Hutchins, M.G.; Bowes, M.J. 2018. **Balancing water demand needs with protection of river water quality by minimising stream residence time: an example from the Thames, UK.**

Balancing water demand needs with protection of river water quality by minimising stream residence time: an example from the Thames, UK.

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Abstract

Freshwater resources in the River Thames basin in southern UK are faced with combined pressures of future population growth and climate change. River basin managers are seeking increasingly innovative methods to meet water demand whilst at the same time maintaining ecological status. Using a river network hydrochemical model modified to account for possible future climate and population, the paper assesses the impact on downstream water quality of changing the location of a major point of abstraction serving the city of Oxford. The rationale behind the hypothetical change, although entailing an increase in energy costs and capital expenditure, was that flows would be maintained along a sensitive stretch of river. Model results at a location a further 23 km downstream suggested that better water quality would arise from this change. The predicted improvements included a decrease in the annual frequency of low DO concentrations (<6 mg L^{-1}) from 8-9 days to 2-3 days and a decrease in 90^{th} percentile (summer) temperatures of 0.6 ^{0}C. It is believed these improvements would primarily be attributable to shortening of river residence time which curtails accelerated phytoplankton growth. The overall conclusion, of relevance both for the Thames basin and elsewhere, is that water quality in a river network can be surprisingly sensitive to the location of abstractions. Changing the location of abstractions should be considered as part of a suite of measures available to river basin managers when making plans to meet future water demand.

1. Introduction

Across the globe, freshwater resources are increasingly being threatened due to population growth and changes in climate. Likely depletion of water resources will be exacerbated by increased pollution loads and treatment costs (Wen et al., 2017). As well as potentially failing to satisfy human needs, decreasing flows/river levels have adverse impacts on aquatic habitats and biodiversity (Laize et al., 2013). The principle of environmental flows is becoming increasingly embedded in management planning and acknowledges that water abstraction must be constrained where possible such that river ecosystems are not impaired (Acreman and Ferguson, 2010). Matching water availability under conditions of escalating demand is increasingly problematic. Identifying suitable combinations of investment strategies for addressing water scarcity in ways that protect ecosystems, in addition to providing sufficient supply, is a common challenge throughout the world (Vorosmarty et al., 2010). Meeting this challenge involves complex socio-economic considerations for which analysis using dynamic models can provide valuable insights (Dadson et al., 2017).

The Thames river basin (southern UK) illustrates many contemporary issues, facing a likely increase in population (12% for England from 2017-2041: ONS 2017) alongside increasing climatic stress. An increase in flood frequency and magnitude brought about by climate drivers is very possible (Bell et al., 2012). At the other extreme, a future downward trend in low flows is likely (Prudhomme et al., 2012). Projections suggest water resources will be threatened unless water is managed more efficiently and sustainably in domestic and industrial sectors (Hutchins et al., 2017). Allied to this it is expected that the increasing river residence time arising from lower baseflow will trigger longer and more severe eutrophication episodes (Bowes et al., 2012; Hutchins et al., 2016). Water companies are seeking concerted and innovative ways of balancing future water supply with environmental requirements. Thames Water (2016) are evaluating a suite of measures such as additional reservoir capacity, water transfers, waste-water re-use and desalinisation to meet the likely shortfall. Equally, regulators are working towards coherence in water quality and water resource planning around climate adaptation.

The present paper seeks to show how an alternative configuration of water infrastructure (abstraction and effluent locations) can, in the right circumstances, help protect river water quality whilst minimising energy usage and capital expenditure. Specifically the alternative scenario we consider maintains river flows in a stretch of the Thames passing through Oxford where rapid increases in chlorophyll concentrations are observed in most years (Bowes et al., 2012). To do this we use QUESTOR, a water flow and quality model developed for the upper Thames (Hutchins et al., 2016).

2. Method

2.1. Study area

The Thames, a river of total length 354 km has the largest catchment area wholly in England (Figure 1). In the basin, 40% of water supply comes from groundwater (predominantly Oolitic Limestone and Cretaceous Chalk aquifers). In terms of the water quality of groundwater bodies, 47% and 38% have poor quantitative and chemical status respectively (Environment Agency 2016). Whilst the majority of surface water bodies have good chemical status, only a small minority (<10%) meet good ecological status.

The upper Thames (catchment area at the town of Wallingford 3445 km^2) is the focus of the present study. The upper basin receives mean annual rainfall of 744 mm (Marsh and Hannaford, 2008). It is predominantly rural despite being highly populous (the dominant land classification being arable 45%, with only 6% urban/suburban). Surface water supply is primarily from Farmoor Reservoir (with an abstraction from the river 2.9 km upstream of Site 2 on Figure 1), which supplies Swindon and Oxford amongst other population centres. The Farmoor abstraction of 1.62 m^{3}s^{-1} effectively reduces mean flow at Wallingford (51 km downstream) by about 5%. In return, the Sewage Treatment Works (STW) effluent downstream of Oxford contributes 0.47 m^{3}s^{-1}. At Wallingford, ample nutrient loads sustain phytoplankton; nitrate-N and total phosphorus in recent years always exceeding 1.4 and 0.09 mgL^{-1} respectively (Bowes et al., 2012).

2.2. Model description

The QUESTOR model application (Hutchins et al., 2016) focuses on a stretch representing 126.4 km of river channel network (comprising the River Cherwell and River Thame tributaries and the main Thames) split into 41 reaches. The model is fed by 23 tributaries and 7 major STWs, and accounts for 2 abstractions and 22 weirs. The main determinands simulated are chlorophyll-a (a proxy for phytoplankton biomass), biochemical oxygen demand (BOD), dissolved oxygen (DO), inorganic phosphorus (equating to SRP, the Soluble Reactive Phosphorus fraction), organic phosphorus, nitrate, particulate organic nitrogen, ammonium, pH, temperature, flow and photosynthetically-active radiation in the water column. The processes represented are aeration, BOD decay, deamination, nitrification, denitrification, benthic oxygen demand, BOD sedimentation, phosphorus mineralisation, in conjunction with a biological sub-model of phytoplankton (comprising growth, respiration and death), which includes nutrient uptake and release. A mixed phytoplankton population is assumed. The equations describing DO, BOD, temperature and chlorophyll-a are given (Online Resource 1). Values of the Nash-Sutcliffe efficiency goodness-of-fit statistic for model performance at Wallingford (Site 4) for a two-year period of testing against weekly data were 0.81, 0.77, 0.13 and 0.22 for temperature, nitrate, SRP and chlorophyll-a respectively. Further model performance statistics are shown below (Table 1) and discussed in detail by Hutchins et al. (2016). Overestimation in SRP is predominantly attributed to especially low flows in the Thame tributary, conditions known to promote attenuation of phosphorus in bed sediments, which is not represented in the model. Mismatches of chlorophyll time series are primarily arise through assuming a constant grazing rate for phytoplankton loss. Data are insufficient to represent grazing in the model in more detail.

Table 1: Paired values under calibration (2009-10) and corroboration (2011-12) conditions (separated by “,”) of NSE for daily flow, and % error in mean for temperature, DO, nitrate (NO_3), soluble reactive phosphorus (SRP) and chlorophyll-a. Values in bold are based on observed data availability at a resolution of weekly or better. a) locations of monitoring sites 1-4 on Figure 1; b) data for Abingdon only available in 2009

[a]**Monitoring Site**	**Flow**	**Temp**	**DO**	**NO_3**	**SRP**	**Chl-a**
Newbridge (1)		**0.5, 5.9**	3.6, 13.1	**0.82, -5.0**	**-9.3, 6.8**	**-25.5, -9.1**
Eynsham (2)	**0.92, 0.91**	**2.2, 8.6**		**-1.3, -5.4**	**2.0, 12.7**	**-27.9, 31.4**
[b]Abingdon (3)		**13.4, n/a**	**-3.6, n/a**			**7.3, n/a**
Wallingford (4)		**6.1, 7.9**	-1.4, 3.6	**-4.3, -3.0**	**12.3, 24.6**	**-29.4, 1.0**

2.3. Model applications

The model was run for a 4 year period, based on 2009-12 weather patterns, but with modifications made to account for changes which may occur under future climate and population growth. Input conditions in tributaries and other influences (e.g. for N and P concentrations) were defined by taking present day monthly mean concentrations. Thereby it was assumed that agricultural nutrient management practice and levels of sewage treatment would remain unchanged. Daily radiation data were provided from Little Rissington near the River Windrush in Gloucestershire (NGR 4299 2107) by the British Atmospheric Data Centre (MIDAS Landsat Data). To account for effects of riparian shading, direct radiation reaching the water surface was reduced by 19% under conditions of full leaf, this reduction equates to riparian canopy occupancy of 27%. Waylett et al. (2013) provide further details of the procedure for quantifying shading. Representations of effects of future climate on river flows and water temperature were guided by modelling of hydroclimatology (Prudhomme et al., 2012) as summarised for the Thames by Hutchins et al. (2016). Focus was made on best capturing summer conditions when river water quality is most vulnerable to deterioration. To represent population growth, the UK Office for National Statistics previously estimated a 16% growth from the period covered by QUESTOR model testing up to 2035. To allow for these projections the following modifications to all present-day daily values of model input were made:

- Flow: scalar multiplier x0.8
- Water temperature: change factor $+3^{o}C$
- Urbanisation: scalar multiplier x1.16 (this represents a combination of population growth and changes in water use efficiency)

To meet the objectives of the paper a pair of model applications was undertaken to represent the following scenarios (Figure 2):

1. A system with the same configuration for abstraction as occurs presently (i.e. abstraction from the River Thames upstream of Farmoor Reservoir). “Present Configuration (PC)”
2. A scenario whereby the volume of water currently abstracted upstream of Farmoor is abstracted from the river further downstream near the town of Abingdon instead (Site 3) and piped back to Farmoor for storage and distribution. “Alternative Configuration (AC)”, This Alternative Configuration (AC) scenario would avoid the reduced flows for the 20 km stretch through Oxford, thereby reducing residence times.

3. Results and Discussion

Whilst simulated flows will differ between the scenarios from reach to reach between Sites 2 and 3, at Wallingford (Site 4) they are identical for the two configurations. In contrast, a set of water quality indicators, representative of summer low flow periods when conditions are most vulnerable, (Table 2) are substantially better under the alternative configuration. These indicators are assessed in the context of regulatory standards. UKTAG (2008) cite the Freshwater Fish Directive values for (i) 98th percentile water temperature of 21.5 C for salmonids and 28 C for cyprinids, (ii) 10th percentile DO set at 6 mg L^{-1} (iii) a 90th percentile BOD value of 4 mg L^{-1} is cited as a good status target for salmonid rivers. Standards related to phytoplankton biomass are absent in the UK. However in the USA, Dodds et al. (1998) cite summer median chlorophyll-a concentrations above 0.03 mg L^{-1} as being indicative of eutrophic conditions.

Table 2: Summary water quality outcomes for a set of indicators.

Water quality indicator	1. Present Configuration	2. Alternative Configuration
5th percentile DO (mg/L)	6.76	7.69
1st percentile DO (mg/L)	5.32	6.21
Days in the 4 year period with DO < 6 mg/L	33	10
90th percentile chlorophyll-a (mg/L)	0.093	0.081
90th percentile water temperature (^{0}C)	25.1	24.5
90th percentile BOD (mg/L)	3.65	2.86

Fluctuating and periodically low DO occurs throughout the summer when river flows are low (Figure 3). These did not occur in the wet summer conditions of 2012. The number of days with low DO are more frequent under PC than AC. The key difference for DO and other indicators of water quality is that in the “present configuration”, when conditions become drier in the summer, the flow rate becomes low in the 20 km stretch of the Thames passing through Oxford (between Farmoor and Sandford where Oxford sewage effluent returns to the Thames). These low flows provide potentially longer residence times and therefore viable conditions for phytoplankton blooms to develop and then crash due to nutrient limitation. Crashes generate BOD and remove DO. Lower flows and longer residence times also lead to higher water temperatures which in turn can further lower the DO. In the “alternative configuration” flow levels are maintained through the stretch between Farmoor and Sandford. Consequently accelerated eutrophication and its impacts are less likely to occur, and the river will warm up to a lesser extent.

It is apparent that the differences in DO between the two configurations by no means wholly arise from differences in water temperature. (Figure 4). Especially in early-midsummer these differences are driven by eutrophication impacts, corresponding more strongly to differences in chlorophyll-a and BOD.

4. Wider Implications

Abstracting from Abingdon instead of upstream of Farmoor Reservoir (for water supply to urban areas such as Oxford) may not initially seem rational from the perspective of economic and energy costs, yet the water quality downstream is predicted to be substantially better. Of particular note is the protected reduction in the incidence of DO falling below the ecological threshold of 6 mg L^{-1}. The incidence of poor water quality is attributable to periods where flows are low in part of the river network, in this case through Oxford. The finding, that flow levels can have direct water quality and ecological implications, is of direct relevance for the pinpointing of environmental flow requirements.

Our results, identifying considerable implications arising from moving a major abstraction point, merit further discussion. Water supply in the Thames has some inherent vulnerability to climate pressures, as the storage space quoted by Thames Water (2016) is only of the order of 100 days. Of the options available to meet future shortfall, raw water transfers from wetter regions of the country have been put forward. Whilst those water transfers deemed plausible (Thames Water, 2016) have a deployable output of approximately 3 m^3s^{-1} and could greatly improve water quality as well as meeting shortfalls, adverse effects are likely. Canal or pipeline construction is complex. Transfers may introduce invasive species and will impair the natural flow regime in the upper Thames. In addition, impacts on the source water body may be detrimental.

An analysis of the differences in capital expenditure and operating costs between raw water transfer options and a re-configuration of reservoir storage outlined above is out of the scope of this paper. Nevertheless, some broad differences are noteworthy. Piping and treating Thames river water from Abingdon, which is more polluted than river water at Farmoor, would clearly incur greater overall costs than that entailed presently. In comparison however, a potential transfer from the adjacent River Severn basin would potentially be much more costly, needing to cover a far longer distance (approximately 4 times as far) and much hillier terrain (approximately 10 times the increase in altitude).

It seems that the option addressed here should be considered alongside other major water supply options when making plans to meet future water demand. We argue that such considerations should be built into strategic appraisals by river basin managers of the various options available. This is important both in basins such as the Thames, but not least in those regions throughout the world where urbanisation is predicted to proceed much more rapidly and where infrastructure is currently minimal or absent.

Acknowledgements

We acknowledge support from NERC-CEH Pollution & Environmental Risk and Water Resources Science Areas. The model was supplied with and tested against hydrological data from the National River Flow Archive. Solar radiation data were accessed from British Atmospheric Data Centre.

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List of Figures

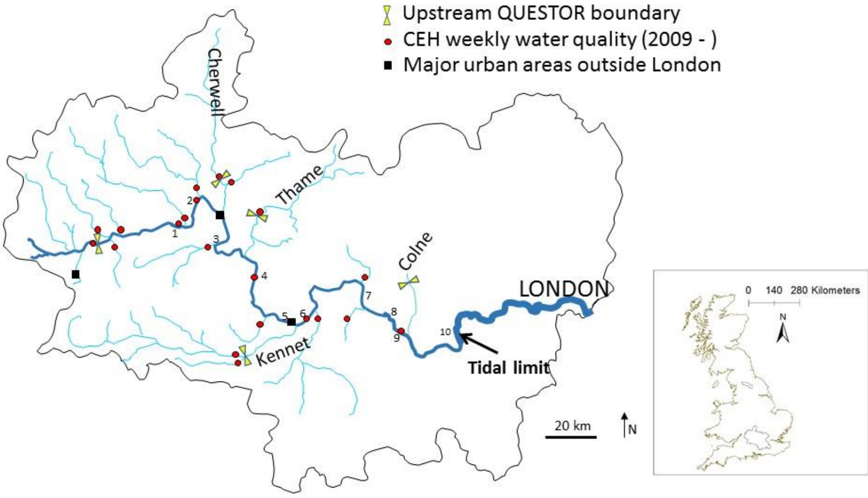

Upstream QUESTOR boundary
CEH weekly water quality (2009 -)
Major urban areas outside London
Cherwell
Thame
Colne
Kennet
LONDON
Tidal limit
1
2
3
4
5
6
7
8
9
10
20 km
N
0 140 280 Kilometers
N

Figure 1

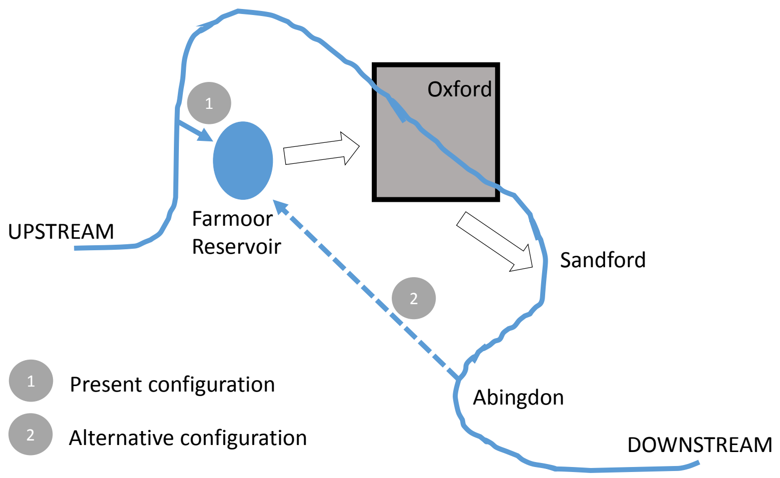
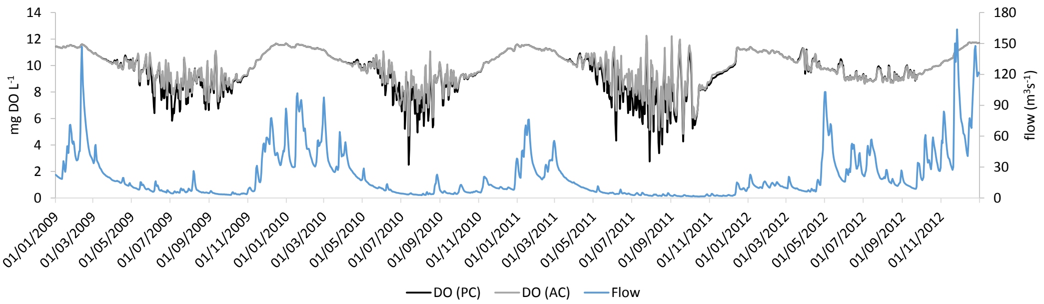

Oxford
1
UPSTREAM
Farmoor
Reservoir
Sandford
2
1
Present configuration
2
Alternative configuration
Abingdon
DOWNSTREAM

Figure 2


mg DO L-1
flow (m3s-1)
0
2
4
6
8
10
12
14
0
30
60
90
120
150
180
01/01/2009
01/03/2009
01/05/2009
01/07/2009
01/09/2009
01/11/2009
01/01/2010
01/03/2010
01/05/2010
01/07/2010
01/09/2010
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01/03/2011
01/05/2011
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01/09/2011
01/11/2011
01/01/2012
01/03/2012
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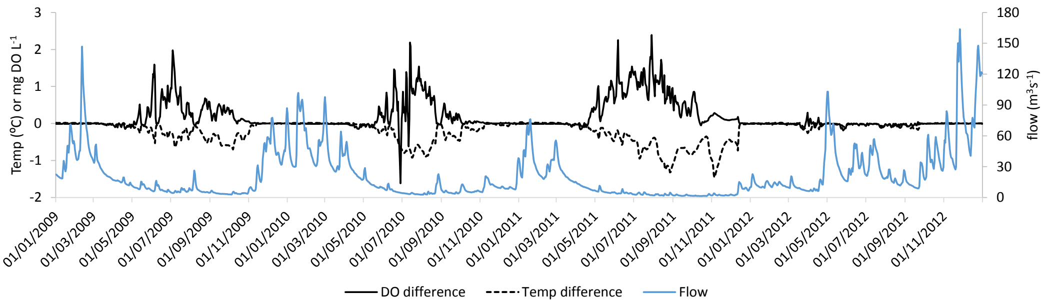
01/07/2012
01/09/2012
01/11/2012
DO (PC)
DO (AC)
Flow

Figure 3



Temp (°C) or mg DO L-1
flow (m3s-1)
DO difference
Temp difference
Flow

Figure 4

Online Resource 1. Theoretical basis to QUESTOR

QUESTOR simulates Chlorophyll-a (Phytoplankton), Biochemical Oxygen Demand (BOD), Dissolved Oxygen (DO), Inorganic (P-in, equating to soluble reactive fraction) and Organic Phosphorus, Nitrate, Particulate Organic Nitrogen, Ammonium, pH, Temperature, Flow and Photosynthetically-Active Radiation in the water column. Processes that the QUESTOR model represents are aeration, BOD Decay, Deamination, Nitrification, Denitrification, Benthic Oxygen Demand, BOD Sedimentation, P Mineralisation, in conjunction with a biological sub-model of Phytoplankton (comprising Growth, Respiration and Death), which includes nutrient uptake and release. To simulate the hydrological and chemical variables the configuration of QUESTOR as described by Boorman (2003) was used. The full sets of equations used are given elsewhere (Boorman, 2003) so only those equations directly impinging on phytoplankton and DO concentrations are given here.

1. Phytoplankton model

The growth of a mixed population of phytoplankton is modelled as described by Hutchins et al. (2010) using a fixed stoichiometry model whereby the ratio chl-a:C:N:P was 1:50:10:1.

Equation A1: Shows the photosynthetic rate with respect to biomass and temperature. For the Mixed Population model the calculation based on the Arrhenius equation.

$$k^{pho} = Phy.k_{ref}^{pho}.\theta^{(T-Tref)}.f(N).f(L) \qquad [A1]$$

k^{pho} = Photosynthesitc rate (day^{-1}),
Phy = Concentration of Chl-a ($mg\ L^{-1}$)
T = Temperature (°C),
T_{ref} = 20 °C
f (N) and *f(L)* = limitation factors for nutrients and light, each holding values between 0 and 1
θ = Arrhenius factor for temperature dependencies (θ =1.08)
k^{pho}_{ref} = Maximum phytoplankton growth rate (day^{-1}) at $T_{ref.}$

Equation A2: Calculates the maximum photosynthetic rate and the limitations by nutrients, this has been taken from Michaelis Menten kinetics

$$f(N) = \min\left(\frac{N}{N+k_N}, \frac{P}{P+k_P}\right) \qquad [A2]$$

N = Nitrate-N plus Ammonium-N ($mg\ L^{-1}$)
P = Inorganic-P (equivalent to SRP) plus Organic-P ($mg\ L^{-1}$)
Where k_N = 0.1 and k_P = 0.01 $mg\ L^{-1}$

Equation A3: Light limitation, attenuation with depth is described by the Beer-Lambert Law

$$\gamma = \gamma_{base} + L_{SS}.SS + L_{Phy}.Phy \qquad [A3]$$

γ_{base} = light extinction coefficient in clean water (0.01 m^{-1})
SS = concentration of suspended sediment ($mg\ L^{-1}$)
L_{ss} = Light attenuation with depth due to suspended sediment ($m^{-1}\ mg^{-1}\ L$)
L_{phy} = Light attenuation with depth due to phytoplankton ($m^{-1}\ mg^{-1}\ L$)

Equation A4: Photolimitation with respect to phytoplankton-specific optimum intensities (Steele, 1962)

$$f(L) = \frac{2.718}{\gamma d}.\left[\exp\left(\frac{R_s L_1 L_2}{L_{opt}}.\exp(-\gamma d)\right) - \exp\left(\frac{R_s L_1 L_2}{L_{opt}}\right)\right] \quad [A4]$$

γd = Water column depth (m),
R_s = Radiation at the surface not reflected (W m^{-2}) (i.e. input solar radiation x 0.6)
L_1 = Fraction of incoming radiation that is visible light (0.5)
L_2 = Fraction of visible light used for phytoplankton (0.5)
L_{opt} = Optimum light intensity for phytoplankton (60 W m^{-2})

Equation A5: Respiration

$$k^{res} = Phy.k^{res}_{ref}.k^{pho}_{ref}.\theta^{(T-Tref)} \quad [A5]$$

k^{res}_{ref} = reference respiration rate for phytoplankton (day^{-1})

Equation A6: Death

$$k^{death} = Phy.k^{death}_{ref}.k^{pho}_{ref}.\left[1-\left(f(N).f(L)\right)\right].\theta^{(T-Tref)} \quad [A6]$$

k^{death}_{ref} = reference death rate for phytoplankton (day^{-1})

Death is a combination of grazing and non-predatory mortality.

2. Dissolved Oxygen model

Equation A7: Change in Dissolved Oxygen.

$$\frac{dDO}{dt} = \frac{1}{T}(DO_i - DO + W) + (P - R)$$

$$-(k_{ben}DO/dep) + k_{rea}(OCS - DO) - 4.57k_{nit}NH_4 - k_{bod}BOD \quad [A7]$$

Where:
T = a time constant representing the average retention time in the reach. This is defined by $L/(bQ^c)$ in which L is length of reach (m), Q is flow out of reach (m^3s^{-1}) and b and c are reach specific constants.
DO = DO concentration leaving the reach (mgL^{-1})
DO_i = input DO concentration (mgL^{-1})
W = aerating effect of a weir as calculated from an empirical relationship based on weir type and height
P = $k^{pho}(133.3Phy)$ = DO increase due to photosynthesis
R = $k^{res}(133.3Phy)$ = DO decrease due to respiration
k_{ben} = benthic respiration rate (day^{-1})
dep = mean water depth of reach (m)
k_{bod}= rate of loss of DO as BOD decays (day^{-1})

k_{nit} = rate coefficient for complete nitrification (day^{-1})
NH_4 = concentration of ammonium in water column (mg L^{-1})
k_{rea}= aeration coefficient at the water surface (day^{-1}) (dependent on velocity, depth and temperature)
OCS = DO concentration at saturation (mg L^{-1})

The amount of oxygen produced in photosynthesis (P) or consumed in respiration (R) per unit mass of algae. For each 1 mg of chlorophyll-a 133.3 mg of oxygen are produced. This same ratio applies for oxygen consumption in respiration, and in additions to BOD on phytoplankton death.

3. Biochemical oxygen demand model

<u>Equation A8:</u> Change in biochemical oxygen demand:

$$\frac{dBOD}{dt} = \frac{1}{\tau}(BOD_i - BOD) - k_{bod}BOD - \frac{(v_{sed}BOD)}{dep} + k^{death}(133.3Phy) \qquad [A8]$$

Where:
BOD = BOD concentration leaving the reach (mgL^{-1})
BOD_i = input DO concentration (mgL^{-1}) (mean from all sources)
v_{sed} = settling velocity of BOD. A value of 0.25 ms^{-1} was used.

4. River water temperature model

<u>Equation A9:</u> Change in water temperature is defined as follows:

$$\frac{dT}{dt} = \frac{1}{\tau}(T_i - T) - \frac{H(R_s - R_o)}{dep} \qquad [A9]$$

Where:
T_i = mean temperature (°C) from all sources
T = temperature in water leaving the reach (°C)
R_o = outgoing long-wave radiation (Wm^{-2})
H = heat flux coefficient (0.005 m^{-1})
The largest component for the outgoing radiation is the long wave back radiation which is given by
R_o=0.97 σ T^4 (in which 0.97 is the emissivity constant of a water surface and σ is the Stefan-Boltzman constant (5.67051 10^{-8} Wm^{-2} k^{-4}) and T is the temperature in ^{o}K)