrospermopsis raciborskii*). de Figueiredo et al. (2012) found that drought conditions in Portugal reservoirs resulted in dominance of cyanobacteria and actinobacteria. Flanagan et al. (2009) found that a large shift occurred in the phytoplankton community composition and biomass in response to an extreme drought in alpine lakes in Colorado, USA. This response was most correlated with water quality changes, rather than temperature and hydraulic residence time. The immediate post-drought period also showed large community shifts related to atmospheric nitrogen deposition.

In contrast, Caramujo et al. (2008) found that chlorophyll-*a* in reservoir biofilms decreased during a drought, which was presumed to be due to lowered nutrient inputs. Despite elevated nutrients during a drought, increased grazing pressure was found to keep algal levels similar to pre- and post-drought levels in a Spanish alpine lake (García-Jurado et al., 2012).

3.3.5. Dissolved oxygen

Reduced dissolved oxygen has been observed in some drought studies in lakes and reservoirs, although surprisingly the number of studies measuring this parameter was low (Fig. 3). Increased water column stratification and settling and breakdown of organic material in reservoirs during drought have resulted in very low dissolved oxygen concentrations in the hypolimnion (Naselli-Flores, 2003; Baldwin et al., 2008), which also extended to surface waters when one reservoir refilled (Bouvy et al., 2003). Olds et al. (2011) also found reduced oxygen concentrations throughout the whole water column of a reservoir which was also linked to demand linked to the breakdown of algae that increased substantially in the drought, and reduced fetch during the low water levels that decreased wind-driven reaeration.

3.3.6. Turbidity/suspended sediment

In some shallow lakes and reservoirs, turbidity markedly increased during droughts due to increased sediment resuspension and concentration of colloidal material (Ludovisi and Gaino, 2010; Olds et al., 2011; Mosley et al., 2012). However in some deeper lakes, reduced nutrient loadings during drought have led to greater water clarity (Lathrop, 2007). Illinois (U.S.A.) water supplies were noted to show on average a 50% reduction in turbidity in the 1930 drought (Armstrong et al., 1931).

3.3.7. pH

pH has shown mixed responses in lakes during drought (Fig. 3). Mosley et al. (2012) only found a minor pH increase at one site that experienced seawater intrusion during drought while other sites remained unchanged despite large salinity and productivity changes. Lake margins where sulfide-containing sediments have been exposed have shown severe acidification ($pH < 5$) during drought (Yan et al., 1996; Mosley et al., 2014). In acidified lakes in Canada, significant improvements in pH and alkalinity have been observed during drought years but rapid deterioration followed in the subsequent wet year due to flushing of the sulfide-oxidation products (Mallory et al., 1998). Ludovisi and Gaino (2010) observed a general trend of increasing pH and alkalinity in long term data in Lake Trasimeno (Italy) during a prolonged hydrological drought due to changed ionic inputs and lowered water levels.

3.3.8. Organic carbon

Increases in organic carbon concentration have occurred in many lakes/reservoirs during droughts which have been linked to higher productivity rates and concentration due to reductions in water volume (Fig. 3, Bouvy et al., 2003; Jirsa et al., 2013) and in some cases point source pollution (Armstrong et al., 1931; Davies, 1978). In contrast, Watts et al. (2001) analysed a long term time series of colour measurements in reservoirs in peaty-type catchments and found generally low levels of colour during drought, followed by in some cases unprecedented high and sustained levels post-drought. The colour was thought to be released from the peat by enhanced bacterial decomposition of organic material during more aerobic drought conditions leading to increased availability of humic and fulvic acids that are subsequently washed out of the soil post-drought. Similar post-drought soluble carbon mobilisation trends were observed from Ylla et al. (2010). However, as noted above some of these observed effects may relate more to hydrological processes than increased DOC mineralisation during drought (Worrall and Burt, 2008).

Yan et al. (1996) reported a DOC decrease during drought, due to acidification and flocculation as a result of dissolved aluminium complexation. This greatly increased UV-B penetration into the water column.

3.3.9. Bacteria

Barbé et al. (2001) found lower faecal coliform concentrations in Lake Pontchartrain (U.S.A.) during a drought that was likely due to reduced urban catchment runoff due to the lower rainfall. Bacteria concentrations in Kentucky (U.S.A.) reservoirs were noted to increase in the post-drought period in the study of Armstrong et al. (1931).

3.3.10. Metals and metalloids

Dissolved concentrations of redox-sensitive metals (typically Fe and Mn) have been observed to increase in the bottom water of reservoirs which have stratified and shown reduced oxygen levels (Baldwin et al., 2008). Jirsa et al. (2013) found that evapoconcentration during droughts was observed to increase concentrations of some metals (As, Mo), while Fe showed a large and unexplained decrease midway through the drought. Mosley et al. (2014) observed large scale release of dissolved metals on acidified lake margins when they were rewet by rainfall and post-drought water level rise.

4. Discussion

It is evident that hydrological drought results in complex and varied water quality effects which operate at various temporal and spatial scales. Nevertheless there were common findings in many

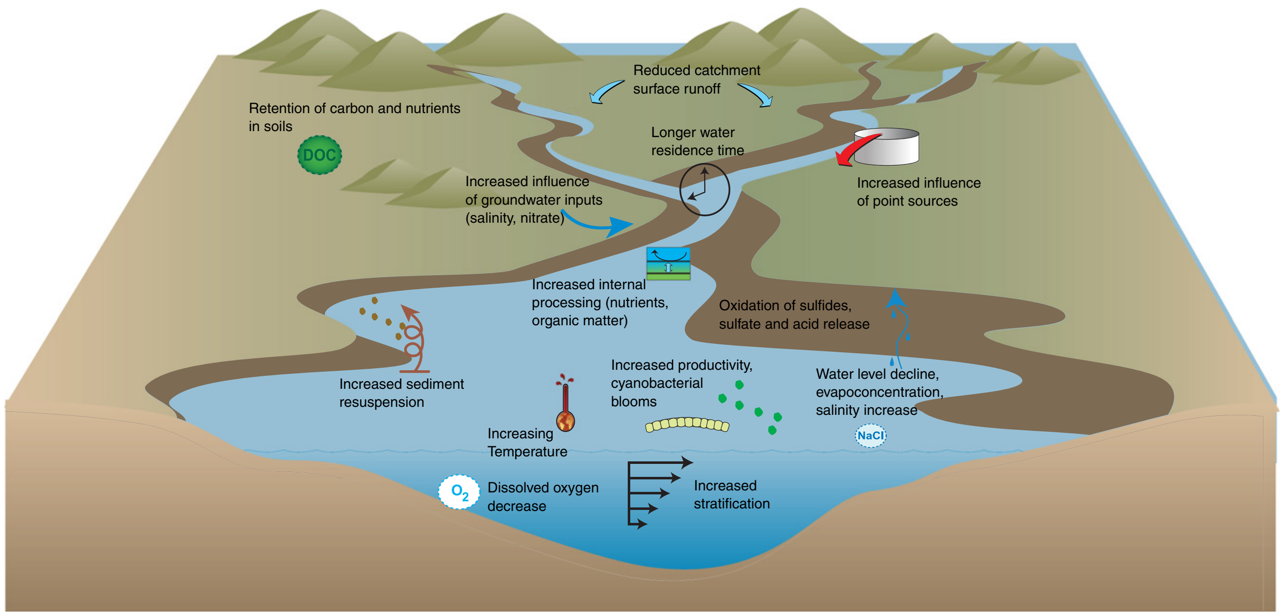

Fig. 4. Conceptual diagram summarising common processes that determine water quality outcomes during drought.

studies that enabled integration and determination of key processes driving water quality outcomes during drought; (1) hydrological drivers and mass balance, (2) role of temperature and stratification, and (3) increased influence of internal processes and resuspension. A conceptual model of the processes found to affect water quality during drought is shown in Fig. 4. The relative importance of these processes, and how their interactions determine water quality outcomes in freshwater systems, are discussed and where possible placed in a quantitative framework (as per methods outlined in Chapra, 1997) below.

4.1. Hydrological drivers and mass balance

Hydrological drivers have direct influences on determining water quality outcomes during drought. Droughts tend to reduce both inflows to and outflows from freshwater systems, increasing hydraulic residence time, and in many cases result in water level (system volume) reductions. At an individual system level the relative proportion of external loadings and how they change during drought is also critical to determining water quality outcomes (Jarvie et al., 2003). The drivers of water quality change during drought can be explored by defining the concentration (c) of a particular water quality parameter in a well-mixed system at steady state by Chapra (1997):

$$c = \frac{W}{Q + kV + vAs} \tag{1}$$

where W is the total mass (M/T) loading from various sources (e.g. $W_{\text{inflow}} + W_{\text{surface runoff}} + W_{\text{groundwater}} + W_{\text{point sources}}$), Q = outflow, losses by biological or chemical reactions are represented by a first order reaction coefficient, k (1/T) multiplied by the system volume, V (L^3), and losses due to settling represented by the apparent settling velocity, v (1/T), multiplied by the surface area of the sediment, A_s (L^2). Concentration is predicted to increase if inflows/outflows and system volume reduce during drought, while loading is constant or proportionally decreases less than the flow. In contrast concentration is predicted to decrease if loadings reduce, and flow and/or volume proportionally decrease less than the loadings.

The predictions of Eq. (1) were generally consistent with the review findings (Section 3). Most water bodies showed concentration increases during drought that could be attributed to reductions in flow and/or volume. Firstly this was particularly apparent in rivers and streams where groundwater provided a significant and more constant loading and concentrations of associated parameters (salinity and nitrate) increased during drought (Jarvie et al., 2003; Sprague, 2005; Mosley et al., 2012). Secondly, where point sources made a significant contribution to total loadings in rivers (and remained relatively constant during drought but dilution reduced) water quality typically deteriorated. Thirdly, the increased retention and concentration of mass in lakes, due to a lack of outflow/flushing and volume reductions, also led to poor water quality (e.g. increasing salinity, total nitrogen and phosphorus) and an increase in the severity of eutrophication (Dillion, 1975; Mosley et al., 2012).

In contrast, in more surface flow dominated rivers and streams, nutrient and turbidity concentrations often reduced in drought as catchment loadings were disrupted. However large loads were often delivered when substantial flows and catchment runoff returned post-drought and delivered material that had accumulated in the landscape during the drought. The large catchment loadings (Eq. (1)) caused concentrations of certain parameters, in particularly organic carbon, nutrients and turbidity to increase. In some cases post-drought mobilisation of this material resulted in extreme water quality effects such as deoxygenation (Whitworth et al., 2012). It is currently unclear how important the length of the drought period is to the severity of these “first flush” loadings and timescale of system recovery. Reynolds et al. (2012) found that increased flooding may not necessarily reset water quality in some lake systems following drought, as nutrients and other material can be stored in the hypoliminion and other poorly-flushed zones. Systems that are inherently poorly flushed, such as many terminal lake systems, are likely to be particularly vulnerable.

4.2. Role of temperature and stratification

In terms of direct climatic effects, many studies found that water temperature increases during drought but this was not universal. The heat budget of water bodies is determined by thermal flux across the air:water interface, advection of heat via tributary or other inflows, and losses via outflows (Imberger, 1985). The thermal flux parameter appears particularly important as increased drought water temperatures occurred when local air temperature increases were also recorded

Table 1
Air temperature versus water temperature change (Δ, °C) during droughts (ns = no significant change).

Study	Type	Description	Δ Air Temp.	Δ Water Temp.
van Vliet and Zwolsman (2008)	River	Mean summer 1976	1.4	2
	River	Mean summer 2003	1.9	2
Ziellnski et al. (2009)	River	Average all sites	1.8	1.5
Hrdinka et al. (2012)	River	Average all sites	2.5	1.7
Mosley et al. (2012)	River	Median	ns	ns
Flanagan et al. (2009)	Lake	Median	1.8	2.1

(Table 1). This finding is consistent with previous research that has shown a strong correlation between air temperature and water temperature (McCombie, 1959; Johnson et al., 2014). The average 1–2 °C degree magnitude of temperature change observed during droughts (Table 1) is at the lower end of the range of overall global mean temperature increase that is predicted to occur due to climate change (1.5–4 °C high confidence value, IPCC, 2013). Lake temperature increases due to global climatic warming have already been reported (Schindler et al., 1996; Coats et al., 2006; Schneider et al., 2009). Hence temperature effects on water bodies during droughts are likely to increase in the future to the rising global baseline air temperature.

Another commonly observed drought effect related to temperature was an increased incidence and persistence of stratification and harmful algal blooms. Stratification is promoted by conditions often arising during droughts, in particular lower flows and increased thermal energy (Bormans et al., 1997). Many harmful algal species (e.g. *Anabaena* sp.) can regulate their buoyancy to enable them to obtain nutrients in the hypoliminion under stratified conditions, and to move into the epiliminion to photosynthesise (Paerl and Paul, 2012). This capability and additional aspects of cyanobacteria (salt tolerance, atmospheric nitrogen and carbon dioxide fixation ability) may promote their dominance during drought (Paerl and Paul, 2012). There is also a strong positive relationship between water temperature and cyanobacterial growth rate that persists up to very high temperatures (>35 °C, Fig. 5 top, and see Chapra, 1997 and Supplementary material S2 for model description). Hence increased water temperatures during drought can promote rapid bloom formation (Ha et al., 1999) due to the exponential nature of algal cell growth (as shown in Fig. 5 bottom, ignoring nutrient or other limitations). Hence due to the predicted continued increases in global temperatures, many systems will likely experience increased incidence and severity of harmful algal blooms in future droughts, including those which have not had such issues previously.

Temperature increases and increased stratification also promoted lower dissolved oxygen in a number of studies, particularly in the hypolimnion of lakes and reservoirs. Temperature increases directly result in a lower dissolved oxygen saturation concentration, while lower and less turbulent flows result in a reduced overall reaeration rate with the atmosphere (Fig. 6). Proportionally greater reductions in reaeration rates are predicted to occur in shallow (<5 m deep) systems. There may also be a higher amount of organic matter (due to increased deposition and algal production) and supply of organic matter to the sediments during drought (due to increased deposition) that exerts an oxygen demand. Higher temperatures during drought increase respiration and other rates which may result in increased decomposition of this organic matter and higher sediment oxygen demand (rates approximately doubling for a temperature rise of 10 °C, Chapra, 1997).

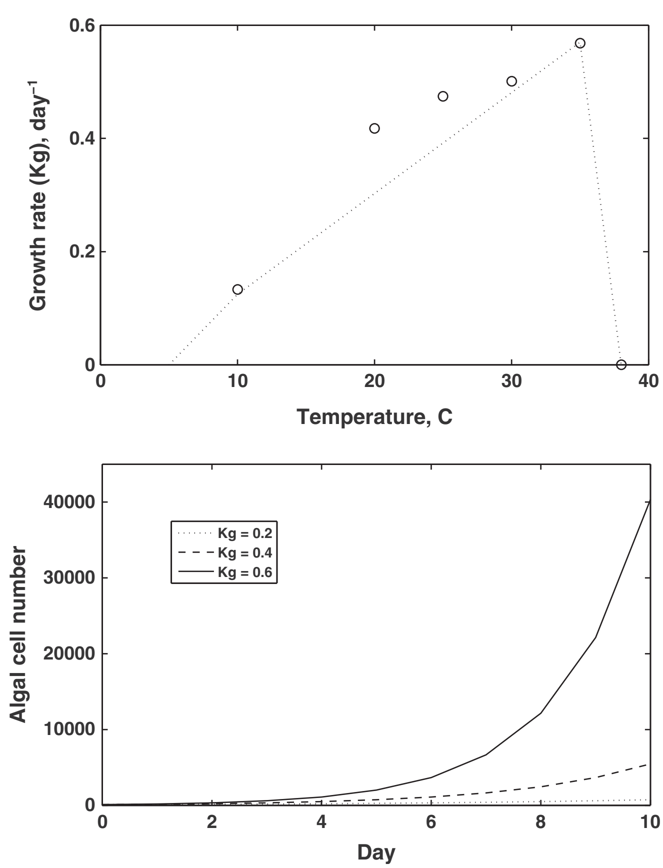

Fig. 5. (Top) Blue-green algal growth rate (K_g) versus temperature. The dashed line is the predicted K_g using the model from Chapra (1997, see Supplementary material S2) while the open circles are the measured growth rate for *Anabaena circinalis* (from UWRAA, 1995), and (bottom) the potential effects of increasing K_g on algal bloom development based on $a_t = a_0 e^{Kgt}$ where a = algal cell number at time t, a_0 is the starting algal cell number (100 in this case), and K_g is as defined above.

4.3. Increased influence of internal processes and resuspension

Reduced flows in drought increase the residence time of soluble and particulate material. Several studies noted that this resulted in internal processes becoming more dominant in determining surface water quality. This can also be seen in Eq. (1) where the assimilation parameters (comprising reaction, kV and settling, vAs) will become more significant in determining water quality if internal processing remains relatively

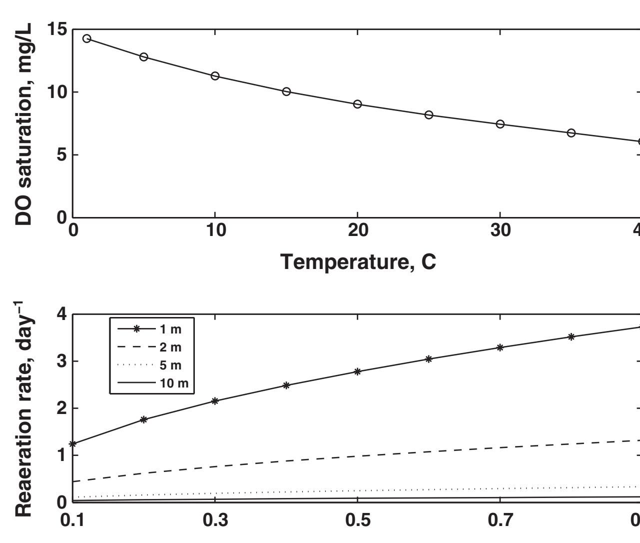

Fig. 6. (Top) dissolved oxygen saturation concentration versus temperature, and (bottom) oxygen reaeration rate (k_a) in rivers based on the O'Connor–Dobbins formula, as provided by Chapra (1997), k_a = 3.93 (U^0.5/H^1.5) where U is flow in m/s and H is water depth in m.

constant while flow and/or volume reduce. There are several internal processes which potentially influence water quality during drought:

(1) benthic algae and macrophytes appear to play a more influential role during drought (Boar et al., 1995; Caruso, 2002; Sprague, 2005). This may be due to a longer timescale for benthic nutrient uptake and/or increased light penetration due to improved water clarity during drought which enables more benthic algal production.
(2) fluxes of material to and from the sediment are likely to be more influential in influencing surface water quality during low flows. For example, denitrification has been noted to increase nitrogen loss during low flow conditions (Harris, 2001; van Vliet and Zwolsman, 2008). van Vliet and Zwolsman (2008) also noted that some of the observed ammonia increase during drought in the Meuse River may have been due to an increased influence of organic matter mineralisation and nitrogen flux from the sediment. Increased stratification and low dissolved oxygen concentrations in the hypoliminion in some systems during drought may also result in increased sediment flux, in particular for Fe, Mn and phosphorus (Di Toro, 2001).
(3) increased hydraulic residence time also enables more time for settling and removal from the water column of particulate inorganic and organic materials (term vA_s in Eq. (1)). This has been linked to improved water clarity and reduced total nutrient levels in some river systems during drought.
(4) reduced water levels during drought were noted to increase wind-driven resuspension and turbidity levels in some lake systems. This could also increase the mobilisation of nutrients and organic matter to the water column. To provide some estimates of the depth and size of systems vulnerable to these effects, the shear stress at the sediment water interface was estimated for different wind (speed and fetch) and water depths using the equations provided by Kang et al. (1982) and Chapra (1997). Larger (>10 km wide) and shallow (<5 m deep) lakes are predicted to be most vulnerable to increased sediment resuspension during drought (Fig. 7) which is consistent with observations (Mosley et al., 2012).

5. Future directions

Some key challenges for research on drought water quality impacts in the future are, in no particular order (1) at a time of increasing hydroclimatic variability and uncertainty of system responses, maintaining long time series of water quality observations remains critically important to characterise drought impacts (Burt et al., 2014b), also for (2) studying the resistance of water quality to recurrent droughts (García-Jurado et al., 2012), (3) additional studies in Asia, South America and Africa would be beneficial due to the very limited amount of published literature on water quality impacts of drought in these regions (Fig. 1), (4) further process-specific research on catchment (Porcal et al., 2009) and internal processes would be beneficial to reduce uncertainties on the drivers of water quality change during drought, (5) further research on links of water quality to higher trophic level and food web responses during drought including trace element (e.g. iron) supply and potential limitation (Donnelly et al., 1997) would be beneficial, (6) increased attention on water quality impacts and interactions in riparian areas as they dry during droughts and rewet post-drought, (7) ongoing development and validation of integrated catchment and receiving water models to predict future water quality impacts of drought (Harding et al., 1995; Sahoo et al., 2013), and (8) improving early warning predictions of hydrological droughts and water quality effects based on preceding climatic and catchment conditions.

6. Conclusions

Droughts and are predicted to increase in many regions of the world due to climate change. There is a need to better understand the effects of droughts on freshwater quality to prevent and/or manage adverse impacts. Over the last 10–20 years there has been an increasing amount

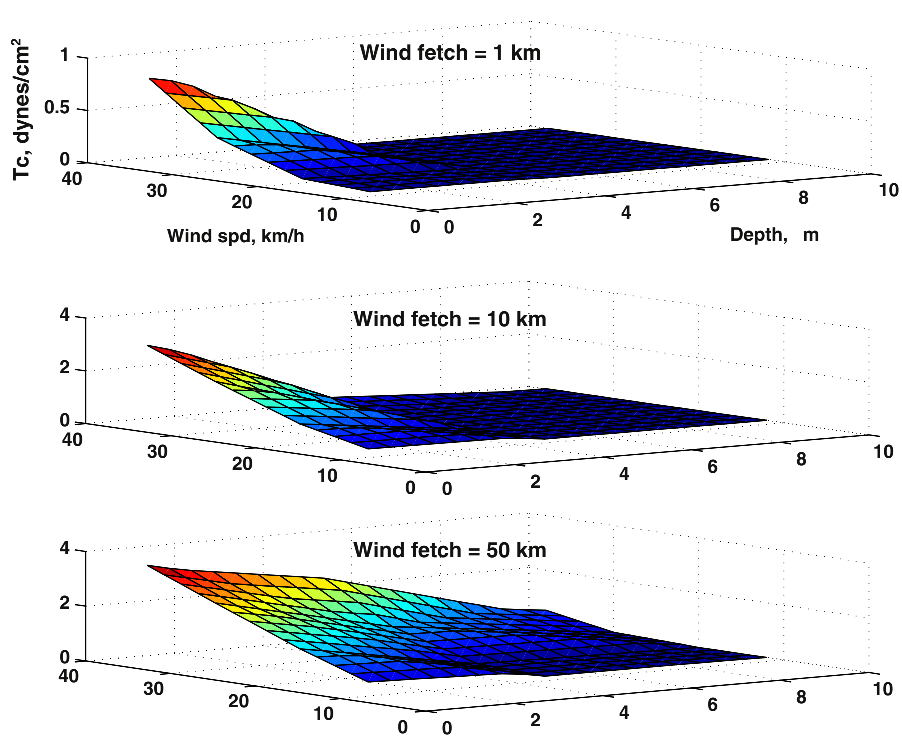

Fig. 7. Shear stress (T_c) estimated at the sediment water interface versus water depth and wind speed using the equations provided by Kang et al. (1982) and Chapra (1997) for different wind fetches. When a critical shear stress value of about 1 dyn/cm^2 is exceeded, appreciable fine sediment resuspension begins to occur.

of observational studies on the water quality effects of drought in freshwater systems, mostly in North America, Europe, and Australia. This review highlighted that a complex variety of water quality responses can occur during drought in rivers/streams and lakes/reservoirs (Section 3). Some of these effects appear somewhat counterintuitive, but many could be explained by consideration of key processes driving water quality outcomes during drought (Section 4), namely (1) hydrological drivers and mass balance, (2) role of temperature and stratification, and (3) increased influence of internal processes and resuspension. Future directions were outlined for research (Section 5) and maintenance of long term water quality monitoring programmes is considered critical for understanding future drought impacts and resilience.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.earscirev.2014.11.010.

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