Assessment of Drought Impacts on Water Quantity and Quality: Implications for Water Resources and the Severn Thames Transfer



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Abstract

Water resource management planners are increasingly aware of the effects that climate change and population growth will have on current, conventional water resources management plans. This requires an understanding of the vulnerability and sustainability of the current water supply options and the development of new water resource options to cope with more intense, longer, and more frequent droughts. Water stress in South East England is projected to increase over the next planning horizon, and water planners are continuously looking to develop drought management options with neighbouring water companies to transfer water cross basins to alleviate some of the water stress. The Severn-Thames Transfer is actively discussed as the preferred investment scheme to secure water supply in the Thames basin in the United Kingdom. In terms of inter-basin water transfers, understanding surplus water availability from the donor catchment is a vital factor. Thus thorough exploration of the wide range of extreme events in both donor and receiving catchments must be incorporated into the feasibility assessments. Together with consideration of water quantity resilience in the system, water quality risks associated with mixing two waters must be addressed prior to implementation.

The overarching aim of this research is to provide a comprehensive water quality and quantity modelling representation of the hypothetical inter-basin water transfer scheme under extreme and spatially extensive drought events. The aim is to explore the possible occurrence of trade-offs between water quality and quantity within the proposed transfer scheme. This research utilised state of the art hydro-climatic data alongside hydrological modelling and water quality modelling to achieve this research aim. A new proposed

inter-basin transfer scheme, the Severn-Thames Transfer, has been chosen to demonstrate the new modelling framework.

The findings herein suggest the value of the new modelling framework, because of the potential to explore multiple aspects of risks on the new proposed water resources. The methodology used has contributed to an enhanced understanding of catchment specific water quality responses to extreme drought, and enabled the identification of critical water quality variables, which might impose risks during inter-basin water transfers. The use of this case study demonstrated that more spatially extensive and severe droughts are projected under a changing climate, and it will threaten the water supply in London and degrade the water quality in the Thames and Severn Rivers. The results also suggest that inter-basin transfers can degrade water quality although water stress in the Thames basin can be alleviated by increasing supply. Regarding water quality impacts by the suggested transfers, total phosphorus and dissolved organic carbon will be a concern and introduce an additional burden on water supply systems and ecological conditions in the Thames. Reduced nitrate and suspended sediments concentrations are expected in the Thames while transfers as the water quality in lower Severn is better. Therefore, the timing of transfers will become the most critical factor for the new proposed transfer scheme.

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1 Introduction

1.1 Background

Water availability is distributed unevenly around the world and achieving water security has been a critical challenge to society. It is estimated that approximately four billion people currently experience severe physical water scarcity for at least one month per year (Mekonnen and Hoekstra, 2016) and almost a quarter of the world's population face economic water shortage due to the lack of infrastructure to access water (UNESCO UN-Water, 2020). In light of these facts, water security is explicitly included in the 2015 United Nation's Sustainable Development Goals (SDG 6: Clean water and sanitation).

Climate change affects temperature and precipitation patterns, and the inter-annual variability of the summer climate in many areas of the world (Seneviratne et al., 2006) leads to intensified hydrological droughts in some regions (Huntington, 2006; Prudhomme et al., 2014). As changes in precipitation and temperature directly influence the terrestrial water budget, a changing climate will require revision of conventional water resources options. Land use alteration and increasing human exploitation will further exacerbate the problem and increase the water pressure. By 2030, a 40% of water deficit or a supply-demand gap is estimated assuming no efficiency gains with a current business-as-usual scenario (2030 WRG, 2009). Therefore, water managers continuously encounter challenges in meeting water demand for sustainable water supply.

Climate change will stimulate water-engineering responses in anticipation of frequent and severe extreme events such as floods and droughts. Inter-basin water transfer is one of the most sought-after supply- and engineering-oriented options in order to mitigate water

shortages. Inter-basin transfers are defined as the "*transfer of water from one geographically distinct river catchment or basin to another, or from one river reach to another* (Davies et al., 1992)". Although it usually does not increase the overall amount of water or water availability in a state, it helps move water to where it is needed most. In 2005, inter-basin water transfer contributed to approximately 14% of total water withdrawals globally, and the amount is expected to rise up to 25% by 2025 (Gupta and van der Zaag, 2008).

Globally, more than 160 inter-basin water transfer projects have been implemented or are currently under construction, including in Canada, the United States, India, Pakistan, China, South Africa, Brazil (de Lucena Barbosa et al., 2021). Inter-basin water transfer is a double-edged sword. While the transferring water alleviates the pressure on water supply and the benefits of inter-basin water transfer are well-recognised, the physical, chemical, and biological impacts on both "donor" and "recipient" systems from transferring waters are still an on-going discussion. Inter-basin water transfers will modify the water flow, chemistry, and temperature of the recipient water body, consequently affecting aquatic communities, ecology, and water supply. Some studies reported that inter-basin water transfer brings positive impacts on the recipient water body. For example, enhancement of water quality was observed through water exchange and self-purification capacity improvement (Zhuang, 2016). However, others have listed negative impacts; the spread of new pollutants and diseases (Davies et al., 1992, Yan et al., 2018); nutrients transport through the water channel (Zhuang, 2016); the onset and acceleration of sediment erosion and transport processes (Davies et al., 1992) and the introduction of invasive species through the transfer (Gallardo and Aldridge, 2018).

Water security is an inclusive term implying the reliable availability of quantity and quality of water for health, ecosystem, and productive economies with an acceptable level of resilience on supply. Inter-basin water transfer is an option to increase water security, whilst ensuring safe water quality in both donor and recipient system deserves greater consideration. Water resources planning has traditionally been based on consideration of water availability, and water quantity and quality are often dealt with separately. Indeed, in many countries, there is a set of specific water quality standards for public water supply. Hence, it will cause concern to the abstractors if there is a poor quality of water in the sources, such as rivers, groundwaters and reservoirs. Additionally, the health of the aquatic environment is profoundly influenced by water quality. However, those important indicators are often neglected when new water resources management options are considered or introduced.

In the UK context, water stress in South East England is projected to increase over the next planning horizon (Environment Agency, 2013a; von Lany et al., 2013). Despite the decrease in Per Capita Consumption (PCC) of water, population growth and a projected increase of aridity in the country will lead to the overall increase of water use demand (Hall et al., 2019b). Continuous efforts have been made in the last decade to cope with the stress and understand uncertainties affecting water resource planning and management in this region. Water resources management is generally viewed as a risk management problem (Hall and Borgomeo, 2013; Veldkamp et al., 2016) and quantification of those risks in planning is becoming a new norm, especially in advanced economies where over-exploitation in their water resources has occurred (Hall et al., 2019a). The risk-based approach provides an explicit framework to quantify the potential

harmful outcomes in the water resources system and the methodology has been evolving with incorporating various risk factors in the water resources system, such as frequency of water use restrictions in UK (Borgomeo et al., 2016). The robust risk-based approach is crucial to assess existing infrastructure adaptiveness or even more on the introduction of the new infrastructure investment to the system.

Water planners in South East England are looking to develop options with neighbouring water companies to transfer water across basins from areas which have a positive supply-demand balance to negative balance, whilst internal adaptation plans are being implemented to overcome some of the water stress. Among those water resources development options, the Severn-Thames Transfer is a large scale transfer project (Thames Water, 2020a). In terms of inter-basin water transfer, understanding surplus water availability from the donor catchment will be a vital factor for the decision making on implementation so that the thorough exploration of wide range of extreme events in both donor and receiving catchments are incorporated into the feasibility assessment. Together with the water quantity resilience in the system, the risk of two mixing waters from a water quality perspective should be addressed. Coincident droughts may result in poor water quality being transferred from one catchment to another. Therefore, any additional risks from water quality deterioration have to be identified prior to implementation. Climate change and water scarcity within two catchments would exert additional pressure on water quality dynamics during inter-basin water transfer. The extent and magnitude of those impacts would vary depending on the timing of transfer and the hydrological and environmental condition of donor and recipient catchment.

This DPhil research thesis will investigate the water quantity and quality trade-offs and quantify those impacts from the new strategic plan entitled, the Severn-Thames Transfer. In the next section, the details and history of the Severn-Thames Transfer scheme will be explained.

1.2 Choice of case study – Severn - Thames Transfer

This doctoral research presents an experimental methodology to evaluate dynamic water quality impacts of inter-basin water transfers under future droughts. The Severn-Thames Transfer is chosen to demonstrate the proposed methodology. It is one of the new strategic water transfers currently being considered for implementation in South East England. This section explains the history of Severn-Thames Transfer scheme development within water resources management plans and the up-to-date planning options.

The idea of transferring water from surplus areas to deficit areas is not new, and options for a water transfer have been discussed and investigated repeatedly in the UK at regional and national levels both while the water utilities have been under public and private ownership. The occurrence of a severe drought event in 1975/1976 across the UK and public awareness of climate change have increased support for transfer schemes (Thames Water, 2014). Especially, as water availability is low in South East England, seeking potential sources for additional water supply is crucial. Therefore, transferring water from the West of England and Wales, the Midlands, and the North of England to the River Thames Basin has been regularly reviewed by water managers (Environment Agency, 2020; Thames Water, 2014). Large spatial variations in precipitation are observed from west to east in England due to the prevailing westerly weather system (Fleig et al., 2011). The Severn and Thames are geographically adjacent, but hydrologically contrasting catchments. The scheme would work efficiently to transfer surplus water from close to downstream of the Severn into the Thames' upper stream. Figure 1.1 shows the Severn and Thames catchments in a map showing the locations and catchment boundaries.

Thames Water Utilities Ltd (hereafter Thames Water), a water company responsible for water resources management in the Thames Basin, supplies about 10 million household customers and 215,000 businesses in London and across Thames Valley (Thames Water, 2020a). The Environment Agency (2013c) classified the Thames Water region as an area of serious water stress under climate change and increasing demands pressure, whereas Severn Trent Water, who oversees the Severn Basin water supplies, was categorised as a moderate stress area. For this reason, the strategic water transfer scheme between the two basins has been reviewed and included in the Water Resources Management Plan 2014 (WRMP14) (Thames Water, 2014). Every 5 years, water companies are responsible for publishing WRMP reports to regulators and customers. WRMPs set out water utility's business plan, supply-demand balance, feasible water resources development, and analysis of future risks in their responsible water resources zone. In WRMP 14, the water plan identified 23 potential new water resources options, including raw water transfers with other water companies, new wastewater re-use schemes, desalination, as well as a small groundwater scheme (Thames Water, 2014). Several options for transferring water from the River Severn to the River Thames were then identified, including options to transfer water between the Severn and Thames directly via a pipeline without a supporting reservoir (unsupported transfer), and to have strategic reservoir storage to support water transfer via a pipeline (supported transfer). In addition, an alternative scheme using the Cotswold canal network, which is supported by a strategic reservoir in the Severn, was considered. The potential transfer capacity was up to 300Ml/d for unsupported transfer and Cotswold canal network, and up to 600Ml/d with support from strategic reservoirs (Thames Water, 2014). A follow-up screening report (Water Resource Management Plan 2019 (WRMP19) suggested transferring the unsupported Severn water to the Thames

via the Cotswold canal or Deerhurst pipeline (Thames Water, 2015). The location of the possible abstraction point in lower Severn was discussed as at, or close to, Deerhurst. However, discharge points in the Thames were not decided but identified between Cricklade and Culham in the upper Thames.

Water UK (2016) and the National Infrastructure Commission (2018) have stated the need of strategic transfers to increase the resilience of the water resource system and management in severe droughts in the UK. In the recent updated WRMP19, Thames Water set out the recent progress of water resource development plan for the 80-year period of planning (2020 to 2100). The plan emphasizes that not a single but multiple strategic options must be required in order to meet the future supply demand challenges in the South East of England (Thames Water, 2020a). With demand management via improving water efficiency (leakage reduction and water metering), the Severn-Thames Transfer is included as a preferred investment scheme at the end of the planning period (2080s). It is suggested that a joint operation with a new reservoir in Oxfordshire (called the South East Strategic Reservoir Option), would facilitate the benefits of the scheme (Thames Water, 2020b). Transfers between 300 to 500Ml/d are considered on the WRMP19 planning options and the larger capacity (500Ml/d) is being supported by the Vyrnwy Reservoir at the head of the Severn.

The extant understanding on the potential impacts of the Severn-Thames Transfer scheme on the water quality in the Thames seems fragmented in the WRMP19. In particular, studies on using probabilistic future climatic change projections to test the suggested transfer options have been rarely researched for possible water quality degradation. The implementation of inter-basin water transfer incurs large financial costs in terms of

construction and operations of infrastructure; thus it is not reversible. Therefore, it is imperative to test possible risks on water quality in the Thames against a wide range of climate change scenarios containing conditions beyond historical observations. This highlights the need for a water quantity-quality modelling approach to explore the water management options during future droughts.

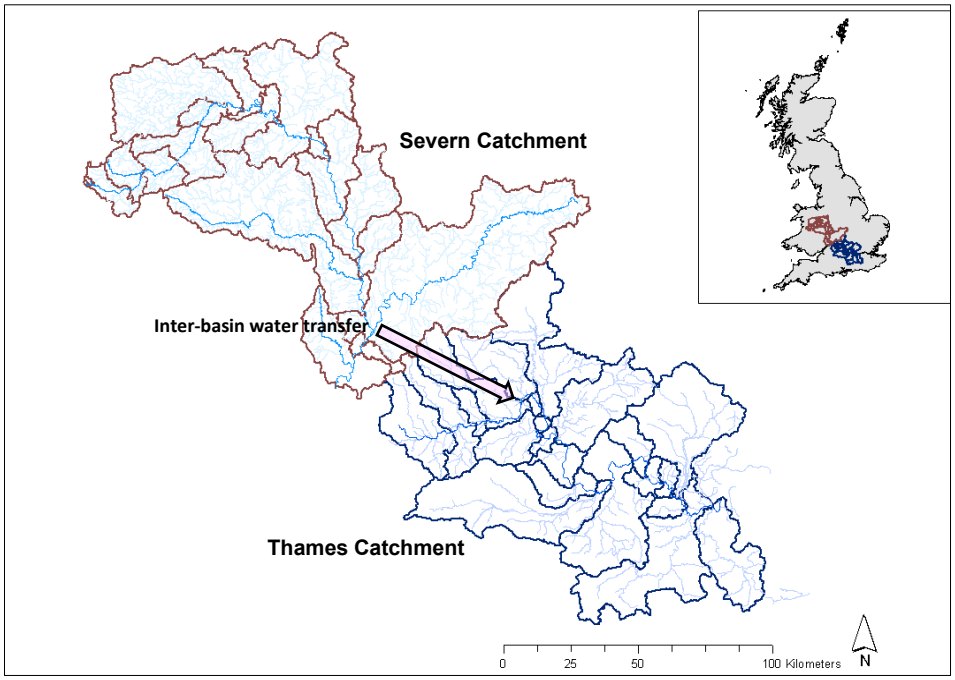


Figure 1.1: The Severn and Thames catchments in the UK

1.3 Research aims and objectives

The overarching aim of this research is to provide a comprehensive water quality and quantity modelling representation of the hypothetical inter-basin water transfer scheme under extreme and spatially extensive drought events. It is to explore the possible occurrence of trade-offs between water quality and quantity within the proposed transfer scheme. A new proposed inter-basin transfer scheme, the Severn-Thames Transfer, has been chosen to demonstrate the new modelling framework.

To achieve the aim, the following objectives have been identified:

1) to assess the impacts of climate change on river flows and water quality (dissolved organic carbon (DOC), nitrogen, phosphorus, and suspended sediment) in future droughts in River Severn and the River Thames (**Chapter 4 and 5**) as a baseline study for the application of transfer scheme;

2) to explore the long-term carbon dynamics in the headwater of the River Severn and investigate the possible risk to the Severn-Thames Transfer scheme (**Chapter 5**);

3) to identify the hypothetical operation rules of the Severn-Thames Transfer scheme by investigating water availability in the River Severn for a transfer during drought periods in the River Thames and the need for additional supply (**Chapter 6**);

4) to assess the impacts and risks on the water quality in the River Thames driven by the inter-basin water transfer (the Severn-Thames Transfer) (**Chapter 6**).

1.4 Thesis structure

This Thesis is comprised of seven chapters and follows a book format, a set of studies presented as a single narrative in a monograph, divided into chapters. This research presents and demonstrates a coupled modelling framework of water quantity, quality, and water resources options for the assessment of drought water resources planning under nonstationary climatic conditions. This method allows an understanding of how water quantity and quality can be linked together to evaluate the water management options, and to test the feasibility and effectiveness of the transfer options.

Chapter 2 presents a review of literature to explore the current understanding of drought impacts on quality and the hydrological and integrated catchment models as a water resources management tool. **Chapter 3** presents an overview of data and modelling tools that are part of the research framework, along with justifying the selection of the various tools, techniques used in the different stages of the thesis.

It is followed by three analytical chapters (**Chapter 4 to 6**) that present the results of the case study application. **Chapter 4** focuses on providing a thorough assessment of drought impacts on hydrology and water quality in two study catchments, the River Thames and River Severn. The water quality analysis focuses on Dissolved Organic Carbon (DOC), nitrogen, suspended sediment, and phosphorus in the Thames and the Severn. Nutrients (nitrogen and phosphorus) and suspended sediment are important water quality factors to pose serious threats to water supply in the River Thames. High nutrient concentrations can cause phytoplankton blooms in rivers, which in turn can clog filters and produce harmful toxins. This chapter covers carbon and eutrophication impacts on two rivers in droughts.

In terms of DOC, the issue here is that rising DOC in upland rivers in Wales (the headwater of the Severn) has been observed over the past decades. With the proposed water transfer, it is possible that high DOC water might be transferred across to the Thames, which might mean extra treatment is required to remove DOC prior to the water supply. This is a very expensive process and hence knowing the quality issues will be important for Thames Water and there could be ecological impacts on Thames ecology (Ritson et al., 2014). Therefore, **Chapter 5** explores long-term DOC concentrations in the headwater of the River Severn, Plynlimon, and investigate the effects of climate change and acidic deposition on DOC.

Based on the in-depth analysis in two catchments in **Chapters 4 and 5**, **Chapter 6** provides the modelling representation of the hypothetical inter-basin water transfer scheme, the Severn-Thames Transfer, under extreme spatially extensive drought events. With projected future river flows (in **Chapter 4**), future hydrological drought events are identified to assess how the characteristics of droughts and coincident droughts between two catchments would change under the projected climate change. Then this chapter explains how the Severn-Thames Transfer scheme would work if it is implemented in the future water resources system, and what would be the expected performance. Consequently, the risk of the inter-basin water transfer (Severn- Thames Transfer scheme) on water quality of the Thames are analysed.

Chapter 7 presents the conclusion of the thesis. Within the chapter, main findings from each chapter are reviewed, then followed by research contributions and suggestions for future research.

2 Literature review

There is a host of literature on hydrology and water quality, but the aim of this literature review is to consider the main factors controlling drought and water quality and to then address the background to flow and water quality modelling.

2.1 Drought and water quality

Drought is a climate event in which precipitation is less than expected in a particular location (Field et al., 2012). As drought is defined as a relative term, and there are several definitions from covering different perspectives and stakeholders' interests. In the scientific literature, three commonly distinguished definitions are the following: 1) meteorological drought, which indicates a deficit of precipitation, 2) soil moisture drought (also called agricultural drought), which indicates a deficit of root zone soil moisture, and 3) hydrological drought, which indicates negative anomalies in streamflow, lakes, and/or groundwater level (Heim, 2002). In addition, socio-economic drought occurs when the demand for an economic good exceeds supply as results of the elements or combination of meteorological, agricultural, and hydrological drought (Cook, 2017). In the UK, public water supply drought is often declared in England by water companies with respect to a shortage of water availability from their sources or infrastructures (Cook, 2017). Drought is a recurring feature of climate, and the effects of droughts are not linear.

The majority of droughts are induced by meteorological droughts (lack of precipitation combined with higher evaporation rates) (Wilhite et al., 2014) and eventually become hydrological droughts propagated via the hydrological cycle. As drought can adversely affect components of human systems, it considered a major hazard. Due to the complex

nature of drought, several indices have been developed to characterise drought and estimate its severity, intensity, and duration. These indicators involve using hydrological, climate, or land surface model simulations and measured meteorological or hydrological variables.

There are two groups of drought identification methods widely used: the threshold-based approach and the probabilistic-based approach. Both methods require a long time series of hydro-meteorological data. The first method, the threshold-based approach, identifies drought onset and termination when the variable of interest falls below or goes up to a predefined threshold (Sung and Chung, 2014; Van Loon and Van Lanen, 2012; Zelenhasić and Salvai, 1987). The threshold can be a constant value (e.g. a constant percentile of flow duration curves) or a variable threshold level (e.g. a value on the 80^{th} percentile of the flow duration curves of a 30-day moving window). The latter is more appropriate to reflect seasonality and has been widely used in recent studies (Fleig et al., 2006; Van Loon and Laaha, 2015). Most of the standardised drought indices follow the second method, the probabilistic-based approach. For example, it assesses drought intensity according to the deviation from the normal condition. Numerous drought indicators have been developed and widely employed, such as the Palmer Drought Severity Index (PDSI) (Palmer, 1968), the Standardised Precipitation Index (SPI) (McKee et al., 1993), Soil Moisture Drought Index (SMDI) (Hollinger et al., 1993) etc.

Some indices are only based on precipitation data. A broadly used index is SPI, in which a long-term precipitation record is fitted and transformed into a normal distribution to be standardised (McKee et al., 1993). The calculation of Consecutive Dry Days (CDD) is widely used for climate model simulations, which indicate the maximum consecutive

numbers of days per year with daily precipitation below the threshold (typically 1 mm/day) (Frich et al., 2002). Some other indices reflect both precipitation and actual or potential evapotranspiration, such as PDSI (Palmer, 1965), the standardised precipitation evapotranspiration index (SPEI) (Vicente-Serrano et al., 2010). The SMDI for the agricultural drought (Hollinger et al., 1993) and The Standardised Streamflow Index (SSI) for the hydrological drought (Barker et al., 2016; Shukla and Wood, 2008) are widely used to make a temporal and spatial comparison of the severity of droughts. Drought indices are effective tools that allow the quantification of drought characteristics, such as duration, frequency, and intensity, in relatively simple way.

Over the next 50 years, most of the world's major rivers are expected to experience more frequent hydrological drought relative to the historical records (Hirabayashi et al., 2008). Climate change is expected to alter the frequency and magnitude of droughts, and in turn, to reduce freshwater availability, and to affect freshwater quality (Whitehead et al., 2009). However, the extent of understanding of drought impact on water quality remains fragmented. Droughts can cause significant impacts on river water quality, with potentially severe consequences on water supply, aquatic ecosystems, and recreational use of water (Mosley, 2015). However, the value of water quality is rarely considered when analysing the economic impacts of drought (Freire-González et al., 2017).

Drought effects on water quality may be variable and complex, and they depend on many different factors. Several studies have been conducted to the systematic review of the effects of droughts on a large number of water quality-related variables on a global scale (Delpla et al., 2009; Lake, 2011; Mosley, 2015) and in focus on UK rivers (Johnson et al., 2009; Watts et al., 2015; Whitehead et al., 2009a). The majority of experimental studies

report the negative impacts on water quality during drought in rivers/streams and lakes/reservoirs by increasing water temperature, the occurrence of toxic algae blooms, salinity, and the reducing the nutrients concentration and dissolved oxygen (Mosley, 2015). Based on the current literature, this section covers the current understanding of drought impact on rivers for the water quality variables of interests in this study.

The most common impact on water quality during drought is increased temperature, and temperature increases during droughts have been observed in many rivers and streams (Davies, 1978; Ha et al., 1999; van Vliet and Zwolsman, 2008; Zielłński et al., 2009). Water temperature is one of the key parameters to determine the overall health of the aquatic ecosystem (Caissie, 2006). It exerts a fundamental control on the metabolisms and lifecycles of aquatic organisms (Elliott and Hurley, 1997), and consequently affects the water quality by changing the rates of the chemical and bacteriological processes (mostly accelerating at higher temperatures) (Whitehead et al., 2009b). For example, nitrification and denitrification in soils and streams, or sorption and desorption of nutrients, are governed by water temperature. The increased occurrence of algal blooms is strongly related to the temperature change (Whitehead and Hornberger, 1984; Whitehead et al., 2009b; Whitehead et al., 2015b). The degree of temperature change is highly variable. One case that did not find a significant temperature increase during a drought was where a large proportion of flow in a stream is being maintained by sufficient constant temperature groundwater (Wilbers et al., 2009). It is important to mention that water temperature changes vary throughout the year and hence impacts on chemical reaction kinetics will be less during colder winter conditions (Whitehead et al., 2009b).

The responses of nutrient concentrations in river/streams to droughts are diverse. The reduced catchment inputs and the lack of dilution capacity during droughts have led to the lower nitrogen and phosphorus concentrations and decreased turbidity as well (Jarvie et al., 2003; Mosley et al., 2012; van Vliet et al., 2013). In contrast, increasing nutrients during droughts have been observed in rivers where the main point sources (industrial, domestic, agricultural, effluent discharge) remain constant during drought due to a lack of dilution (Mosley, 2015). Water quality degradation is predominantly caused by the development of algae blooms which result from the combination of higher nutrient concentrations, higher water temperatures and longer residence times (Hellwig et al., 2017).

Mixed and counteracting drought responses between different nutrients are common due to the complex combinational influences of internal factors (such as biological uptake of nutrients and denitrification processes) and external factors (such as catchment inputs, land use, geology, and sewage effluents). Drought impacts on water quality tend to have a greater influence in the agricultural or pastoral catchments, which are the sources of chemical fertilisers and bacterial contamination (Caruso, 2002). In this case, impacts depend on the type of fertiliser application or diffuse sources within agricultural catchments which contribute to water quality in different ways. For example, Zielliński et al. (2009) found that increases in total phosphorus were not linked to changes in the total dissolved nutrients in agricultural and forest dominant catchments. Sprague (2005) reported increased nitrate and nitrite concentrations in urban and agricultural catchments, but ammonia and total phosphorus did not show notable changes during drought.

Dissolved oxygen and dissolved organic matters in rivers are also affected by drought. Reduced dilution capacity during low flows influences organic pollutants concentrations, with increased Biochemical Oxygen Demand (BOD) and consequently dissolved oxygen becomes depleted (Cox and Whitehead, 2009; Whitehead et al., 2009a). When an algal bloom occurs, there are large diurnal variations in dissolved oxygen and, on poor-quality rivers, low oxygen levels can be exacerbated by pollution events during summer low-flow conditions. Worrall and Burt (2004) showed that drought had a strong inter-annual control on carbon releases, and it decreased with drought intensity in lowlands rivers in the UK. Zielłński et al. (2009) also found out that dissolved organic carbon in most natural catchments decreased as a result of the lack of dilution capacity.

Concerning the post-drought phase, flushing of nitrate has been observed in many catchments, as a response to concentrations in the catchment during the drought and increased mineralisation and nitrification when dried soils are rewetted (Morecroft et al., 2000). Flushing of sediment during the drought termination has also been frequently highlighted in the literature (Bussi et al., 2016a). Increasing Dissolved Organic Carbon (DOC) concentrations in post-drought were found in the long-term monitoring data in the UK (Worrall and Burt, 2008). The extreme weather events (drought) could trigger occasional releases of DOC, flushing DOC in late summer (post-drought) and subsequent autumn storm events (Delpla et al., 2015). Higher DOC concentrations in drinking water can cause human health issues as DOC reacts with the chlorine used in some drinking water treatment processes to produce potentially carcinogenic trihalomethanes (Chow et al., 2003). Additionally, droughts cause several ecological perturbations, including

disruption to hydrological connectivity, loss of habitat and river ecosystem service (Lake, 2003).

In British rivers, according to Whitehead et al. (2009), nutrient loads are expected to increase under climate change. In the River Kennet (UK), for example, Whitehead et al. (2006) forecast an increase in nitrate and ammonium concentrations due to increased soil mineralisation, although they highlighted that this is particularly significant under high flows conditions following a drought, when the nitrogen is flushed out of the catchment soils. Crossman et al. (2013) and Bussi et al. (2016c) showed that climate change is expected to increase phosphorus concentrations in the River Thames (UK) due to a decrease in the effluent dilution capacity, while at the same time changes in land use, land management and wastewater treatment strategies might heavily affect the river's response to climate change. Bussi et al. (2016a) found that the sediment transport in the River Thames is likely to decrease due to reduced runoff, reduced soil erosion and decreased stream transport capacity. However, this effect could be reversed under the following scenarios: (i) an increase in extreme precipitation and/or (ii)an expansion in agricultural land.

Important aspects of hydrological droughts and their impacts include the persistence and pre-conditioning of aquatic storage and systems. Although the complex interaction between changing climate and anthropogenic land use interventions could result in different outcomes depending on the catchment conditions and management practices (Van Loon et al., 2016). In general, droughts and the immediate recovery period were found to have profound effect on water quality, and those impacts are diverse depending on the characteristics of the water body and its catchment (Bussi et al., 2017). Thus the

extent and magnitude of climate change and droughts impacts on water quality must be assessed locally.

2.2 Hydrological and Integrated catchment (water quality) modelling

River systems are highly dynamic, complex, and strongly hierarchical systems as each confluence of rivers is the sum of its upstream processes, which makes each river systems unique (Poole, 2002). Naturally, rivers have been investigated as a catchment functioning perspective (Allan, 2004). A catchment can be defined as an area of land that receives precipitation and drains off into the river network. The catchment is a hydrological response unit and a holistic system block to understand system behaviour where water flows into the river with unique catchment information. Therefore, the catchment has been used for not only scientific analysis but also socio-economic and political aspects that link to the management perspective. Hydrological connectivity within the catchment, such as rainfall, surface water, subsurface water, soil zones, land use, vegetation, and anthropogenic factors (sewer pipes, ditches), can be described in hydrological models with mathematical equations to represent rainfall-runoff system dynamics. The types of model can be classified as conceptual, empirical, mathematical, or physical-based depending on the representation of physical process complexity. With advancements in modelling techniques, simulation accuracy and the ability to predict hydrological processes (including runoff generation, evapotranspiration) have been enhanced. The spatial resolution of the hydrological model can be classified as lumped, distributed, and semi-distributed. Lumped models consider a catchment as uniform state variables, while distributed models simulate each of the divided catchments as model elements to

maximise model heterogeneity. Semi-distributed models are the most widely applied, which involves aggregates of lumped models.

Surface water quality models and catchment models are simplified versions of natural processes. The first water quality model was built by Streeter and Phelps (Phelps and Streeter, 1958) to control river pollution in Ohio, in the US. Since then, the simple oxygen model (the relationship between BOD and DO) by O'Connor (1967), nitrogen and phosphorus cycling and phytoplankton models (Yih and Davidson, 1975) are followed. A rapid improvement to the water quality model over past decades, at this stage, water quality models are capable of simulating water quality variables with consideration of multiple pollution sources, including diffuse and non-diffuse components, wet and dry atmospheric pollutant depositions in the catchment (for example, QUAL2K (Chapra and Pelletier, 2003), WASP models (Di Toro et al., 1983), QUASAR (Whitehead et al., 1997), SWAT (Arnold et al., 1998), MIKE models (Danish Hydraulic Institute, 1991)).

Types of water quality model have been evolved in order to respond to water quality problems and stakeholders' needs. The steady-state water quality models have been widely used for understanding water quality problems for the long-term with required routine monitoring. The dynamic and stochastic model applications look at relatively short-term water quality dynamics to continuously monitor the water quality system (Whitehead, 2016). There are relatively few water quality models that all of river channel dynamics, soil and groundwater components are incorporated as a whole catchment. **IN**tegrated **CA**tchment model, INCA model, was developed to include the three compartments (stream, soil zone, groundwater zone) and models are running process-based and semi-distributed spatial domain (Whitehead et al., 1998a). The concept of mass

balance is used for a fundamental basis of INCA modelling, and reaction kinetics equations are adopted to simulate the principal mechanisms in changes of water quality variables.

Hydrological models become powerful tools and can serve many purposes of applications, including climate change impacts, risks assessment and human impacts on the hydrological system. The advantage of hydrological models is that they can be coupled to the water quality modelling, climate modelling, or other environment modelling has hydrological components. Hence, having a specific problem and solution is necessary when employing modelling approaches. Dynamic, process-based integrated catchment models are techniques for formalising current knowledge of complex catchment dynamics. Hence, it is widely accepted that catchment models are useful catchment management tools. Once a model can capture the dominant modes of behaviour in a system, it can then provide science-based evidence to support decision making.

Although the extant literature is predominantly focused on water quantity in water resources management, in the last decade, combined modelling approaches to water quantity, quality and water resources system have received greater scholarly attention. For example, several studies have involved the development of integrated quality-quantity approaches in examining the current operation rules to improve water quality in river basins by changing water allocations (Paredes et al., 2010, Zhang et al., 2010). Yuan et al. (2015) also proposed a joint-operation model of dams and floodgates to simulate water quality and quantity simultaneously to balance flood control and pollution interventions.

The aforementioned studies drew on historical data to investigate improvements in current operation rules. While these works contributed to our understanding of the need for integrated approaches, the potential risks and trade-offs between water quality and quantity, and the reliability of water supply under changing climates and extreme droughts, remains under-researched. Mortazavi-Naeini et al. (2019) started filling the gap by introducing a coupled modelling system that combines hydrological modelling, surface water quality modelling, water resources systems modelling and simulations of algal blooms in storage reservoirs within a catchment. This coupled modelling framework successfully identified and quantified the impact of extreme droughts, water demand growth, and land-use change on the sustainability of public water supplies. The results implied that effluent from upstream sewage treatment works and diffuse pollution in the catchment could potentially be risks to the reliability of public water supplies.

It is difficult to address all sources of model and parameter uncertainty in hydrological and water quality modelling. Models are a simplified version of reality and allow an approximation of any system to be quantified and evaluated (Whitehead, 2016). Especially, water quality modelling attempts to rep resent a system that is dynamic, and chemical processes that are kinetic in nature and interacting with hydrology. This makes modelling a very complex process. Furthermore, the uncertainty is cascading and propagates along with model chains. Extensive studies of uncertainty using the water quality models, INCA models, have been done. The general sensitivity analysis (Hornberger and Spear, 1980; Spear and Hornberger, 1980) has applied to test INCA model uncertainty to evaluate both model parametric uncertainty and the uncertainty in the data used to drive model simulations (Wade et al., 2002a, Wade et al., 2002b; Wade

et al., 2002c). A generalised likelihood uncertainty estimation (GLUE) methodology has also been used to quantify uncertainty within INCA-N (Rankinen et al., 2006) and INCA-P (Dean et al., 2009). GLUE framework can be coupled with other calibration techniques, and Rankinen et al. (2006) combined GLUE analysis with quantitative experimental knowledge (also known as 'soft data') of the processes to judge whether 'soft data' improves the robustness of the INCA-N parameters. Dean et al. (2009) insisted that the degree of uncertainties in the observed data from the river and the uncertainty associated with the model structure and parameters are similar. Observed data uncertainty cannot be underestimated. For example, the observed water quality data normally comprise of weekly, fortnightly, or monthly samples, which are collected at a particular time on a particular day of the month. Then it is typically compared with daily time-step model simulations. Furthermore, the observed data uncertainty always exists due to probable errors in sampling, handling, and analytical methods (Jarvie et al., 2002). Thus, treating this data as a summary of the whole month may mislead the understanding of the system.

Nevertheless, uncertain models are still a useful tool and can provide beneficial information for planners and managers. However, it should be used carefully to evaluate the impact of future environmental scenarios, highlighting that the percentage change matters more than the absolute values (Cosby et al., 1986, Bussi et al., 2017).

3 Data and models

Water managers and researchers have been quick to identify the challenges of future uncertainty and competing objectives whilst preparing for climate change and population growth (Whitehead et al., 2015a). Among many approaches and attempts to respond to increasing water-related risks to people and the environment, simulation models offer an appropriate tool for representing the complex nature of the hydrological cycle and water resources system. Simulation modelling can facilitate the exploration of possible near and far futures and testing of alternative management options. Furthermore, it enables the quantification of observable outcomes for future scenarios. This doctoral study is focused on the Severn -Thames Transfer scheme, which is a potential water transfer scheme being actively considered. This chapter reviews the data and modelling tools used in **Chapter 4,5,** and **6.** The work recorded in this thesis brings together a range of different datasets and models, the outputs of which are used as inputs to others. Although each analytical chapter will include a brief description and overview of the methods and materials employed, this section attempts to provide a thorough overview. This chapter will start to give overview information of the two study catchments, the River Thames, and the River Severn, followed by the selection of modelling tools and data.

3.1 Study catchments

3.1.1 The River Thames system

The River Thames is located in Southern England in the UK. The study catchment is from the headwaters at Cricklade down to the Teddington Weir at Kingston which is the downstream tidal limit. The catchment covers approximately 9,948 km^2. The Thames supplies water to about 10 million households, as well as 215,000 businesses, and the area

includes London as well as other urbanised cities such as Oxford, Swindon, Slough, Reading and Maidenhead (Thames Water, 2020a). Despite the high human population density (ca. 960 people per km^2) (Merrett, 2007), land use in the uplands and the western Thames is relatively rural, predominantly arable land and grassland (Environment Agency, 2016) (Figure 3.1). The catchment is underlain by limestone and clay/mudstone in the upper reaches and a chalk across the downstream, so that the River Thames can be classified as a moderately groundwater-dominated stream (Whitehead et al., 2015b). The base flow index at the end of the freshwater boundary at Kingston is 0.63 (from National River Flow Archive (NRFA), https://nrfa.ceh.ac.uk/). The base flow index is an effective means of indexing catchment geology and a measure of the proportion of the river runoff that derives from stored sources (Institute of Hydrology, 1992). The high base flow index reflects the high groundwater component in the river system. According to the NRFA, the Thames receives an average annual rainfall of 725 mm (computed in the period of 1961-2015 at Kingston, with a minimum of 540 mm in 1973 and a maximum of 977 mm in 2000). The mean daily air temperature is 10.1 °C. The average summer temperature is 16.4 °C, and the average winter temperature is 4.6 °C (computed in the period of 1961-2015 with data from Met Office, 2012). Flow in the Thames varies throughout the year, with high flows in winter between December and March of 85.8 m^3/s (50^{th} percentile) and low flows in summer from June to September of 16.1 m^3/s (50^{th} percentile) (computed in the period of 1961-2015).

3.1.2 The River Severn system

The River Severn is the longest river in Britain running from the source at Plynlimon in mid-Wales to the commencement of its tidal estuary system near at Haw Bridge, near

Gloucester in western England. The study catchment covers approximately 10,512 km^2 from Plynlimon to the freshwater boundary on the Severn. The Severn undergoes a sudden change in its direction (almost 90°) as it crosses the border and flows south towards the Severn Estuary. The catchment hydrology is dynamic and complex. The headwater, Plynlimon, can be described as a composition of peatland, semi-natural grassland, and conifer forests. It also has steep profiles; thus a 'flashy' catchment is created due to the high elevation of the Cambrian Mountains plateaux. Large spatial variations in precipitation between the uplands and the lowlands are observed in the catchment. According to the NRFA, the upper Severn receives an average annual rainfall of 1,179 mm (computed in the period of 1961-2015, with a minimum of 876 mm in 2010 and a maximum of 1,610 in 2000) and the lower Severn receives an average annual rainfall of 812 mm (1961-2015, with a minimum of 589 mm in 2011 and a maximum of 1,078 in 2000). The high-rainfall and flashy head catchment carry the water and sediments to the middle and lower Severn, where the river meanders the extensive floodplains underlain by permeable rocks and sandstones. As mostly arable farming catchments in the middle and lower Severn (Figure 3.1) and the lower rainfall, the additional discharge is suppressed and regulated. Two major tributaries, the River Avon (from the east) and River Teme (from the west), join the lower Severn and add almost 60% to overall mean flow in just a 30 km length reach. A long-term mean flow at Bewdley in the Severn is 61.4 m^3/s (the period of 1921-2019), and the mean flow at Deerhurst is 100.2 m^3/s (the period of 1995-2012) after two tributaries' confluences (NRFA, https://nrfa.ceh.ac.uk/). The mean daily air temperature is 9.5 ℃. The average summer temperature is 15.1 ℃, and the average winter temperature is 4.5 ℃ (computed in the period of 1961-2015 with data from Met Office, 2012). Severn Trent Water is a water company responsible for

supplying water within the basin and the River Trent basin, and supplies about 8 million people and businesses to many towns in the Midlands, including Strewsbury, Birmingham, Mansfield, Nottingham, and Warwick (Severn Trent, 2019). The extent of floodable land and flood risks has been a significant challenge in the Severn water resource management.

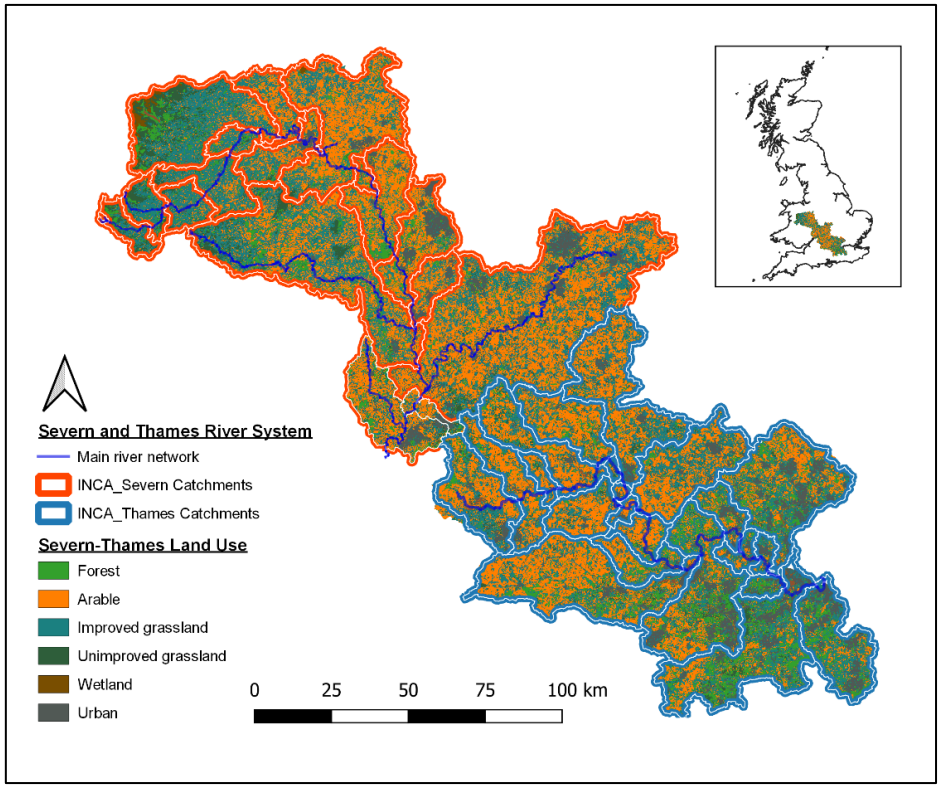


Figure 3.1 Land use map of case study area (The River Thames and River Severn)

3.2 Climate data

The assessment of drought impacts based solely on historical events data is strongly limited by the shortage of severe events within the observational records. Climate models are valuable tools as they allow generating a larger sample of extreme events and projecting changes in the future. In studies of the impacts of climate change on hydrology

and water resources system, the projected response of the climate system can be obtained from global climate models (GCMs) with outputs that include precipitation, temperature, energy fluxes or other hydrological variables. As the coarse spatial resolution of GCMs (a wide range from 25km to typically 100-150km horizontal resolution), the direct application of GCMs projections to catchment scale is not suitable and downscaling to a finer resolution is necessary (Dadson et al., 2019). Regional climate models (RCMs) are widely used as the dynamic downscaling methods for a designated region, which produce high-resolution simulation at a finer resolution (from 1km up to 25km) within GCMs output boundaries. Dynamic downscaling is often required to the intensive computation power. Alternatively, statistical downscaling methods are available with efficient computation and simple data manipulation, and typically used where variables of interest can be downscaled using historical observations (Wilby and Dawson, 2013, Maurer and Hidalgo, 2008).

Previously, the UK Climate Projections 2009 (UKCP09) used for the UK-based hydrology and water quality impact studies (Bussi et al., 2016a; Bussi et al., 2016b; Jin et al., 2012; Lu et al., 2015) or previous version the UK Climate Impacts Programme 02 (Cox and Whitehead, 2009; Whitehead et al., 2009a). UKCP09 projections provide probabilistic distribution of climatic variables in temperature, precipitation and evapotranspiration relative to the baseline climatology, using HadAM3P GCM and a selection of emissions scenarios from IPCC (Intergovernmental Panel on Climate Change) (Murphy et al., 2009). A statistical model (stochastic weather generator) is normally used alongside and at least 100 samples from the full 10,000 projections for each emission scenario and time periods are recommended to employ these scenarios. However, the

structural uncertainty of the modelling framework increases by cascading methodological steps (Wilby and Dessai, 2010).

The methods presented in this thesis rely on the modelling system, weather@home (w@h, Guillod et al., 2017; 2018; Massey et al., 2015), which consists of a global climate model (GCM, HadAM3P) with a nested regional climate model (RCM, HadRM3P). They are driven by historical and projected sea surface temperatures (SST) and sea ice, allowing for the generation of a very large ensemble of weather events with the help of volunteers (members of the public) and their distributed computing networks. In the model applications, weather@home 2 (w@h2, which includes an improved land surface scheme) was used to generate 30-year-long weather time series of rainfall and temperature projections for three different time periods. The w@h model can generate only relative short simulation results using volunteers' personal computing networks. Therefore, a stitching method was developed to formulate a longer time series (i.e. 30-year long) from a short, single-year simulation. The stitching process was done based on the soil moisture content for the w@h2 data set. As soil moisture exhibits memory of a few weeks to months, it is essential to keep this memory for the case of droughts. In addition, the temporal soil moisture is mainly driven by precipitation and evapotranspiration, it can be used for an approximate integrator of temporal precipitation and evapotranspiration.

The w@h is process-based; thus, the physical forcing processes can be better represented throughout space, and the output weather sequences are more spatially consistent than those produced by weather generators (UKCP09). The w@h2 data set is specifically designed to support a risk-based approach for the study of extreme events such as drought and heavy precipitation, both of which require the spatial consistency of hydro-

meteorological data sets (Guillod et al., 2018). The scenarios in each future time slice all follow the Representative Concentration Pathway 8.5 (RCP8.5) and sample the range of sea surface temperatures and sea ice changes from CMIP5 (Coupled Model Intercomparison Project Phase 5) models. In w@h2 dataset, a number of daily and (or) monthly output variables are available including temperature, precipitation, surface air humidity, mean sea level pressure, and potential evaporation. The following three grouped ensembles are used to project the future impacts on flows and water quality variables in this thesis:

i. Baseline: 100 realisations of a 30-year (1975–2004) period, with different initial atmospheric conditions to create the ensemble but each using the same historical SST and sea ice observations from the HadISST (Hadley Centre Sea Ice and Sea Surface Temperature data set) (Rayner et al., 2003, Titchner and Rayner, 2014);
ii. Near Future: 100 realisations of a 30-year (2020–2049) period, which are driven with the 50th percentile SST and sea ice projections from CMIP5 (Taylor et al., 2012) for the RCP8.5 greenhouse gas concentration scenario (Meinshausen et al., 2011);
iii. Far Future: 100 realisations of a 30-year (2070–2099) period, which generated using 50th percentile SST and sea ice from CMIP5 under RCP8.5 emission scenarios.

All realisations have been generated from the same climate model and intended to signify multiple samples of climatic variability. Thus, they could be considered to be equiprobable. RCP8.5 is the upper bound of projected global emissions scenarios. Consequently, the w@h2 projections are able to cover more severe synthetic drought

events than the historical droughts (Guillod et al., 2018), which enables to assessment of the water resources options under wider range of extreme events.

The precipitation data was bias-corrected in order to reproduce the patterns and distributions of observed data better, and a linear approach was applied (Guillod et al., 2018). Figure 3.2 shows the monthly average of w@h2 precipitation (Figure 3.2(a)) and w@h2 temperature (Figure 3.2(b)) and compared with the observation. A very little bias was observed from temperature, so temperature bias was not explicitly corrected. The bias correction method was conducted by using spatially-variable monthly bias-correction factors, obtained by the ratio of the modelled (w@h2) and observed monthly average precipitation (CEH GEAR (Gridded Estimates of daily and monthly Areal Rainfall) , Tanguy et al., 2016). For each month, the ratio was computed over all the cells of the RCM domain, based on the period of 1961-2006. The resulting spatial field of values was smoothed spatially with a 3x3 cells running window, and temporally with a 1-2-1 monthly running filter. The bias-corrected w@h2 precipitation provides an accurate estimation of the actual precipitation over the Thames catchment (Figure 3.2(a)). More details about w@h2 bias-correction method can be found in Guillod et al. (2017) and Guillod et al. (2018).

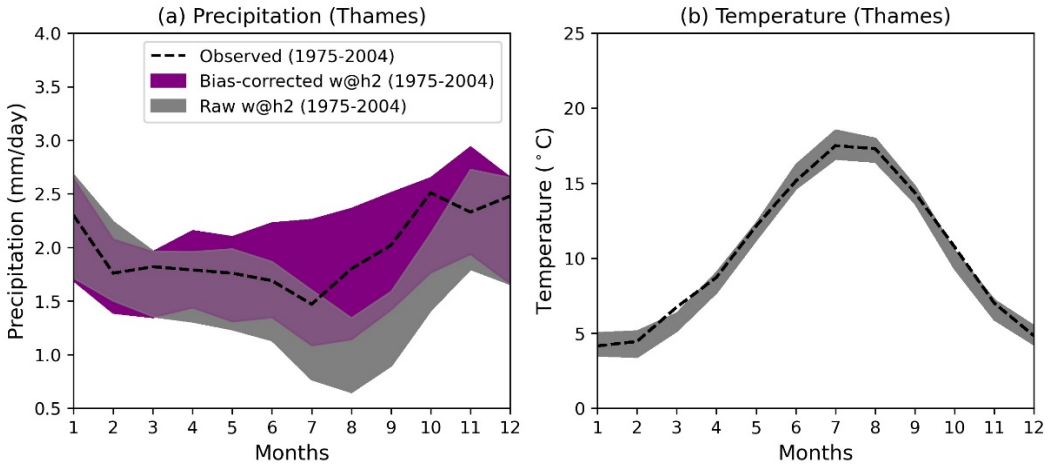


Figure 3.2: Bias-correction of the w@h2 monthly average precipitation (a) and validation of the w@h2 monthly average temperature (b) in the Thames catchment (Observation data source (1975-2004) : precipitation records are taken from CEH GEAR (Tanguy et al., 2016) and temperature records are taken from Met Office (2012)).

Several application of w@h climate ensembles have been used to assess the impact of climate change on water resources system modelling in the Thames Basin (Borgomeo et al., 2018; Mortazavi-Naeini et al., 2019) and in England and Wales (Dobson et al., 2020; Murgatroyd and Hall, 2020). Borgomeo et al. (2018) and Dobson et al. (2020) thoroughly validated the w@h2 dataset by comparison to historical records (precipitation from CEH-GEAR (Tanguy et al., 2019) and potential evapotranspiration from CHESS (Climate Hydrology and Ecology research Support System) (Robinson et al., 2016)) and w@h2 modelling results. Temperature and precipitation of baseline scenarios have been reproduced using historical observations, and the near and far future ensembles include extreme droughts not presented in historical records. Therefore, w@h2 climate data will be capable of providing a wider range of possible impacts on catchment hydrology and water quality. Of particular relevance for water quality modelling are the changes in intense rainfall, which can affect leaching and drought, which influences chemical retention times and river flows. Recently, the w@h climate projection has been used for the UK-based water quality impact studies (Bussi and Whitehead, 2020, Lee et al., 2020).

3.3 Water quality modelling

Water quality modelling has been widely accepted as a reliable method to inform the development of environmental policies and implementation. This is because water quality models are capable of estimating the contribution of various sources to an environmental problem and of calculating the expected effects of a measure. In this study, the Integrated Catchment Modelling System INCA has been utilised, as it provides a process based and dynamic simulation system for representing flow and water quality.

The ***In****tegrated* ***Ca****tchment* model (INCA) is a process-based model that represents the elements and processes governing flow and water quality dynamics in both the land and in-stream components of river catchments. The INCA model simulates and calculates daily estimates of river discharge, in-stream water quality concentrations and fluxes along with a river network. The model is semi-distributed, so the spatial variations in land use and management could be fully considered in the calculations. It was developed initially to investigate the nitrogen dynamics in the catchment scale (Whitehead et al., 1998a; Whitehead et al., 1998b). Subsequently, the INCA framework has been extended to phosphorus (INCA-P, Wade et al., 2002b, Jackson-Blake et al., 2016), carbon (INCA-C, Futter et al., 2007), sediments transport and soil erosion (INCA-Sed, Lazar et al., 2010), pathogens (Whitehead et al., 2016) and an organic contaminant model (Nizzetto et al., 2016).

3.3.1 INCA hydrological model

The INCA model includes a rainfall-runoff component; thus, it simulates the hydrological process from rainfall to streamflow. The hydrological model provides information on the flow moving through the soil zone, the groundwater zone, and the river system. The

hydrological model is conceptualised to a simple two-box system with hydrologically effective rainfall moving through the soil system and then either recharging the groundwater system or leaching into the river. The groundwater flows are also routed to the river reaches after a suitable delay controlled by a residence time. INCA model requires a daily time series of soil moisture deficit (SMD), hydrological effective rainfall (HER), air temperature and precipitation as inputs. The first two input data were estimated by a semi-distributed rainfall-runoff model, named PERSiST (Precipitation, Evapotranspiration and Runoff Simulator for Solute Transport model). This is because the INCA does not include an evapotranspiration module. PERSiST simulates water fluxes in a catchment scale, with a series of user-defined buckets indicating different hydrological processes such as snowmelt, surface runoff generation, soil water storage, groundwater and streamflow. It is particularly designed to provide input time series for the INCA models (Futter et al., 2014). In addition, spatial data describing catchment land cover and soil properties, the location of point sources, abstraction and effluent discharge are required to run the INCA hydrology module.

3.3.2 INCA- Nitrogen model

Fluxes and concentrations of water quality concentrations are simulated by solving mass balance equations for terrestrial processes whilst simultaneously solving flow equations. The mass balance equations for both nitrate-nitrogen and ammonium-nitrogen in both the soil and groundwater zones are processed while solving the flow equations. The key processes that require modelling in the soil zone are plant uptake for NH_4-N and NO_3-N, ammonia nitrification, denitrification of NO_3-N, ammonia mineralisation, ammonia immobilisation and N fixation. All processes will vary to the types of land use, and a set

of governing equations is required for which parameter sets can be derived for different land uses. The land phase model must also account for all the inputs affecting each land use, including dry and wet deposition of nitrogen and fertilisers used in agriculture. Temperature and soil moisture are key control factors to certain processes. Nitrification reactions are temperature dependent, and denitrification and mineralisation are both temperature and soil moisture dependent. Nitrogen (nitrate and ammonium) is then routed to the river network, where nitrification and denitrification processes occur and where point sources pollutants are discharged, such as effluents. More information about modelling equations and processes can be found in Whitehead et al. (1998a) and Wade et al. (2002a).

3.3.3 INCA- Phosphorus and sediment model

The phosphorus sub-model of INCA is initially developed in 2002 (Wade et al., 2002b), based on the same structure as the nitrogen version of INCA (Whitehead et al., 1998a). It is a process-based representation of processes controlling phosphorous dynamics in soils, stream water, and bed sediments of river catchments. The relevant processes include sorption and release, plant uptake, solute, and particulate exchange. The model accounts for organic and inorganic phosphorous stocks in the soil and groundwater pools, and total phosphorous and soluble reactive phosphorus in the stream reaches. The model also simulates the flow of water through the plant/soil system from different land-use types to deliver the phosphorous load to the river system, which is then routed downstream after accounting for direct effluent discharges, and in-stream processes.

The sediment sub-model of INCA is a representation of soil erosion and sediment transport to the stream and subsequent entrainment and deposition (Lazar et al., 2010).

The soil erosion process is divided into splash detachment and flow erosion of bulk sediment, based on rainfall intensity and direct runoff, and the detached soil is routed into the river network where five grain size classes are considered separately for in-stream processes. The sediment and phosphorus sub-models are interconnected due to the in-stream adsorption processes of phosphorus. The improvement of the model has been made by incorporating INCA-sediment module to a new version of INCA-P (version 1.4.4 or above), and a full model description is provided in Jackson-Blake et al. (2016).

In this Thesis, INCA-P version 1.4.10 was used to simulate total phosphorus and suspended sediment concentrations. Although this version of INCA-P is able to simulate the macrophyte, epiphyte and algal biomasses, the ecological impacts were not predicted and discussed here due to the lack of data available in both Thames and Severn catchments.

3.3.4 INCA- Carbon model[1]

The INCA-C is a dynamic, semi-distributed catchment scale process-based model of DOC concentrations and fluxes in surface waters (Futter et al., 2007). The carbon model is based on a similar structure to the other versions of INCA with terrestrial and aquatic compartments. Within the terrestrial compartments, there are an organic and mineral soil layers to facilitate the modelling of carbon dynamics at the soil surface and within the sub-soil. The aquatic compartment represents any water body. Within each compartment

[1] Detailed processes are explained here as the modification of model parameters will be introduced in chapter 5.

or layer, there are four carbon pools. These are: PDC (potential dissolved carbon); SOC (soil organic carbon); DOC; and DIC (dissolved inorganic carbon).

INCA-C considers both living and abiotic carbon in the model simulation. Sources of terrestrial PDC are leaf litter, root exudates and the soil micro-flora. Aquatic PDC includes leaf litter from the terrestrial compartments, particulate organic carbon in the water body and the aquatic biota. The abiotic component of PDC consists of particulate organic carbon (POC). SOC is only present in the soil layers and SOC comprises all the organic carbon that is unavailable for mobilisation out of the terrestrial compartments. As such, it includes organic carbon bound to the mineral and clay constituents of the soil and the microbial community attached to the soil substrate. Carbon in the SOC pool may be adsorbed to the soil, bound to iron or aluminium minerals, held by ligand exchange or retained in the soil by electrostatic forces. SOC levels are typically much higher in the organic soil layer than the mineral soil layer.

The DOC pool consists of humic substances, operationally defined as *"a continuum of organic molecules of different sizes and structures that pass through a filter of 0.45μm pore size"* (Kalbitz et al., 2000). Unlike SOC, DOC may be lost from compartments or transferred between soil layers through convective flow. DIC consists primarily of CO_2 dissolved in soil water in the terrestrial compartment or aquatic compartment. In neutral or basic systems, the DIC pool may include carbonate species. Minor but non-trivial amounts of methane (CH_4) may also be present, especially when compartments are anoxic. The mass, transformations and fluxes of each of the four carbon pools can be modelled in the organic soil compartment. The mineral soil compartment includes SOC, DOC and

DIC pools. In the aquatic compartment, PDC, DOC and DIC can be/is modelled, as shown in Figure 3.3.

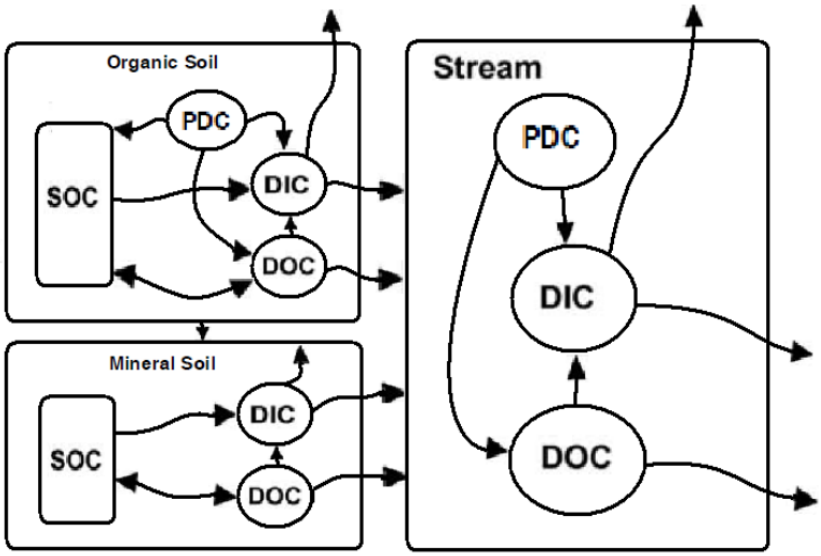


Figure 3.3: Pools of carbon in terrestrial and aquatic compartments in INCA-C (modified from Futter et al., 2007)

In the organic soil layer (Figure 3.3, Organic Soil), there are input fluxes in of PDC (primarily through litter-fall) and SOC (through organic fertiliser additions). PDC may be lost from the organic soil to the aquatic compartment through overland flow. Both DOC and DIC are lost from the organic soil layer to both the mineral soil and the aquatic compartment. PDC may be transformed to SOC or DOC through decomposition, humification or leaching. While some PDC will be broken down to CO_2, it is not modelled since it is not expected to enter the soil water solution. SOC may be transformed to DOC, through decomposition, de-sorption, dissolution or decomplexation. SOC may also be transformed to DIC through microbial decomposition. DOC may be broken down to DIC by chemical or biological processes. It may also be transformed to SOC through precipitation, adsorption, complexation, protonation, or microbial uptake. In some

anaerobic environments, DIC may be transformed to SOC through microbial respiration. The mineral soil (Figure 3.3, Mineral Soil) receives incoming fluxes of DOC and DIC from the organic soil layer. Both DOC and DIC are lost from the mineral soil through convective transport to the aquatic compartment. The same fluxes between SOC, DOC and DIC pools that are included for the organic layer are modelled for the mineral soil layer. However, the rates of transformations, especially for SOC and DOC to DIC are higher in the mineral soil than the organic soil layer. The aquatic compartment (Figure 3.3, Stream) receives PDC, DOC and DIC inputs from the terrestrial compartment. When functional units have been assembled into a network, the aquatic compartment may also receive inputs of the three previously mentioned carbon pools from upstream reaches. PDC and DOC may be lost from the aquatic compartment through sedimentation or convective transport to downstream reaches. DIC is lost through convective transport and through atmospheric evasion as CO_2 or CH_4. Abiotic PDC is transformed to DOC through the same processes driving SOC to DOC transformation in the terrestrial compartments. Biotic transformation of PDC to DOC occurs mainly because of death of the micro-flora. PDC is also transformed to DIC through both biological and chemical processes. DIC is transformed to biotic PDC through photosynthesis and anaerobic respiration. Further details of model equations can be found in Futter et al. (2007) and Lee et al. (2020).

3.4 Water resource system model information

Water supply in London is principally from direct abstractions from the River Thames to a series of London reservoirs, with additional water supplied from the underlying chalk aquifer (Thames Water, 2014). A water company, Thames Water Utilities Ltd, is responsible for the water supply in the London water resources zone. Surface water

abstraction to meet public water demand is subject to a maximum limit, which is set to maintain environmental flows. The Lower Thames Operating Agreement (LTOA) regulates the operation of the London water system to maintain a healthy aquatic environment, with limiting abstraction and restricting environmental flows that the amount of water that can be withdrawn from rivers and reservoirs in the system. In each month, the total volume of water stored with the London reservoirs determines the level of restrictions to water users and the thresholds are defined in the Lower Thames Control Diagram (LTCD, Figure 6.3 in Chapter 6).

In order to incorporate the existing water system regulatory rules into the water resource modelling, the runoff values generated with the INCA hydrological model are used as inputs to the WATHNET. A water resource system model, WATHNET (Kuczera, 1992), is used to represent London's urban water supply system. WATHNET is a generalised node-arc simulation model using a network flow programming and has been applied in many water resources planning applications (Mortazavi-Naeini et al., 2014; 2015). WATHNET was chosen because it includes a user-friendly interface, efficient computation time and flexible scripting for water system rules and constraints. Furthermore, this model is already well set up for the Thames basin (Borgomeo et al., 2018; Dobson et al., 2020; Mortazavi-Naeini et al., 2019). In the model simulation, a water shortage occurs every time the reservoir level falls below the reservoir level thresholds defined in the LTCD. For each model run, daily storage level, level of services and supply shortages were recorded, and these outputs were taken to develop the hypothetical Severn-Thames Transfer operation rules. This model setup was provided by Mortazavi-Naeini et al. (2019).

3.5 Summary of data and models

Figure 3.4 shows the summary of modelling selection and data used to drive each modelling process in the Thesis.

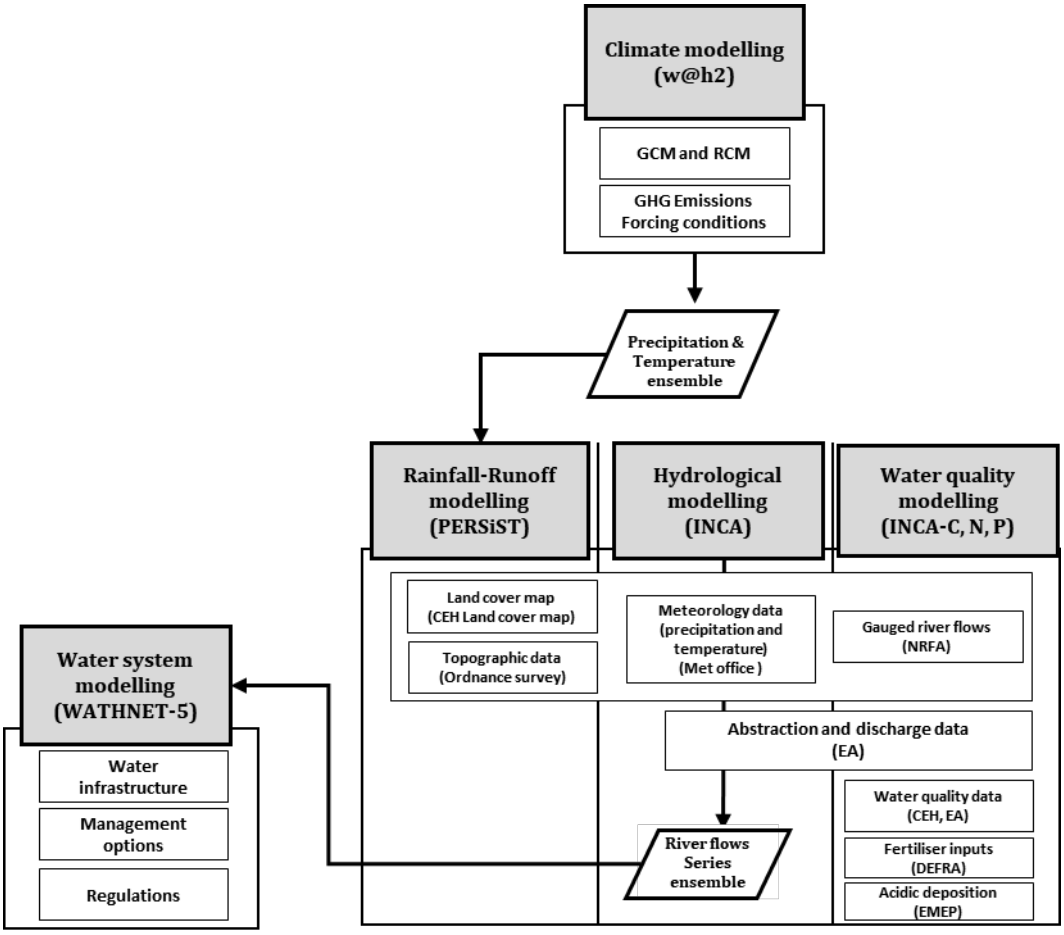


Figure 3.4 Modelling framework and data used in this study

4 Drought and climate change impacts on river flow and water quality of the River Thames and River Severn systems

4.1 Introduction

As established in **Chapter 2**, droughts and low flows can cause significant impacts on surface water quality, with potential severe consequences on water supply with additional treatment costs, on aquatic ecosystems, and on the recreational use of water (Mosley, 2015). Water quality responses to droughts are diverse and localised study is necessary. The aim of this chapter is to provide an assessment of the impact of drought and climate change on river flows and water quality to improve understanding of catchments responses. Carbon, nutrients, and suspended sediment concentrations are used to characterise river water quality. The study has been focused on two catchments in the UK, the River Thames, and the River Severn, where the catchments are subject to a strong anthropogenic pressure due to extensive agricultural landscape and a large population living its boundaries and downstream. Ultimately, this chapter results are to provide the baseline for the proposed transfer scheme.

The specific objectives are:

1) to present a methodological modelling approach for the analysis of the impacts of drought and climate change on the river systems;
2) to assess and quantify potential impacts of droughts on river flows and water quality (dissolved organic carbon, nitrate, total phosphorus, and suspended sediment) of the River Thames and Severn.

4.2 Methodology

4.2.1 Study catchments

This study investigates the hydrological drought in South England and subsequent impacts on water quality in the River Severn and the River Thames. The detailed catchment characteristics can be found in the previous chapter (Chapter **3**), and more information about modelling setup for two catchments will be following in section **4.2.3**.

4.2.2 Model data

Table 4.1 shows a summary of the data used in this chapter. The meteorological data (daily precipitation and temperature) of two catchments were obtained from the UK Met Office (Met Office, 2012). Records of continuous daily discharge at the multiple sections of the River Thames and Severn were obtained from the National River Flow Archive (NRFA, https://nrfa.ceh.ac.uk/). Weekly or fortnightly water quality data were obtained from the Thames Initiative (TI) platform database (Bowes et al., 2012), managed by the UK Centre for Ecology and Hydrology (CEH) and by from the Water Information Management System (WIMS), managed by the Environment Agency of England and Wales. Abstraction and Discharge data in both catchments are obtained from the Environment Agency. All abstractions and discharges in a catchment are aggregated and applied as a constant for the simulation period. Statistics of fertiliser usage and number of grazing animals (for manure inputs) were obtained from the Department for Food and Rural Affairs (DEFRA) to calculate a phosphorus daily application rate.

Table 4.1: Data used in this study for model setup and calibration.

Type	Source	Spatial coverage	Time coverage
Land use – UK Land cover map	UK CEH (Centre for Ecology and Hydrology)	River Severn and River Thames- All catchments	2007
Topography	Ordnance Survey	River Severn and River Thames- All catchments	-
Meteorology (precipitation and temperature)	Met Office	All open stations for both catchments (interpolated on a 5x5 km grid within this study)	2000-2014
Flows	National River Flow Archive	12 and 10 stations on the main stem of the River Thames and the River Severn, respectively.	2000-2014
Water quality	UK CEH – Thames Initiative platform	6 stations on the mainstream of the River Thames	2009-2014
Water quality	Environment Agency (EA) – WIMS database	7 stations on the mainstream of the River Thames, 8 stations on the mainstream of the River Severn	2000-2014
Abstraction and Discharge	Environment Agency (EA)	River Severn and River Thames- All catchments	2010-2014

4.2.3 The Integrated catchment model (INCA)

In this study, the INCA water quality model, in its specific formulations for carbon (Futter et al., 2007), for nitrogen (Wade et al., 2002a, Whitehead et al., 1998a, Whitehead et al., 1998b), for phosphorus and for suspended sediment (Wade et al., 2002b), has been employed to simulate the water quality of the River Thames and River Severn. The models INCA-Carbon (INCA-C) version 1.1, INCA-Nitrogen (INCA-N) version 1.0.15, and INCA-Phosphorus (INCA-P) version 1.4.10 were used for the study.

The INCA models simulate daily flow, water quality concentration and flux in streams. INCA reproduces the hydrological cycle with its own hydrological model that provides information on water flow through the soil zone, the groundwater zone, and the river system. The hydrological module represents rainfall-runoff transformations and river routing by solving simple mass-balance first-order differential equations. Fluxes and

concentrations are simulated by solving mass balance equations for terrestrial processes whilst simultaneously solving flow equations. The INCA model does not include an evapotranspiration module, and therefore it requires daily time series of hydrologically effective rainfall (HER) and soil moisture deficit (SMD), with precipitation and temperature. The daily time series of HER and SMD were derived using a semi-distributed rainfall-runoff model, named PERSiST (Precipitation, Evapotranspiration and Runoff Simulator for Solute Transport model). PERSiST simulates water fluxes in a catchment scale, with a series of user-defined buckets indicating different hydrological processes such as snowmelt, surface runoff generation, soil water storage, groundwater and streamflow. It is particularly designed to provide input time series for the INCA models, and its structure and application to the River Thames catchment are described in Futter et al. (2014). In addition, spatial data describing catchment land cover and soil properties, the location of point sources, abstraction and effluent discharge are required to run INCA models. The outputs of simulations are daily time series of flow, water quality concentrations at each reach boundary, as well as profiles along with descriptive statistics of these variables at selected sites. Three INCA models estimate the following water quality variables in the stream: 1) INCA-C: Dissolved organic carbon (DOC) and dissolved inorganic carbon (DIC), 2) INCA-N: Nitrate (NO_3) and Ammonia (NH4), and 3) INCA-P: total phosphorus (TP) and soluble reactive phosphorus (SRP), suspended sediment (SS). The INCA models are calibrated using the time series of observed and modelled daily discharge and point estimates of in-stream water quality concentrations to minimise the differences between observed and modelled values.

4.2.3.1 The Thames system

As the River Thames is one of the most heavily studied catchments in the world, several previous INCA applications in the Thames can be found in the literature following. The main source of water pollution in the catchment are fertilisers used in agriculture, causing high levels of nitrate in the water (Howden et al., 2010) and contributing to the high levels of phosphorus (Johnes, 1996), and sewage effluent, which provides the most significant risk for river eutrophication (Jarvie et al., 2006). The INCA model has been capable of simulating those issues in the Thames and projected future changes in nitrogen concentrations (Jin et al., 2012), phosphorus concentrations (Crossman et al., 2013), and suspended sediment concentrations (Bussi et al., 2016a). Other water quality issues in the River Thames include faecal coliform contamination (Whitehead et al., 2016), organic contamination (Lu et al., 2016), and microplastics (Nizzetto et al., 2016; Rowley et al., 2020). Not only have the INCA models explored the Thames water quality concentrations in various temporal scales, but also the uncertainty and sensitivity of model parameters have been widely discussed to get a better understanding of INCA model performance in the Thames and the UK catchments (Bussi et al., 2016a, Jackson-Blake and Starrfelt, 2015).

In order to model the Thames, the river system has been divided into 22 reaches and sub-catchments from the source at Cricklade to the lowest weir on the freshwater downstream boundary at Teddington (Figure 4.1). Reach boundaries were drawn at confluences, gauging stations, water quality monitoring stations and the abstraction points for water transfer options. The sub-catchment boundaries were derived using a Digital Terrain Model (DTM). The Ordnance Survey Terrain 50 was obtained to define each sub-

catchment boundary and calculate their areas, length of reach, and slope. The Land Cover Map 2007 was used to classify the land uses in the catchment (Smith et al., 2007). The INCA model allows for six different land-use types. Land use types used in the INCA models are re-classified from the 23 types in the Land Cover Map 2007 to 6 land uses, namely, forest, arable, improved grassland, unimproved grassland, wetland and urban. The catchment land use is dominated by agriculture with a mosaic of fertilised grassland (improved grassland) and arable land, especially in the upper reach area (about 80% of catchments in reach 1 to 5 in Table 4.2). A relatively large part of the lower reaches is occupied by urban area (up to 30% in reach 18). Reach and sub-catchment information, including the reach length, sub-catchment area and the six land use percentages are shown in Table 4.2.

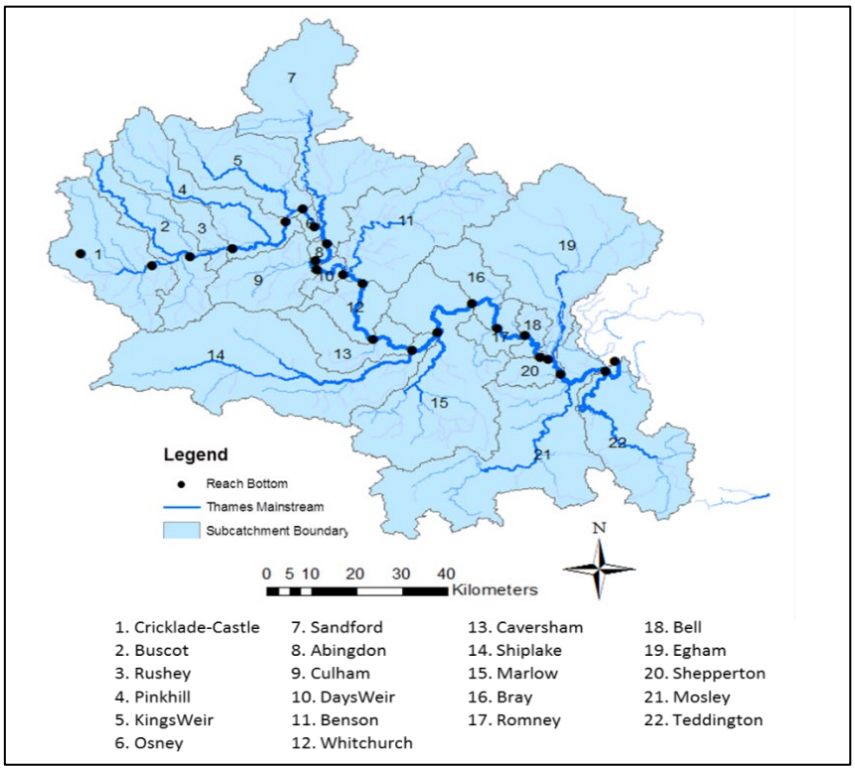


Figure 4.1: Thames sub-catchments (numbers indicate the INCA sub-catchment IDs)

4.2.3.2 The Severn system

Although there are few previous applications to the sub-catchments of the Severn basin (Plynlimon: Lee et al., 2020) or nearby river systems (River Wye: Bussi et al., 2018; River Trent: Bussi and Whitehead, 2020), to date there has not been an integrated water quality study of the whole River Severn System.

The Severn river system has been divided into 20 reaches and sub-catchments, from the source at Plynlimon to the lowest boundary of freshwater downstream at the Severn tidal way (Figure 4.2). The Ordnance Survey Terrain 50 and The Land Cover Map 2007 were used to draw catchment boundaries and geometry by following the same method used for the Thames. The modelling setup includes all tributaries in the whole Severn area. Two major tributaries, the River Teme (Sub-catchment 16) and River Avon (Sub-catchment 17), are included in the INCA simulation and join the main river at above reach 18, as shown in Figure 4.2. The six land use types used for the Severn Catchment are the same as the Thames model setup. The land use in Severn is dominated by agriculture (arable land) throughout the whole catchment, with an average of 65%. Reach and sub-catchment information is shown in Table 4.3. Two different spatially distributed input time series were applied for the Severn catchment due to the large spatial variations in precipitation between the uplands (reach 1-9) and the lowlands (reach 10-20). As described in the previous chapter (**3.1** Study catchments), the mean annual rainfall difference between uplands and lowlands is approximately 367mm. Thus, it was crucial to apply spatially distributed climate input to the simulations. Two major tributaries (River Avon and River Teme) have contributed to high nutrients concentration in the lower Severn (Bowes et al., 2005) and defined as a eutrophic concerned area (DEFRA, 2002b). It is worth noting that

Severn's water supply system is complicated and managed together with the River Trent. Hence the problems that had been looked at by water companies and/or regulators might be different from the purpose of this study.

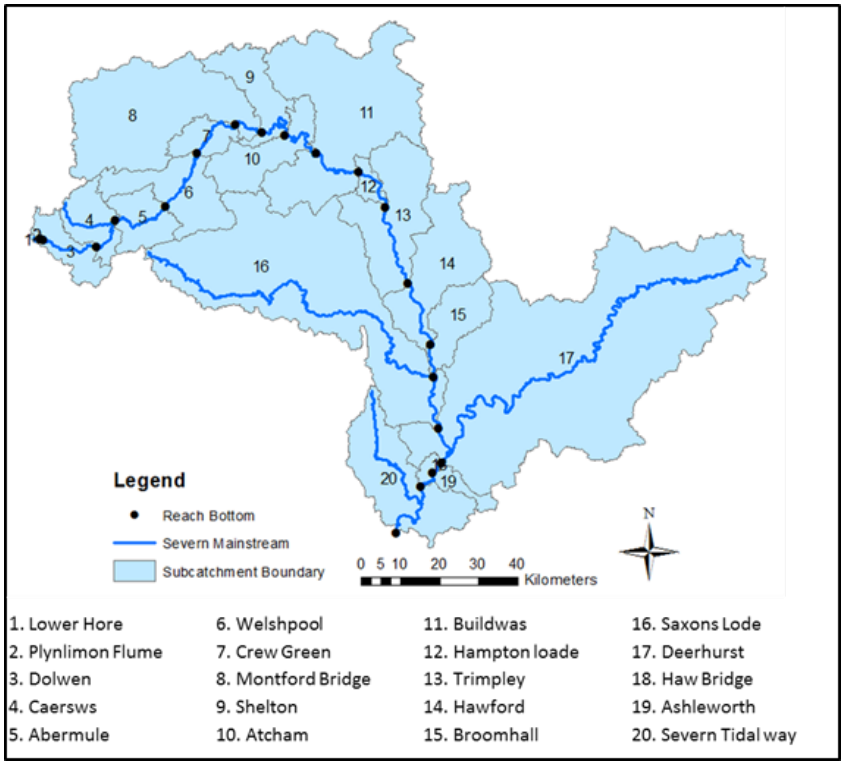


Figure 4.2: Severn sub-catchments (numbers indicate the INCA sub-catchment IDs)

Table 4.2: INCA reach and sub-catchments information on Thames Catchment

			Land use percentage %								Monitoring stations		5-year catchment averaged (2010-2014)	
Reach	**Reach length (m)**	**Sub-catchment area (km^2)**	**Forest**	**Arable**	**Improved grassland**	**Unimproved grassland**	**Wetland**	**Urban**	**Baseflow index**	**Location**	**Flow**	**WQ**	**Discharge (m^3/s)**	**Abstraction (m^3/s)**
1	8,000	554.8	7	44	32	8	2	7	0.7	Cricklade-Castle	Y	Y	0.0215	
2	12,800	439.0	6	54	24	9	1	6	0.71	Buscot	Y		0.0125	
3	12,600	103.8	4	58	26	9	0	3		Rushey		Y		
4	20,700	512.2	7	51	27	11	1	3	0.67	Pinkhill	Y	Y	0.0216	0.4958
5	6,750	474.6	10	54	26	7	0	3	0.67	KingsWeir	Y*		0.0441	0.4958
6	5,670	30.7	9	21	33	12	2	23		Osney				0.4958
7	13,000	937.9	4	49	33	7	0	7		Sandford			0.6452	
8	1,820	29.5	11	49	15	5	3	17		Abingdon				
9	4,140	307.6	5	57	26	7	0	5	0.65	Culham	Y		0.1780	
10	9,320	55.1	5	55	16	5	2	17	0.64	DaysWeir	Y*	Y	0.1538	
11	6,380	721.4	6	46	36	7	0	5		Benson			0.0075	
12	18,010	186.5	9	55	22	7	1	6		Whitchurch		Y	0.0842	
13	10,760	246.9	18	45	24	5	0	8	0.66	Caversham	Y		0.0188	
14	8,910	1,183.1	12	52	23	7	1	5		Shiplake				
15	18,990	974.3	22	29	25	6	1	17		Marlow			0.3843	
16	13,660	196.2	17	28	27	7	1	20	0.7	Bray	Y*		0.5371	
17	8,850	115.6	15	27	31	5	2	20	0.72	Romney	Y		0.5964	0.2464
18	9,540	80.3	18	13	32	3	4	30		Bell			0.0973	8.0053
19	2,000	1,000.5	14	30	30	4	1	21	0.67	Egham	Y	Y		2.1516
20	8,460	93.1	34	7	25	6	4	24		Shepperton			0.3015	6.5802
21	9,540	1,097.4	27	20	28	8	1	16	0.64	Mosley	Y		1.1098	5.5227
22	7,740	586.6	21	15	36	2	1	25	0.63	Teddington	Y	Y		

* now closed gauging station

Table 4.3: INCA reach and sub-catchment information on Severn Catchment

			Land use percentage %									Monitoring stations		5-year catchment averaged (2010-2014)	
Reach	**Reach length (m)**	**Sub-catchment area (km^2)**	**Forest**	**Arable**	**Improved grassland**	**Unimproved grassland**	**Wetland**	**Urban**	**Spatial classification**	**Baseflow index**	**Location**	**Flow**	**WQ**	**Discharge (m^3/s)**	**Abstraction (m^3/s)**
1	2,627	3.4	70	0	0	24	6	0	Upland	0.32	Lower Hore	Y	Y		
2	765	5.4	62	0	0	11	27	0	Upland	0.34	Plynlimon Flume	Y			
3	17,574	165.0	18	5	38	35	3	1	Upland	0.44	Dolwen	Y		0.0386	
4	10,846	198.2	11	6	50	31	2	0	Upland		Caersws			0.0159	
5	16,576	200.4	7	11	65	14	0	2	Upland	0.44	Abermule	Y		0.0893	
6	17,161	360.1	9	17	62	12	0	1	Upland		Welshpool			0.0793	
7	17,151	129.0	8	20	67	5	0	1	Upland		Crew Green			0.0371	
8	9,979	957.4	12	12	45	24	6	1	Upland	0.48	Montford Bridge	Y	Y	0.0803	
9	18,943	252.7	4	44	46	1	1	3	Upland		Shelton		Y	0.0316	0.2494
10	17,809	293.7	7	36	45	4	0	8	Lowland		Atcham		Y	0.3058	
11	22,188	1,157.2	7	51	34	3	1	4	Lowland	0.56	Buildwas	Y		0.0141	
12	14,276	59.1	26	35	26	2	1	11	Lowland		Hampton loade			0.2035	
13	24,781	545.2	14	54	25	2	1	5	Lowland	0.53	Trimpley	Y		0.0138	2.3686
14	20,154	517.7	11	41	22	1	1	25	Lowland		Hawford			0.2880	
15	9,436	230.2	5	44	32	1	1	17	Lowland		Broomhall			0.4763	
16	14,974	1,783.0	11	35	43	10	0	2	Lowland	0.57	Saxons lode	Y	Y	0.2127	
17	11,170	3,009.1	5	50	30	7	0	8	Lowland	0.57	Deerhurst	Y	Y	0.0659	2.2608
18	4,057	2.0	3	48	25	8	15	2	Lowland	0.57	Haw Bridge	Y	Y		
19	6,046	96.4	4	45	26	12	1	12	Lowland		Ashleworth		Y	0.0030	
20	18,943	546.9	10	49	22	9	1	10	Lowland		Severn Tidal way			0.6157	

4.2.4 Model calibration and parameterisation

As discussed above, three different INCA models (INCA- Carbon, -Nitrogen and -Phosphorus) have been applied to the Thames and Severn catchments, and the model evaluation was conducted from 2000 to 2014. The period was split in two: 01/01/2010 - 31/12/2014 for model calibration and 01/01/2000 -31/12/2009 for validation.

The model calibration was conducted in the following two steps: firstly a manual calibration and then a Monte Carlo simulation. Manual calibrations were carefully conducted so that land phase hydrology parameters and water quality process parameters were allowed to vary so as to generate a correspondence between modelled and observed water quality values. The initial values of parameters and their ranges were obtained from previous model applications in the UK catchments (Bussi et al., 2016a; Futter et al., 2007; Jackson-Blake and Starrfelt, 2015; Jin et al., 2012; Wade et al., 2002a). Ledesma et al. (2012) highlighted the importance of manual calibration for INCA models before proceeding with Monte Carlo simulations, so that sensible parameter estimates could be used as the starting points for the Monte Carlo analysis. The manual calibration approach has been proved to be a robust method for obtaining acceptable simulations results for INCA-C (Cremona et al., 2017). Thus, manual calibrations for INCA-C, N, and P were repeated until the visual seasonal patterns were satisfactory and the Nash-Sutcliffe (NS) (Nash and Sutcliffe, 1970) statistics showed a reasonable agreement for water quality variables.

After an acceptable manual calibration had been achieved, the Monte Carlo simulation analysis was then applied. The parameter sets derived from manual calibrations were used as a starting point for the Monte Carlo simulation. Hydrological parameters were fixed,

and only water quality processing parameters were allowed to vary by ± 20%. The random values of the most influential parameters were generated to create 10,000 different model parameter sets. Then the INCA model was run with all the parameter sets, and the results were analysed in terms of NS statistics to find the best performing parameter sets. The performance of INCA model results was assessed based on observed flow and water quality at two stations for the Thames (reach 4 and reach 19) and for the Severn (reach 9 and reach 18) respectively. The results section below gives the details of the model fit statistics.

The following parameters were identified as the most sensitive from the previous general sensitivity analysis of INCA models (Spear and Hornberger, 1980, Whitehead et al., 2015b) and previous INCA model applications (details below) and taken to the Monte-Carlo simulation:

i. Carbon (Futter et al., 2007; Lee et al., 2020): DOC mineralisation rate, the rate of change in mass of DOC to SOC, the rate of change in mass of SOC to DOC, retention volume, residence time, minimum soil flow, soil temperature, maximum SMD (soil moisture deficit) at which carbon processing can occur, and enzymatic latch rate.
ii. Nitrogen (Jin et al., 2012; Wade et al., 2002a): soil denitrification coefficient, nitrification, mineralisation and immobilisation rates in the soil, nitrogen uptake rate by crops, groundwater nitrate concentration, instream nitrification rate and instream denitrification rate.
iii. Phosphorus and sediment (Bussi et al., 2016a, Jackson-Blake and Starrfelt, 2015; Lazar et al., 2010): sorption coefficients of the soil matrix, water column, and

stream bed (which adjusts the sorption capacity in each compartment), splash and flow erosion parameters (describing the erodibility of soils), flow erosion direct runoff threshold (describing the threshold above which flow erosion occurs), transport capacity scaling factor and non-linear coefficient (which adjusts the transport capacity on the hillslopes), instream sediment transport parameters (which adjust the transport capacity in the channel).

4.2.5 Climate Scenarios

As discussed in the previous chapters, climate models are useful tools to allow extending the range of current climatological variability beyond the observational records by generating a large ensemble of weather sequences. The weather@home2 (w@h2) synthetic weather sequences (Massey et al., 2015; Guillod et al., 2018) have been used to study the impact of climate change on water quality, especially during droughts. The data set is specifically suitable for the study of extreme events (drought and heavy precipitation), which requires the spatial consistency of hydro-meteorological datasets (Guillod et al., 2018). Of particular relevance for water quality modelling is the change in precipitation, which can affect retention times and river flows and temperature, and these, in turn, influence the rates of the chemical reactions occurring in the soils and waters. From a 25 x 25 km grid across Europe, w@h2 weather sequences are available for baseline (1975-2004), near future (2020-2049), and far future (2070-2099). Each time frame provides a 100-time series, and all are generated using the same climate model. Outputs from grid cells include the Thames (Figure 4.3) and the Severn (Figure 4.4) were averaged to generate INCA models' input time series. For the Severn catchment, two

different spatially distributed input time series were generated for climate projections (Upper Severn: Figure 4.4 (a) and (b), Lower Severn: Figure 4.4 (c) and (d)).

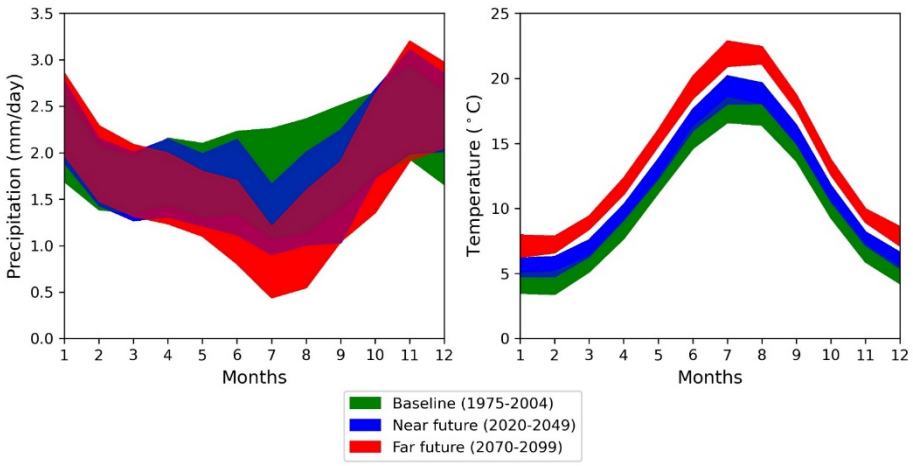


Figure 4.3: Precipitation and temperature projections (w@h2) for the Thames catchment

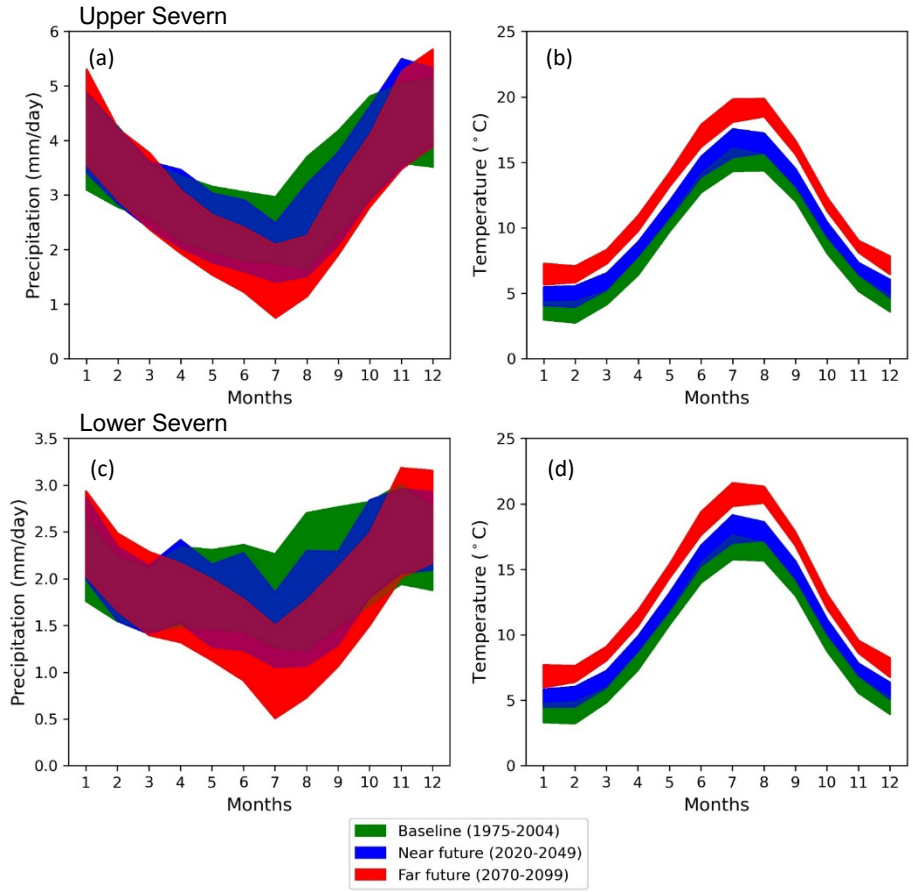


Figure 4.4: Precipitation and temperature projections (w@h2) for the Upper (a,b) and Lower (c,d) Severn catchment

4.3 Results and Discussions

4.3.1 Model evaluation - Hydrology

The INCA hydrological model was employed to reproduce the river flows in the Thames and Severn rivers. The INCA hydrological model was calibrated using observed meteorological data as input and observed values of flow from the gauging stations in the study area as the output references. In order to successfully model water quality using the INCA water quality module, it is also necessary to ensure that the flow is correctly modelled. The calibration and validation results are shown in Figure 4.5, where the grey shaded area indicates the period of calibration (2010–2014). The calibration period was chosen from the relatively recent past to reflect the availability of input data and the input data characteristics such as land-use, catchment conditions, abstraction activities and wastewater treatment standards within the catchment. The validation period was 2000–2009. Simulation results are shown at two locations for both catchments. For the River Thames, reach 4 (upper Thames, Farmoor) and reach 19 (lower Thames, Runnymede) were chosen. Reach 8 (middle Severn, Montford) and reach 18 (lower Severn, Haw Bridge) were chosen to be representative locations on the River Severn.

One of the calibration strategies is to represent the recent past drought 2010-2012. The development of the drought was a result of persistent blocked weather patterns in 2010. It intensified during 2011 due to an extreme exaggeration in the normal northwest to southeast rainfall distribution until the early spring 2012 when a dramatic termination happened with the wettest April to July over England and Wales in almost 250 years (Kendon et al., 2013). Thus, the 2010-12 drought ranked among the ten most significant one- to two- year droughts in England in the last 100 years. With the purpose of the study,

it is crucial to represent the dynamic changes in hydrology with model simulation for the future scenario projections. As shown in Figure 4.5, the simulation results can be considered satisfactory in terms of reproducing seasonal patterns and low and high flow periods. The performance indices for both calibration and validation are shown in Table 4.4. The metrics used for the model performance assessment were the Nash and Sutcliffe Efficiency (NSE) (Nash and Sutcliffe, 1970) and correlation coefficient R^2. Perfect matching can be represented by the value of one. The NSE values in the River Thames were 0.73 and 0.82 in the calibration period, and 0.81 and 0.78 for the River Severn. In general, model simulations can be judged as satisfactory if NSE > 0.5 or R^2 > 0.6 for daily flow simulations of watershed-scale models (Moriasi et al., 2015). Although the performance index shows the satisfactory range for the two catchments, an overestimate of peak flows is observed from the model simulations, especially during the validation period. However, the model was able to reproduce the low flows for 2010-2012 droughts and the post-drought peak.

Table 4.4: Performance indices of INCA hydrological model (calibration and validation). R^2: correlation coefficient, NSE: Nash and Sutcliffe Efficiency

Catchment	Reach	Calibration		Validation	
		R^2	NSE	R^2	NSE
Thames	4	0.76	0.73	0.68	0.63
	19	0.82	0.81	0.76	0.74
Severn	8	0.85	0.81	0.89	0.87
	18	0.80	0.78	0.84	0.73

The results presented above is from the INCA-C hydrological model simulations. However, the INCA hydrology module is run to proceed with water quality simulations for each of INCA models, the performance of hydrology for all INCA models (INCA-C,

N, and P) is assessed by NSE to ensure the appropriate water quality simulations. The general flow regime of the River Thames and River Severn was simulated with good matches between modelled and observed flow data (NRFA) with NSE between 0.60 to 0.87 (shown in Table 4.5).

Table 4.5: Performance index (NSE) of all INCA hydrology models

Catchment	Reach	INCA C		INCA N		INCA P	
		Calibration	Validation	Calibration	Validation	Calibration	Validation
Thames	4	0.73	0.63	0.74	0.64	0.78	0.70
	19	0.81	0.74	0.82	0.75	0.81	0.73
Severn	8	0.81	0.87	0.74	0.78	0.74	0.77
	18	0.78	0.73	0.74	0.67	0.60	0.66

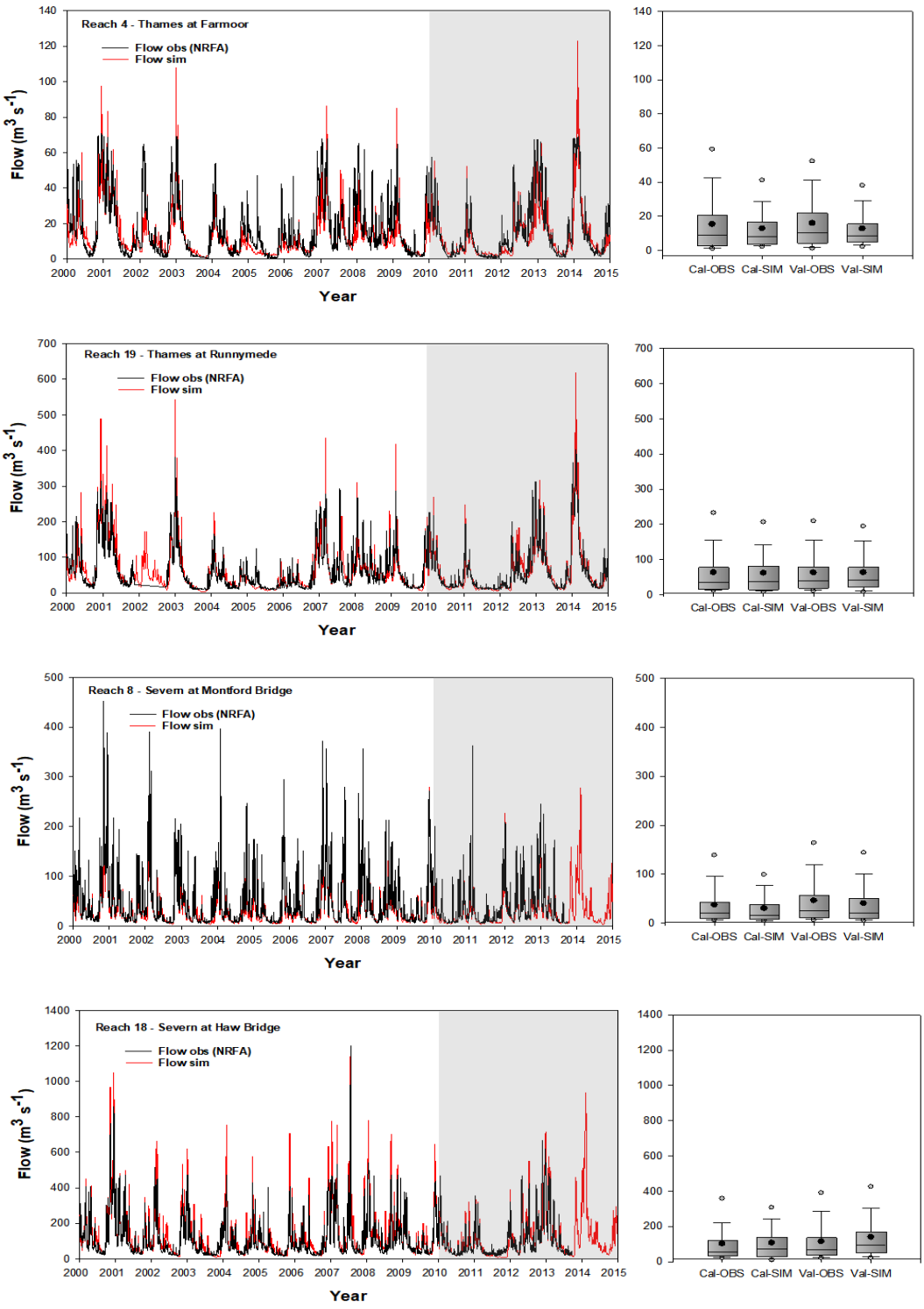


Figure 4.5: INCA hydrological model calibration and validation results at two locations on the River Thames (first row) and the River Severn (second row). Observed data: National River Flow Archive, daily flows, 2000–2015. The shaded area indicates the calibration period 2010–2014.

4.3.2 Water quality evaluation and current water quality status in two catchments

The INCA models were calibrated and validated to reproduce the observed DOC, nitrate, total phosphorus, and suspended sediment concentrations for two river systems. The results are shown in Figure 4.6 for the Thames and Figure 4.7 for the Severn, where the shaded area indicates the calibration period (2010-2014) and the unshaded area is for the validation period (2000-2009). The observed and simulated water quality at two stations in each catchment are shown, at reach 4 and 19 for the Thames and at reach 9 and 18 for the Severn, respectively.

From the visual assessment, the water quality calibration results can be generally satisfactory in terms of reproducing the seasonal pattern and magnitude of peaks, as shown in the shaded area in Figure 4.6 and Figure 4.7. Percent bias (PBIAS) index for DOC, nitrate, total phosphorus, and suspended sediment was measured to judge model performance for all INCA models. PBIAS measures the average tendency of the simulated results to be greater or smaller than observed counterparts (Gupta et al., 1999). The advantage of using the PBIAS index is that (i) it can be used to decide how well the averaged magnitudes for the model behaviour of interest, (ii) it is useful for a long continuous simulation, (iii) it is commonly used in the field which means that there is extensive information on reported values in the literature, and (iv) it can help to identify model bias whether it is underpredicted or overpredicted (Moriasi et al., 2015). The optimal value is 0, and positive values mean the underestimation of modelled values, while negative values indicate the overestimation bias. Table 4.6 shows PBIAS indices for the model calibrations and validations in the River Thames and Severn. Satisfactory simulations for the water quality simulations can be indicated if $|\text{PBIAS}| \leq 30\%$ for

water quality variables (Moriasi et al., 2015), but they vary with the model type and the catchment characteristics. For the calibration period, PBIAS is less than 10% bias of DOC and nitrate for both catchments. Total phosphorus (the Thames) and soluble reactive phosphorus (The Severn) concentrations are shown satisfactory agreement with less than 15% biases. Suspended sediment is acceptable results in lower reaches for both catchments but showing greater bias in the upper reaches (underestimation).

In the validation period, there is an apparent discrepancy between model simulations and observed values for nutrients. The constant fertiliser inputs applied for the calibration period, and the same amount was used for validation. However, nutrient inputs from catchments to river systems have dynamically changed since the early 2000s in England and Wales. Fertiliser application rate has been gradually reduced over the validation period (2000-2010) and remain relatively constant during the calibration period (2010-2014). According to the British survey of fertiliser practice (DEFRA, 2019), the overall nitrogen applications made to grassland have shown a notable decline from 2000 to 2018, whereas nitrogen levels applied to tillage have remained constant. The average nitrogen rates to all crops and grassland have been reduced by 8%, from 109 kg/ha (validation period average) to 100.4 kg/ha (calibration period average). This reduction in nitrogen levels is attributed by the decline in cattle numbers in grassland in conjunction with some improvement in manure use efficiency, which was encouraged by a higher nitrogen fertiliser price (DEFRA, 2019). Overall phosphate application rates rapidly declined between 1999 and 2009, where the rates were 20 kg/ha and 9 kg/ha, respectively, and then have remained relatively constant since 2010, which were at range of 8 – 10 kg/ha. This is contributed by the combination of decreased average application rates on both

grassland and tillage and the reduced land proportion receiving a phosphate dressing. An average 30% decrease in phosphate fertiliser application was found in the calibration period compared to the validation period. In addition to the changes in fertiliser inputs, phosphorus stripping at the major sewage treatment plant under the UWWTD (the Urban Waste Water Treatment Directive) has been applied in the UK widely, which immensely reduce phosphorus effluent water quality since 2000 (Crossman et al., 2013; Wade et al., 2002c). A total investment of £39,126 M was made to sewerage services between 1990 and 2015 across the UK to secure compliance with the UWWTD (DEFRA, 2012). Civan et al. (2018) analysed a long-term trend of total phosphorus concentrations in British rivers and reported significant reductions by 69% since a peak in 1985 (0.33 mg/L in 1985 to 0.10 mg/L in 2012). The step decline to a lower annual average concentration was observed since 1994, which can be ascribed to actions taken at sewage treatment as part of UWWTD together with the impact of declining fertiliser use (Civan et al., 2018, Bowes et al., 2009, DEFRA, 2002a,Environment Agency, 2018).

Undeniably, the stationary nutrient inputs for 15 years are not realistic. Besides, these inputs are highly variable in time as they depend on anthropogenic activities in catchments. However, the study aims to analyse long-term water quality changes under changing climate and droughts rather than a reconstruction of water quality variability in two catchments. For this reason, given the uncertainty in both observation data, model parameterisation and model inputs, the model can be acceptable to proceed to the future projection.

Table 4.6: PBIAS (%) for calibration and validation

Catchment	Reach	2010-2014 Calibration				2000-2009 Validation (post-2008 Validation)			
		DOC	N	P	SS	DOC	N	P	SS
Thames	4	-8	1	-1	38	-16	11	18 (13)	29 (27)
	19	4	-1	-14	7	-22	7	2 (2)	24 (20)
Severn	9	8	-3	-8	35	19	13	35 (16)	36 (32)
	18	-6	0	14	27	2	5	58 (43)	59 (22)

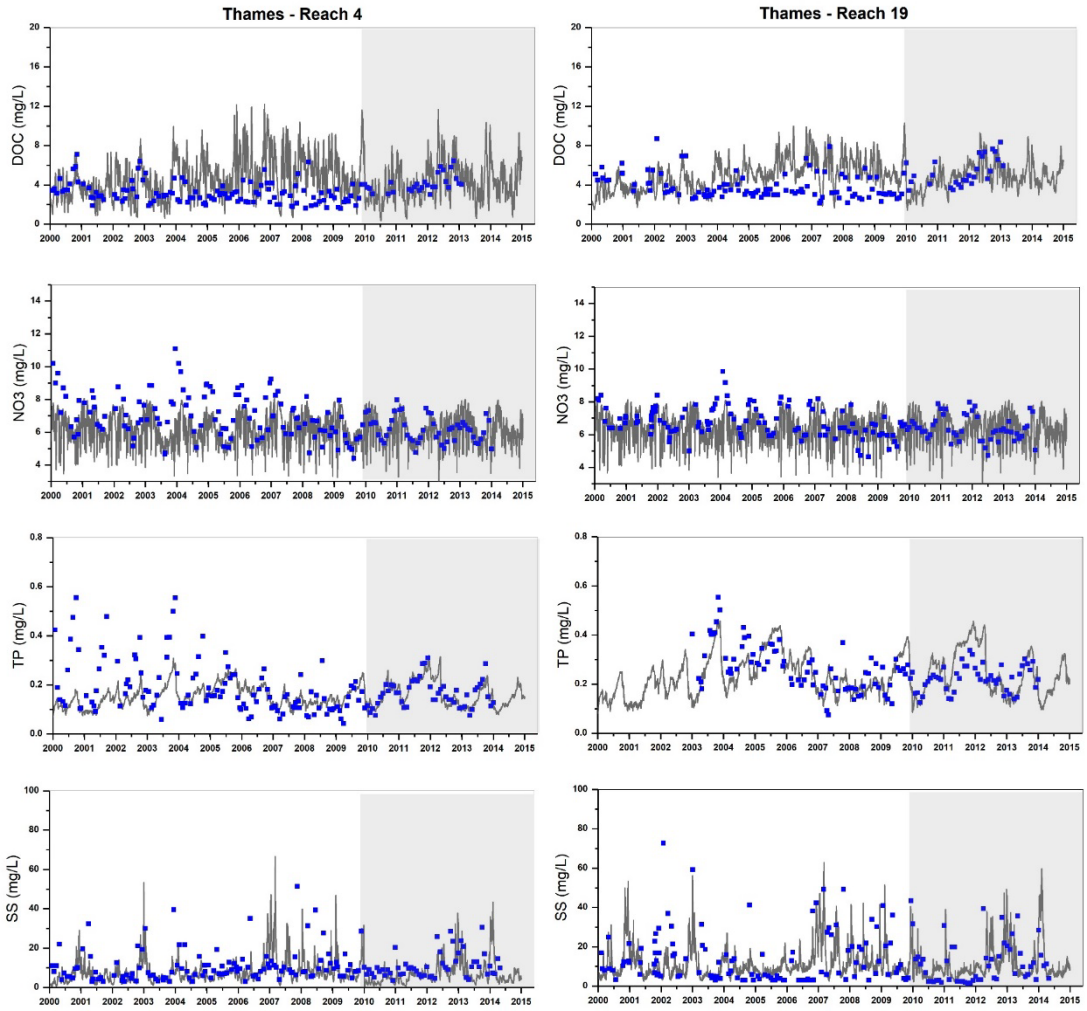


Figure 4.6: Model evaluation results for DOC concentration, nitrate concentration, total phosphorus concentrations, and suspended sediment concentrations for Reach 4 and Reach 19 in the Thames. The dark grey continuous line indicates the model simulations, and blue dots are the observations. The shaded area is for calibration, and the unshaded area is for validation.

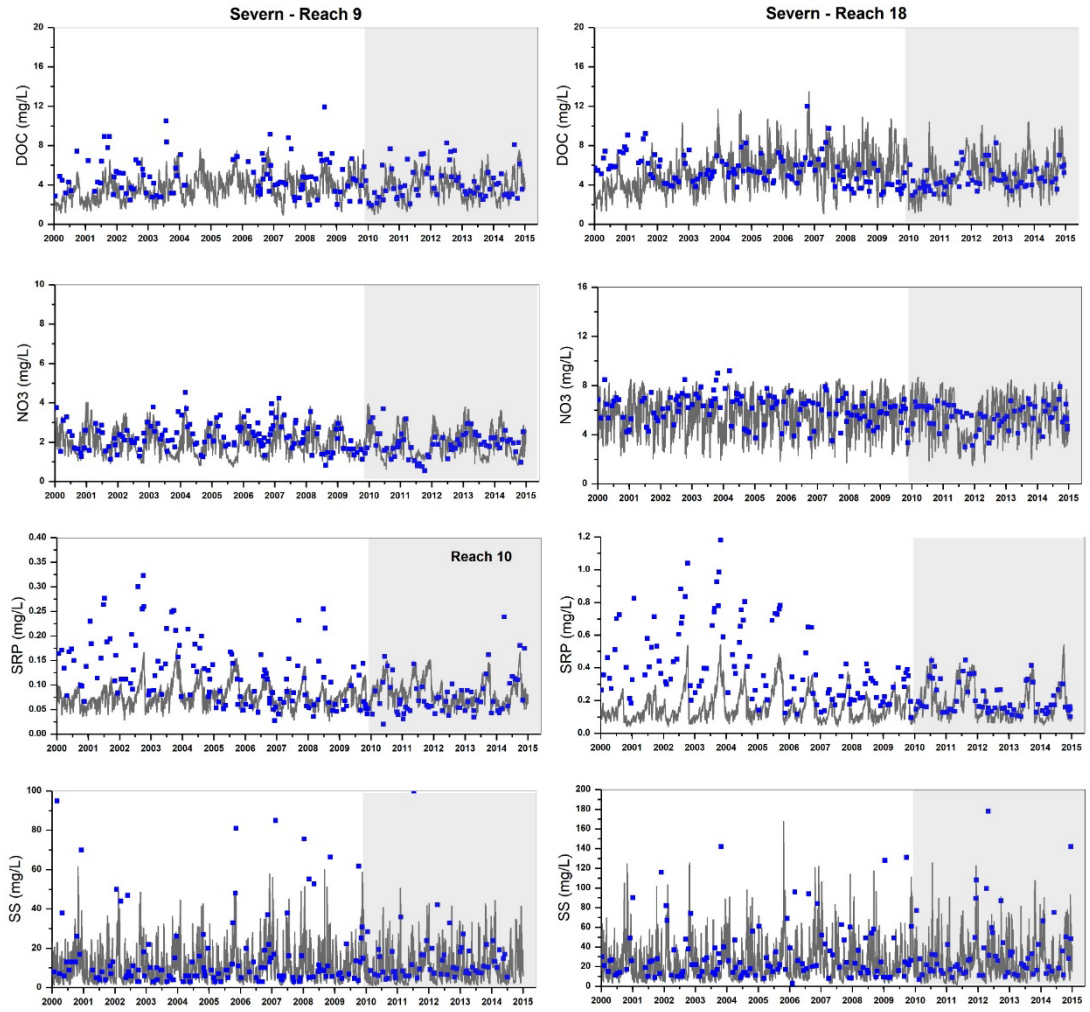


Figure 4.7: Model evaluation results for DOC concentration, nitrate concentration, SRP concentrations and suspended sediment concentrations for Reach 9 (or Reach 10) and Reach 19 in the Severn. The dark grey continuous line indicates the model simulations, and blue dots are the observations. The shaded area is for calibration, and the unshaded area is for validation.

4.3.3 Climate change in the Severn and Thames

Wetter winters and drier summers are predicted by the climate model used in this study (as shown in Figure 4.3 and Figure 4.4). A similar pattern of climate forecast in the UK has been reported (Alexander and Jones, 2000; Watts et al., 2015), and the results of the present study are in line with available scientific evidence. According to the results of w@h2 climate scenarios, precipitation in the Thames and the Severn catchments is

expected to show an increase in winter months and an apparent decrease in summer months (Figure 4.3 and Figure 4.4). The gradual trends have been shown from the baseline to the near and to the far future time frame. In the far future, December mean precipitation is expected to increase by 17% in both the Thames and the lower Severn, and approximately 11% in the upper Severn catchment. While the increase in precipitation is shown in the winter months, almost half of the summer precipitation is expected to decrease in the far future. In July, 47% and 49% of the average decrease in precipitation are expected in the Severn and Thames, respectively in the far future. In general, projected changes in the seasonal patterns of precipitation are more notable than the total annual average precipitation. The Thames' total annual precipitation is expected to decrease by 4% in the near future and 10% in the far future (698 mm in the baseline, 674 mm in the near future and 629 mm in the far future). For the Severn, it shows a similar percentage decrease as the Thames. The average daily air temperature is anticipated to increase for both future time periods for both catchments for all months. The Thames' annual mean temperature changes are from 10.03℃ as the baseline to 11.33℃ for the near future and 13.54℃ for the far future. The lower Severn shows a similar mean temperature range and its change as the Thames whilst the upper Severn is slightly colder, which shows 8.67℃ as the baseline to 9.91℃ for the near future and 12℃ for the far future. More details of the range of w@h2 projections in the Thames and the Severn can be found in Table 4.7.

Table 4.7: Precipitation and temperature (w@h2 climate scenarios) for the Thames and Severn catchments. The numbers inside of the bracket are the range of 100 ensembles for each time frame (BL: baseline (1975-2004), NF: near future (2020-2049), and FF: far future (2070-2099)).

Precipitation (mm/day)									
	Thames			**Upper Severn**			**Lower Severn**		
Month	**BL**	**NF**	**FF**	**BL**	**NF**	**FF**	**BL**	**NF**	**FF**
1	2.14 (1.69-2.64)	2.27 (1.89-2.77)	2.39 (1.97-2.85)	4.02 (4.87-3.10)	4.09 (3.41-4.89)	4.16 (3.55-5.31)	2.20 (1.76-2.65)	2.33 (1.99-2.88)	2.45 (2.02-2.94)
2	1.69 (1.39-2.08)	1.78 (1.44-2.15)	1.88 (1.48-2.29)	3.46 (2.78-4.25)	3.46 (2.83-4.24)	3.49 (2.91-4.22)	1.88 (1.55-2.23)	1.97 (1.55-2.34)	2.07 (1.66-2.49)
3	1.64 (1.35-1.96)	1.64 (1.27-2.00)	1.70 (1.32-2.09)	3.01 (2.57-3.58)	2.96 (2.39-3.61)	3.00 (2.38-3.77)	1.75 (1.45-2.13)	1.75 (1.41-2.14)	1.81 (1.40-2.29)
4	1.77 (1.44-2.15)	1.72 (1.31-2.15)	1.64 (1.24-2.00)	2.73 (2.15-3.35)	2.67 (2.03-3.46)	2.50 (1.93-3.09)	1.89 (1.49-2.34)	1.86 (1.52-2.42)	1.77 (1.32-2.17)
5	1.66 (1.31-2.10)	1.64 (1.21-1.99)	1.44 (1.11-1.80)	2.45 (1.95-3.16)	2.38 (1.76-3.04)	2.10 (1.54-2.64)	1.81 (1.46-2.31)	1.76 (1.27-2.16)	1.57 (1.13-2.00)
6	1.76 (1.35-2.23)	1.59 (1.12-2.14)	1.23 (0.81-1.70)	2.44 (1.80-3.06)	2.20 (1.60-2.92)	1.73 (1.23-2.41)	1.88 (1.43-2.36)	1.69 (1.23-2.28)	1.32 (0.91-1.79)
7	1.59 (1.09-2.26)	1.26 (0.91-1.66)	0.81 (0.44-1.20)	2.30 (1.75-2.97)	1.83 (1.41-2.49)	1.24 (0.75-2.10)	1.73 (1.26-2.26)	1.36 (1.06-1.86)	0.90 (0.51-1.50)
8	1.82 (1.14-2.36)	1.47 (1.01-2.01)	0.94 (0.55-1.59)	2.76 (1.64-3.70)	2.24 (1.51-3.21)	1.59 (1.15-2.26)	2.01 (1.23-2.70)	1.62 (1.07-2.30)	1.10 (0.73-1.77)
9	2.03 (1.42-2.51)	1.79 (1.03-2.25)	1.41 (1.05-1.91)	3.24 (2.30-4.19)	2.91 (2.09-3.80)	2.46 (1.91-3.29)	2.12 (1.49-2.77)	1.88 (1.29-2.29)	1.54 (1.07-2.12)
10	2.23 (1.77-2.65)	2.18 (1.73-2.68)	2.04 (1.36-2.63)	3.84 (2.94-4.81)	3.74 (2.92-4.63)	3.42 (2.79-4.12)	2.27 (1.71-2.82)	2.23 (1.80-2.84)	2.07 (1.50-2.49)
11	2.36 (1.94-2.94)	2.43 (1.99-3.11)	2.57 (1.92-3.20)	4.25 (3.59-5.05)	4.34 (3.49-5.50)	4.49 (3.48-5.27)	2.37 (1.94-3.01)	2.46 (2.06-2.96)	2.60 (2.01-3.19)
12	2.24 (1.66-2.65)	2.37 (2.02-2.85)	2.62 (2.06-2.97)	4.40 (3.52-5.15)	4.58 (3.90-5.33)	4.90 (3.89-5.68)	2.38 (1.88-2.77)	2.52 (2.09-2.93)	2.78 (2.17-3.16)
Annual (mm)	698 (648-745)	674 (613-720)	629 (586-685)	1184 (1100-1264)	1138 (1064-1228)	1067 (1008-1170)	739 (681-795)	713 (661-768)	669 (626-731)
Temperature (°C)									
	Thames			**Upper Severn**			**Lower Severn**		
Month	**BL**	**NF**	**FF**	**BL**	**NF**	**FF**	**BL**	**NF**	**FF**
1	4.40 (3.52-5.02)	5.45 (4.78-6.18)	7.14 (6.28-7.95)	3.76 (3.00-4.35)	4.81 (4.07-5.49)	6.48 (5.68-7.28)	4.15 (3.33-4.77)	5.20 (4.49-5.85)	6.88 (5.99-7.71)
2	4.53 (3.43-5.13	5.63 (4.77-6.30)	7.30 (6.63-7.86)	3.74 (2.76-4.39)	4.83 (3.96-5.57)	6.47 (5.85-7.09)	4.27 (3.24-4.87)	5.36 (4.50-6.05)	7.02 (6.41-7.64)
3	5.67 (5.17-6.33)	6.95 (6.31-7.59)	8.89 (8.44-9.42)	4.59 (4.16-5.24)	5.86 (5.26-6.56)	7.75 (7.29-8.34)	5.32 (4.85-6.01)	6.60 (5.98-7.27)	8.52 (8.05-9.11)
4	8.29 (7.73-9.05)	9.62 (9.13-10.30)	11.81 (11.13-12.34)	7.00 (6.42-7.70)	8.30 (7.83-8.95)	10.42 (9.78-10.95)	7.89 (7.33-8.60)	9.20 (8.73-9.90)	11.36 (10.70-11.87)
5	11.75 (11.24-12.34)	13.06 (12.28-13.80)	15.43 (14.84-16.04)	10.23 (9.75-10.76)	11.47 (10.78-12.09)	13.72 (13.19-14.26)	11.26 (10.77-11.82)	12.54 (11.80-13.21)	14.85 (14.34-15.40)
6	15.44 (14.65-16.22)	16.84 (16.01-17.62)	19.38 (18.48-20.15)	13.47 (12.72-14.15)	14.75 (13.89-15.46)	17.06 (16.20-17.90)	14.77 (13.99-15.52)	16.11 (15.26-16.84)	18.53 (17.63-19.39)
7	17.56 (16.64-18.53)	19.13 (18.05-20.19)	21.95 (20.95-22.86)	15.14 (14.33-16.11)	16.53 (15.38-17.56)	19.05 (18.10-19.83)	16.66 (15.77-17.65)	18.14 (17.01-19.15)	20.81 (19.84-21.61)
8	17.26 (16.44-17.98)	18.81 (18.07-19.65)	21.80 (21.14-22.44)	15.04 (14.38-15.60)	16.44 (15.72-17.23)	19.19 (18.51-19.87)	16.40 (15.68-17.02)	17.87 (17.22-18.63)	20.75 (20.08-21.34)
9	14.18 (13.68-14.80)	15.63 (15.01-16.39)	18.12 (17.59-18.70)	12.49 (12.03-13.14)	13.79 (13.13-14.50)	16.07 (15.61-16.65)	13.49 (13.01-14.15)	14.88 (14.25-15.63)	17.28 (16.81-17.85)
10	9.91 (9.32-10.50)	11.08 (10.54-11.75)	13.18 (12.61-13.73)	8.73 (8.06-9.26)	9.82 (9.29-10.43)	11.84 (11.30-12.33)	9.39 (8.77-9.95)	10.53 (10.05-11.20)	12.60 (12.02-13.11)
11	6.48 (5.92-7.21)	7.67 (7.18-8.19)	9.51 (8.96-9.96)	5.65 (5.18-6.35)	6.83 (6.42-7.34)	8.63 (8.13-9.03)	6.11 (5.57-6.80)	7.29 (6.88-7.82)	9.11 (8.63-9.57)
12	4.88 (4.26-5.49)	6.09 (5.40-6.62)	7.96 (7.15-8.59)	4.25 (3.61-4.83)	5.44 (4.65-6.05)	7.25 (6.46-7.84)	4.59 (3.95-5.22)	5.80 (6.37-5.07)	7.64 (6.79-8.24)

4.3.4 Climate change impacts on river flows

Two sets of future river flow driven by w@h2 climate scenarios were compared to the baseline simulations in the Thames and Severn. Figure 4.8 shows the mean monthly flow for the baseline period, the near future, and the far future, obtained by running w@h2 climate change projections through the calibrated INCA hydrological model. As shown in Figure 4.8, the projected summer flows show a significant decrease in both the River Thames and the River Severn and a gradual decrease over the two future projections (near future and far future). Changes in winter flow show a mixed pattern in both rivers. The winter flows in the River Thames and lower Severn project a drier or similar climate. In contrast, the projections indicate a small increase in winter flows in the upper Severn. On average, the w@h2 hydrological simulation results suggest that mean flow will decrease by up to 14% in the near future and by 32% in the far future in the River Thames. For the River Severn results show similar range of changes: 14% in the near future and 34% in the far future.

Figure 4.9 shows flow duration curves with monthly flow totals for the lower Thames (at Runnymede) and Severn (at Haw Bridge). Flow duration curves identify the proportion of time that the flow is below any given stage (threshold) and the shape of the flow duration curve indicates a catchment's characteristic response to rainfall. The curves at the two sites have a similar flat shape, indicating that the stream is not 'flashy', even far downstream. The catchments may have runoff regulators such as groundwater storage or large reservoirs. The shape also reflects the fact that they are the mainstream channels in each river system. The flow duration curves for the near future and far future show a general decrease of flow rate at all frequencies. For the winter months, the near future

projects a minor shift towards lower monthly total flows in both rivers. In the far future, however, the changes in the winter months show a significant decrease from baseline. Changes in summer month flow predominantly show decreases at all frequencies. Flow decreases in the mid-range flows could have implications for water resource management and may trigger the imposition of more restrictions in the future, because this is related to the range of maximum allowed abstraction from the river.

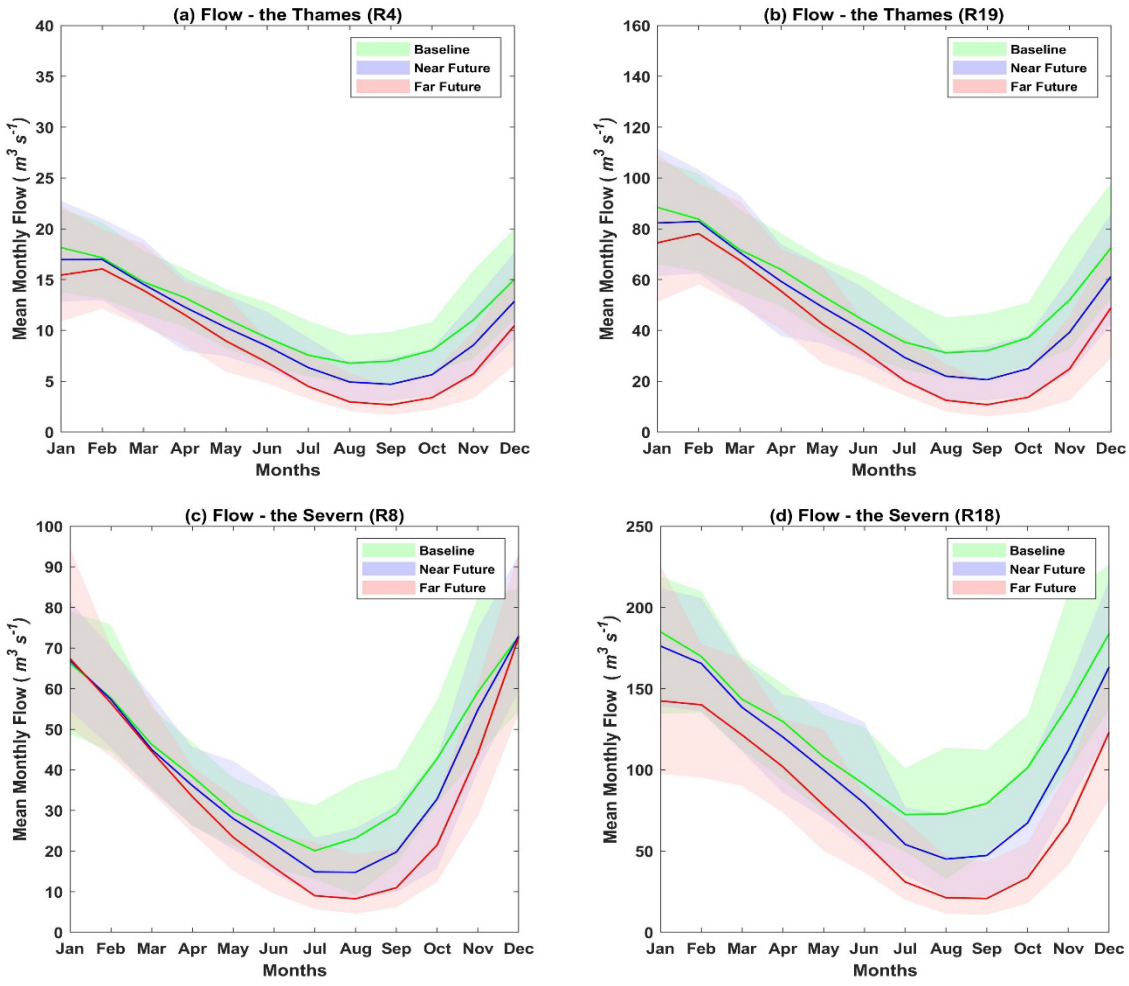


Figure 4.8: Impact of climate change on mean monthly flows for the River Thames (a, b) and the River Severn (c, d)

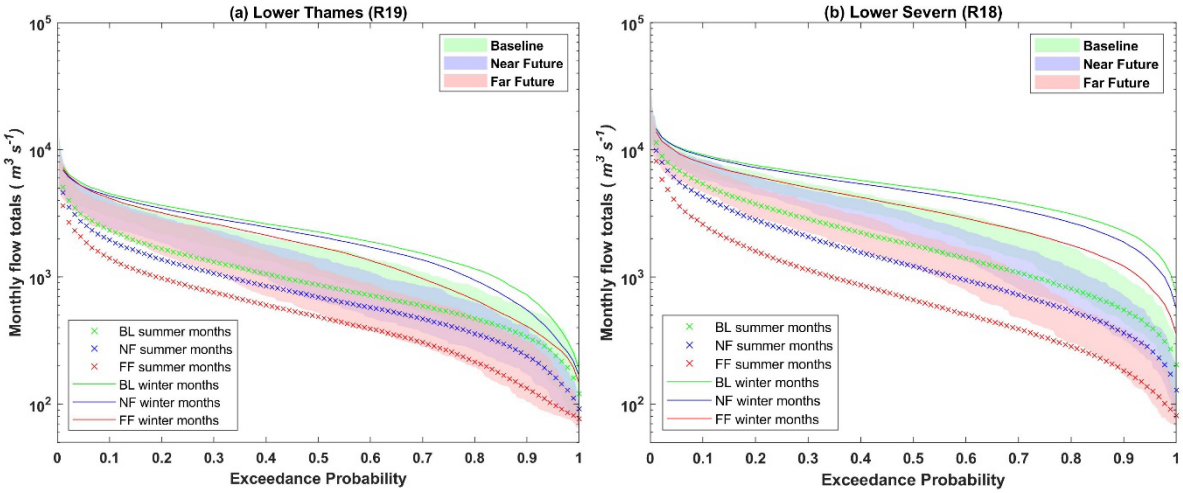


Figure 4.9: Flow duration curves for the River Thames (a) and Severn (b) at lower reaches

4.3.5 Climate change impacts on water quality

Using w@h2 projections for the near future (2020-2049) and the far future (2070-2099) coupled with the INCA models, estimates of the impact of climate change on water quality were assessed and quantified for the Thames and Severn catchments. According to the INCA models results, the concentrations of DOC, nitrate, phosphorus, and suspended sediment are expected to vary under a future changing climate. Figure 4.10 - Figure 4.13 will show the variation of the monthly average concentrations of DOC, nitrate, total phosphorus, and suspended sediment in two sections of the River Thames and Severn. Figures display the expected ranges of each water quality variable's variations with the median line for each time slice (green: baseline, blue: near future and red: far future).

4.3.5.1 Dissolved organic carbon (DOC)

In the Thames (Figure 4.10 (a) and (b)), the general seasonal pattern of DOC concentration across the year is predicted to remain the same, with minimum concentrations in August and maximum concentrations in December and January. The notable changes in summer DOC concentrations (May to September) are observed than those changes in the winter months (November to January). The monthly average DOC concentration in August (at Reach 4) is expected to decrease from 4.31 (3.96-4.79; 5-95% confidence interval) to 3.59 (3.21-3.98) mg/L for the near future and to 2.49 (1.99-2.91) mg/L for the far future, which shows a 42% decrease in the far future from the baseline. Figure 4.10 (c) and (d) shows the variation of the DOC monthly average concentrations for two sections of the River Severn. In the upper Severn, there is a slight decrease in summer months (July-August) but a significant increase in later summer to autumn (August, September, and October) which shows the gradual increase from baseline to near future and to far future. The upper Severn's seasonal pattern is related to the Severn's headwater carbon dynamics (following Chapter **5,** Figure 5.9) where the major source of carbon is peatlands. As it will be discussed in Chapter **5**, the enzymatic latch effect can explain the DOC surge in the later autumn followed by droughts (Lee et al., 2020) that impact on the Upper Severn catchments. The lower Severn showed increased trends for all months for two future frames. The annual average concentration in lower Severn is expected to increase from 6.68 (5.85-7.77) to 7.03 (5.96-8.27) mg/L for the near future and to 7.78 (6.38-9.31) mg/L for the far future. The average 5% and 17% of annual increase is predicted in the near and far future, respectively. The DOC levels in the

Thames are quite high and reflects the inputs from sewage treatment discharges rather than any other sources (Bowes et al., 2018).

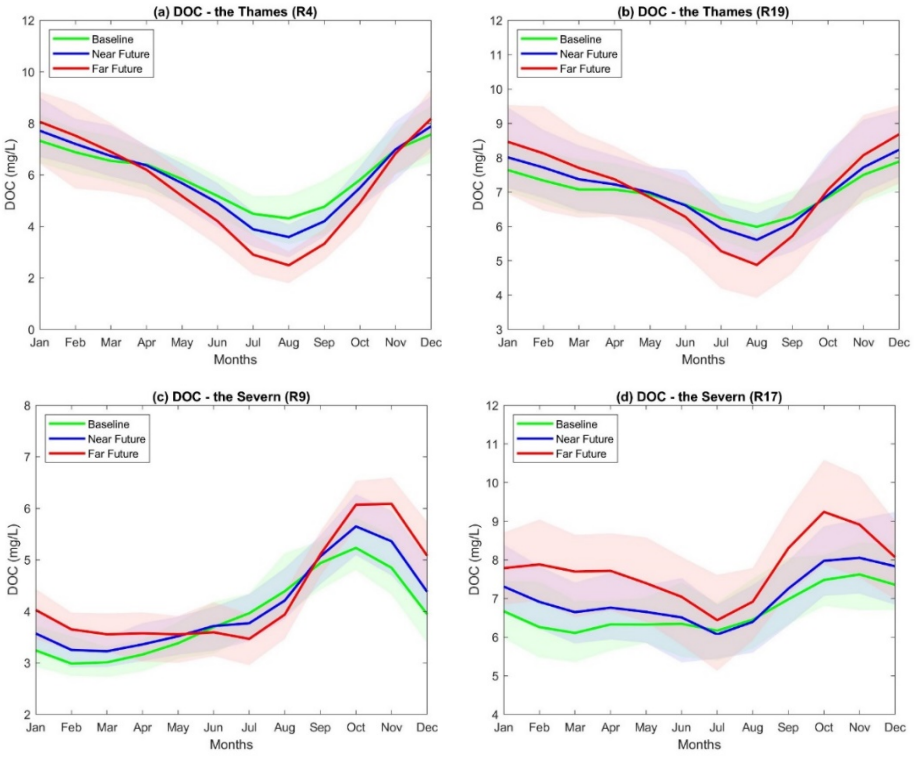


Figure 4.10: Impact of climate change on monthly average DOC concentrations at the Thames (a,b) and Severn (c,d) catchments

4.3.5.2 Nitrogen

Nitrate monthly average concentrations are expected to reduce for all months and two future time series, apart from summer months (July and August) in the lower Severn (Figure 4.11(d)). The Thames and upper Severn (Figure 4.11(a)-(c)) present the consistent seasonal patterns that nitrate concentrations will gradually decrease over summer with the lowest concentrations in August and September, and subsequently increase throughout autumn months. In spring (March to May), nitrate concentration is expected to stay almost the same in the near future and have less than 5% change in the far future

than the baseline. Lower Severn shows different variation from the others. Although nitrate concentration is expected to decrease in two future projections, summer nitrate concentration (July – August) is unlikely to reduce. There is a significant decrease in winter. The monthly average nitrate concentration in November (at Reach 17) is expected to decrease from 4.16 (3.84-4.44; 5-95% confidence interval) to 3.53 (3.19-3.91) mg/L for the near future and to 2.64 (2.35-2.96) mg/L for the far future, which shows a 37% decrease in the far future from the baseline.

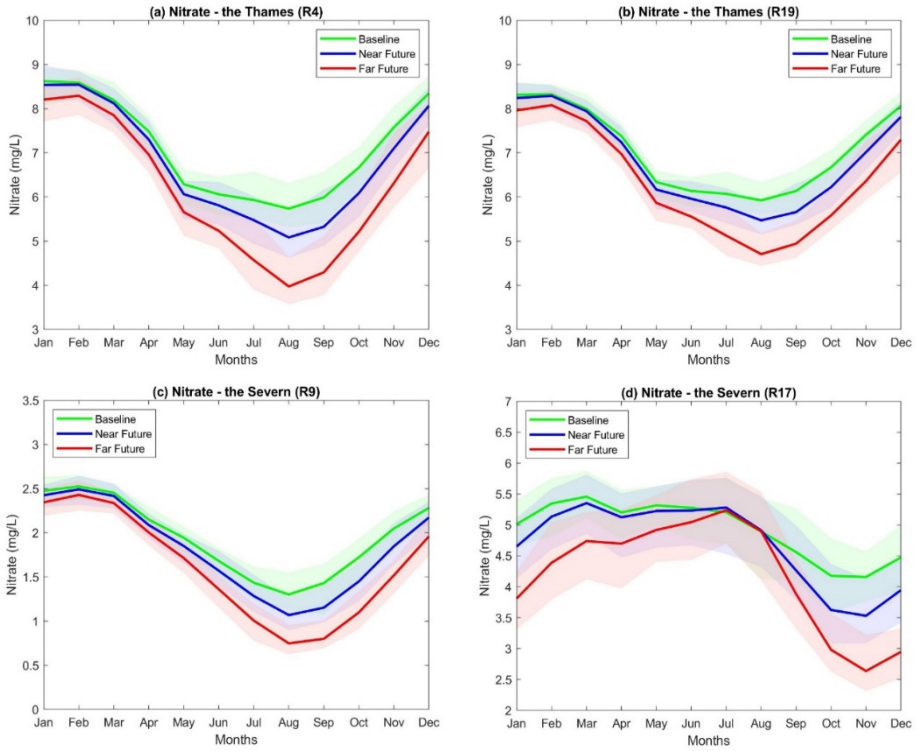


Figure 4.11: Impact of climate change on monthly average nitrate concentrations at the Thames (a,b) and Severn (c,d) catchments

4.3.5.3 Phosphorus and suspended sediment

The INCA- Phosphorus model results show large increases in summer total phosphorus concentrations in the River Thames and Severn for all four locations (Figure 4.12 (a)-(d)).

It appears to have the maximum concentrations in August and September, with a significant surge in the far future. The Severn shows a more significant increase in summer phosphorus peak than the Thames in general. From July to October in the far future, an average 66% increase of monthly phosphorus concentration is projected in the lower Severn compared to the baseline, while a 35% increase is expected in the Thames (Reach 19). Although the annual average concentration is still higher in the Thames, the maximum monthly concentration in the lower Severn becomes larger than the Thames in the far future (0.25 (0.19-0.34) mg/L in the Thames (reach 19) and 0.27 (0.13-0.51) mg/L in the Severn (reach 17)). This change is probably a consequence of a projected lower summer flows and reduced dilution capacity in the two-river system as the main sources of phosphorus are effluents discharged from sewage treatment plants, especially in the Thames (Crossman et al., 2013; Whitehead et al., 2013) and lower Severn (Bowes et al., 2005). Lower Severn has higher average phosphorus concentrations than the Thames.

The future changes of suspended sediment concentrations in the Thames and Severn are quantified using the INCA-P sub-module, and the results are shown in Figure 4.13. It can be noticed that the impact of the w@h2 climate scenarios is to decrease summer suspended sediment concentration and to increase winter sediment concentrations for all locations in both rivers. The Thames catchment shows a relatively large decrease in summer suspended sediment compared to the Severn. The monthly average suspended sediment concentration in August is expected to decrease by 13% in the near future, and by 32% in far future, compared to the Thames baseline (Reach 19). However, the changes in winter in the Severn appear to be larger than those changes in the Thames. In December, an average 22% increase of monthly suspended sediment concentration is projected in the

lower Severn in the far future compared to the baseline, while an 11% is expected to increase in the Thames (Reach 19).

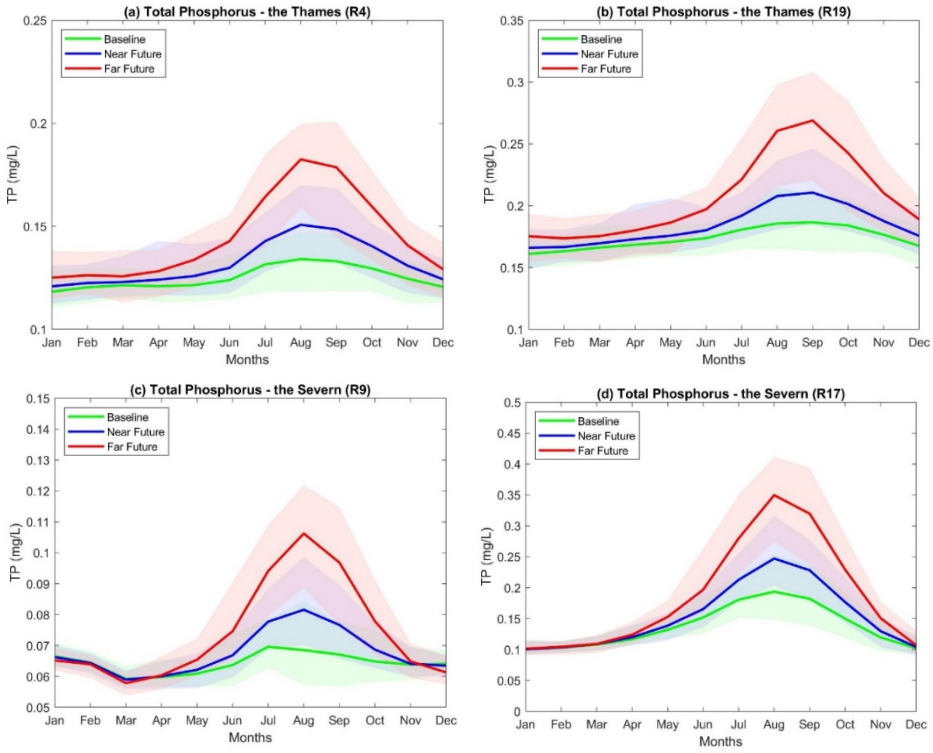


Figure 4.12: Impact of climate change on monthly average TP concentrations at the Thames (a,b) and Severn (c,d) catchments

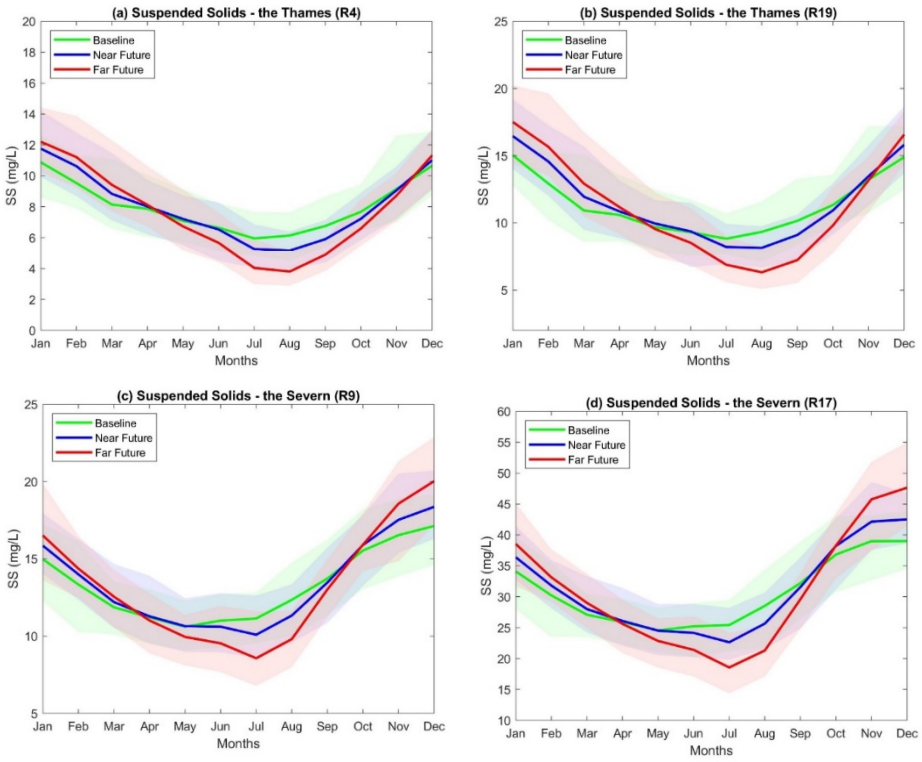


Figure 4.13 Impact of climate change on monthly average SS concentrations at the Thames (a,b) and Severn (c,d) catchments

4.3.5.4 Overall discussions

According to the INCA model results, the concentration of DOC, nitrate, phosphorus, and suspended sediments is projected to vary due to the changing climate in two future time series. It can be noticed that the impact of the scenarios is to reduce the summer flows considerably, thus causing a decrease in summer DOC, nitrate, and suspended sediment and an increase in summer phosphorus in both catchments in general.

A decrease in nitrate and suspended sediment concentration is due to their diffuse-source nature. In the Thames, nitrate is mainly due to diffuse sources such as fertiliser used in agriculture (Bussi et al., 2017; Jin et al., 2012). Thus, the nitrate concentration in the river is proportional to runoff. The decrease in runoff in two catchments are expected due to

the projected decrease in precipitation and an increase in temperature (consequently increase evapotranspiration) in future. Furthermore, low flows mean reduced velocity in the river channel and longer residence times, which will enhance the denitrification process reducing nitrate concentrations (Jin et al., 2012). This affects reduced soil erosion and stream transport capacity, which is related to reduction on suspended sediment concentration.

However, lower Severn shows different system responses to DOC and nitrate concentrations compared to others (the Thames and the upper Severn). Lowland catchments in the Severn are intensively farmed and a high proportion of flow moving along throughflow. Together with the effluent inputs from sewage treatment plants in two major tributaries, the higher rates of runoff possibly wash greater quantities of organic materials from the catchment into the river. The lower Severn is not a groundwater-dominated stream; thus, runoff contributions are naturally high.

Summer droughts cause a sharp increase in phosphorus concentration in the River Thames and Severn due to a reduction of the effluent dilution capacity of the river. The primary source of phosphorus in the Thames and Severn are household effluents discharged by sewage treatment plants (Bowes et al., 2005; Crossman et al., 2013; Whitehead et al., 2013). Therefore, phosphorus concentration can be inversely proportional to flow as the less flow means the less dilution capacity, which contributes to the high phosphorus concentrations in the river. The projected increase in temperature could possibly cause an increase in phosphorus concentration.

Although the monthly average nitrate concentrations are projected to decrease in two catchments, there is still a chance to have the coincidence of increased nitrate and phosphorus concentrations. Increasing nutrient loadings along with warm air and water temperature are to make a favour environment for algal blooms (Paerl and Huisman, 2009; Whitehead et al., 2009b). Among the group of phytoplankton, cyanobacteria is known to produce harmful toxins (Carmichael, 1992) and will cause the unfavoured condition for water supply if the population grows quickly.

The degree of degradation in the lower Severn water quality is a concern for the proposed transfer between two catchments. In this chapter, the changes on water quality are discussed in the monthly average time scale for a 30-year long future conditions. However, the transfers between two catchments will be carried out on a daily scale in the real world. Thus the information of dynamics on water quality on a daily scale should be thoroughly obtained from the modelling study to understand potential risks and trade-offs, and the only modelling study can support the information required.

While the trends found in this study appear important and conclusive, all model simulations and predictions involve some degree of uncertainty. The sources of uncertainty are diverse, including the data, model structure and parameterisation uncertainty. In the modelling application, the input data uncertainty has been evaluated during the model evaluation process. The common observed data uncertainty comes from the errors of sampling, handling, and analytical methods. The policy responses can be the different source of uncertainty. Especially the changes in nutrients inputs were not fully addressed by using a stationary input of fertilisers. The transformation of land use in two catchments were not included as well.

Nonetheless, this study attempted to address the parametric uncertainty by applying a Monte Carlo based calibration technique to find the best parameter sets, based on the pre-defined sensitive parameters. The hydrological and water quality model used in this study, the INCA, is a physically based model and its reliability has been tested in many catchments in the world not only under the data-rich catchments but also under the data-scarce catchments, including the Buriganga River (Bangladesh) (Whitehead et al. (2019a)), the Upper Awash River (Ethiopia) (Bussi et al. (2021b)), the Mekong delta (Whitehead et al. (2019b) and Bussi et al. (2021a)). Some part of the uncertainty is addressed by using the ensemble of climate scenarios. However, it still has a limitation to address all sources of uncertainty in the environmental modelling approach. However, uncertain models can still provide beneficial information for planners and managers and should be used to evaluate the impact of future environmental scenarios, highlighting that the percentage change matters more than the absolute values (Cosby et al., 1985, Bussi et al., 2017)

4.4 Conclusion

This chapter investigated the climate change impacts on river flow, DOC, nitrate, phosphorus, and suspended sediment in two neighbouring catchments, the River Thames and Severn. The modelling framework was developed to support the generation of a large ensemble of future weather scenarios, river flow and water quality time series for two catchments. These times series were employed to assess the impact of droughts and climate change on water quality in-depth.

The Results shows that, as expected, climate change and drought condition cause the changes in river flow and water quality in the Thames and Severn. The impact of the

climate change is to reduce the summer flows considerably, thus causing a decrease in summer DOC, nitrate, and suspended sediment and an increase in summer phosphorus in both catchments. However, lower Severn appears to have a different system response to DOC and nitrate concentrations compared to others (including upper Thames). Summer DOC concentrations in lower Severn is continuously higher than those in upper Thames, which might be the water quality concerns for the proposed transfer during low flows.

5 Impacts of droughts and acidic deposition on long-term surface water dissolved organic carbon concentrations in upland catchments in Wales

5.1 Introduction

The rising trends of dissolved organic carbon (DOC) in upland rivers in Wales has been observed over the past decades and with the proposed water transfer there is a chance that high DOC water might be transferred across to the Thames which might mean extra treatment is required to remove DOC prior to water supply. This would create problems for Thames Water as there are significant costs associated with treatment process to remove carbon in water. Also, water colour is highly correlated with DOC, so as DOC increases water colour will be increasingly brown. This would create water supply problems from a public concerned that drinking waters are going brown. Whilst colour *per se* is not a public health issue, a serious health issue is related to the chlorination of water at treatment plants. The chlorination processes at water treatment plants, when applied to high DOC waters, generate by-products, such as trihalomethanes, which are carcinogens (Chow et al., 2003). This poses a serious health risk and would require alternative water purification processes such as UV radiation or ozone treatments. Both of these are much more expensive options than chlorination. Thus investigating the likelihood of DOC transfer is a key question of concern to Thames Water and the UK Environment Agency. Therefore, this chapter aims to look at the uplands carbon dynamics and forecast the future DOC trends under projected droughts. The transfer considerations and issues are then considered in **Chapter 6**.

While the peatlands cover a relatively small amount of the Earth's terrestrial area, they contain approximately one-third of the world's soil carbon pool (Gorham, 1991; Pastor et al., 2003) and about 40% of the UK's total soil carbon storage (UK National Ecosystem Assessment, 2011). Release of carbon from peatlands includes dissolved organic carbon (DOC) and carbon dioxide (CO_2) in rivers, and direct release of CO_2 and methane (CH_4) to the atmosphere. Rising trends of DOC concentrations in freshwater system have been reported across northern, central Europe and eastern North America, particularly in the industrial areas (Monteith et al., 2007). The changes in the flux of DOC from the land to surface waters have been attracting much interest from academics (Freeman et al., 2001a, Evans, 2015) and water companies due to the problems for water treatment (Ritson et al., 2014). Several mechanisms have been proposed to explain these trends including changes in climate and declines in sulphate deposition across Europe (Evans et al., 2005; Monteith et al., 2007). Fluvial DOC export determines a significant proportion of the peatland carbon balance, and widespread increases in DOC fluxes suggest that this carbon balance may be altering, with major implications for terrestrial carbon budgets. These increases appear to be driven by regional or global-scale environmental changes, although the key driving mechanisms have yet to be conclusively identified.

Major DOC increases have been observed over the timescales that have seen reductions in the acidity of precipitation, therefore, sulphur deposition is thought to have a significant effect by altering microbial populations and suppressing the formation of DOC under high deposition loading (Evans et al., 2006; Evans et al., 2012). Monteith et al. (2007) reported that the increasing DOC concentrations are in proportion to the rates at which atmospheric deposition of anthropogenic sulphur and sea salt are declining. As sulphur

deposition controls soil water acidity, it leads to changes in DOC solubility in the soil (Evans et al., 2012). A recent study by Kang et al. (2018) investigated biogeochemical linkages between pH increase and DOC production in peatlands and demonstrated that increases of pH stimulate key enzyme activities, which involved the decomposition of peat, resulting in the enhancement of DOC concentrations in the pore water. This experiment supports the hypothesis that the recovery from acidification could cause the increase in DOC concentrations in freshwater ecosystems. Clark et al. (2005) discovered that DOC concentrations and sulphate dynamics in soil solution have a strong relationship under drought conditions in peat soils, which are a major source of DOC to surface waters. Climate-related mechanisms present a particular concern as they would imply that increasing atmospheric CO_2 could lead to an increase in peatland DOC release (Freeman et al., 2004). Conversion of this DOC to CO_2, by microbial or photochemical processes, could therefore create a positive feedback effect, raising atmospheric CO_2 concentrations and stimulating the further release of DOC from wetlands. Other proposed drivers include factors related to the increase of air temperature (Freeman et al., 2001b), changes in precipitation and runoff (Tranvik and Jansson, 2002) and changes in land use (Yallop and Clutterbuck, 2009).

Under future climate conditions, drier summers and wetter winters are projected in the UK (Lowe et al., 2018). Changes in climatic conditions will affect catchment hydrology and river water quality (Whitehead et al., 2009a). DOC production rates and mineralisation processes in pore water are sensitive biochemical processes directly affected by these changes (Bell et al., 2018). Drought conditions will consequently affect the carbon storage in peatlands and fluvial DOC concentrations caused by the “enzymatic

latch" process (Freeman et al., 2001b, Fenner and Freeman, 2011). Drought events introduce oxygen into the system and change the redox conditions in peatlands, which controls DOC solubility. Organic carbon is not released during the drought, but subsequent re-wetting and re-flooding accelerate carbon losses from the peat to the atmosphere and the receiving waters.

The long-term monitoring of surface water quality networks in the UK has also revealed increasing DOC concentration trends in upland catchments since the late 1980s (Evans et al., 2005; Worrall et al., 2004b, Worrall et al., 2004a, Sawicka et al., 2017). DOC may be now stabilising in catchments where reduced sulphur deposition has reduced ionic strength in soils. However, this would not be the case for peatlands that continue to leak stored sulphur. One upland catchment, Plynlimon, in the upper Severn catchment, is dominated by organomineral soils, with smaller areas of blanket peat so that the upward movement of DOC concentrations could continue for some years. In addition, extreme weather events could trigger occasional releases of DOC, flushing DOC in late summer and autumn storm events (Delpla et al., 2015). A DOC surge could be expected during storm events which could create problems downstream. Peatlands supply over a quarter of drinking water in the UK (ONS, 2019) and higher DOC concentrations in drinking water can cause human health issues as DOC reacts with the chlorine used in some drinking water treatment processes to produce potentially carcinogenic trihalomethanes (Chow et al., 2003). It affects watercolour, transparency, and pH, which all directly link to stream ecology and increases in DOC represent a change in the quality and ecology of streams, thereby creating difficulties from a Water Framework Directive (WFD) perspective (Whitehead et al., 2006). Additionally, the removal of DOC in raw water in

the treatment plant is costly for water utility companies (Ritson et al., 2014, Ritson et al., 2016). There has been considerable debate about the changes in DOC concentration in the upland UK, and the Acid Water Monitoring Network report rising trends in many upland peat catchments across the UK.

Several DOC simulation models have been developed for both terrestrial and aquatic environments (de Wit et al., 2016). The INtegrated CAtchments model for Carbon (INCA-C) has been widely used to simulate seasonal and long-term patterns in soil and surface water DOC (Futter et al., 2007). Changes in climate, sulphate deposition and land management contribute to variability in surface water DOC concentrations, and INCA-C is capable of simulating complex, interdependent processes in catchment-scale peatland as a multi-parameter process-based model. The initial model development and application was to catchments in Canada (Futter et al., 2007) and further applications including Norway (Futter and de Wit, 2008), Finland (Futter et al., 2009), Ireland (O'Driscoll et al., 2018) and several UK catchments (Xu et al., 2020).

Whether or not rising trends of DOC in upland catchments will continue, or will stabilise, is an important consideration for peatland catchment responses to future drought and acidic deposition, in addition to the implications for downstream drinking water treatment and supply. The aim of this chapter is to present an assessment of potential future climatic and acidic deposition impacts on river water quality in peatland catchments in the upper Severn that has been showing large rising trends of DOC over the last decades. The INCA-C model was first used to simulate current flow and DOC concentrations and fluxes. Here, we introduce new parameter sets of INCA-C to simulate the enzymatic latch effect in peatlands during severe droughts. A large ensemble of climate scenarios

generated by weather@home2 project and acidic deposition scenarios produced by Emissions Monitoring and Evaluation Programme (EMEP) were then projected to drive INCA-C model simulations.

5.2 Study area

The study area, Plynlimon sub-catchments, are headwater catchments of the River Severn in the UK. Two sub-catchments are the Hafren and the Hore at between 300m and 700m above sea level (Figure 5.1). They are 5.5 km^2 and 3.2 km^2 in area, respectively. Peatland, semi-natural grassland, areas of plantation conifer forest (mainly Sitka spruce with some Norway spruce), are all represented within Plynlimon catchments (Neal et al., 2011). The upper parts of the Hafren and the Hore comprise hilltop plateau dominated by up to 2m of deep blanket peat whereas the valley bottoms include areas of seasonally saturated peat and gley soil[2]. The Upper and Lower Hafren represent a single main channel system with a clear divide between the upstream blanket peat and the downstream forested peaty podzols areas. The upper and lower parts of the Hore were harvested in 1985-1988 and 2006-2009, respectively. Logging residue was left on the site after clear-felling, and then the area was subsequently replanted with conifer forest. The catchment soils were ploughed and drained by creating ditches in the between harvest periods, so consequently 'flashy' catchments were created with rapid responses to rainfall and enhanced sediment transport rates.

[2] A wetland soil developed under conditions of poor drainage, resulting in the reduction of iron and other elements, with a typical grey/blue soil colouring (NSRI Staff, 2011).

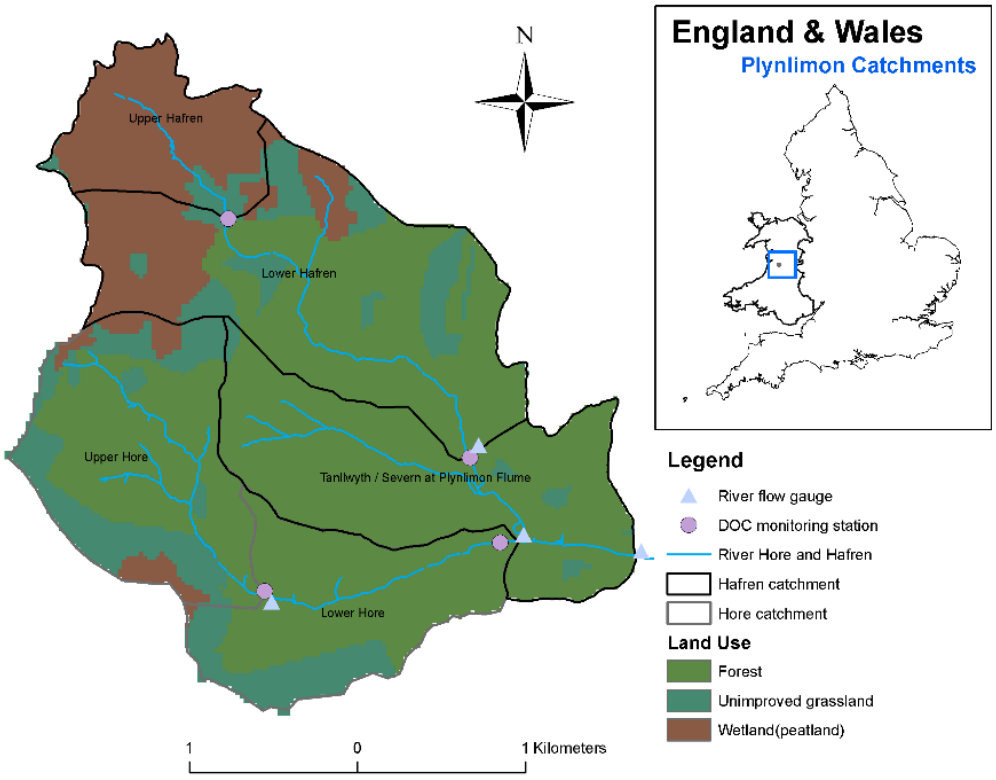


Figure 5.1: Plynlimon catchment with Hore and Hafren sub-catchments

According to the UK National River Flow Archive (NRFA), the Plynlimon receives an average annual rainfall of 2653 mm (computed in the period of 1961-2015, with a minimum of 1855 mm in 1976 and a maximum of 3801 mm in 2000). The annual average temperature is 9.2 ℃ (1995-2015, a minimum of 7.9 ℃ in 1996 and a maximum of 9.8 ℃ in 2014). The average summer temperature is 14.7 ℃, and the average winter temperature is 3.8 ℃. The land uses of the Plynlimon catchments were categorised as forest (both managed and unmanaged), unimproved grassland (i.e., unfertilised grassland), and peatland. The Hore catchment was divided into two reaches, upper Hore and lower Hore. The land use of Hore catchment is predominantly forest cover with 60% of forest, 31% of unimproved grassland and 9% of peatland in upper Hore. Lower Hore has a similar

land use to upper Hore. The Hafren catchment was divided into three reaches, upper Hafren, lower Hafren and Severn at Plynlimon flume where the confluence of Hore and Hafren and an inflow of river Severn and includes Tanllwyth stream. The land use of upper Hafren is 90% of peatland and 10% of unimproved grassland. Forest cover dominates from lower Hafren and Plynlimon flume. Figure 5.1 shows the catchment area and land use proportions for the Plynlimon catchments.

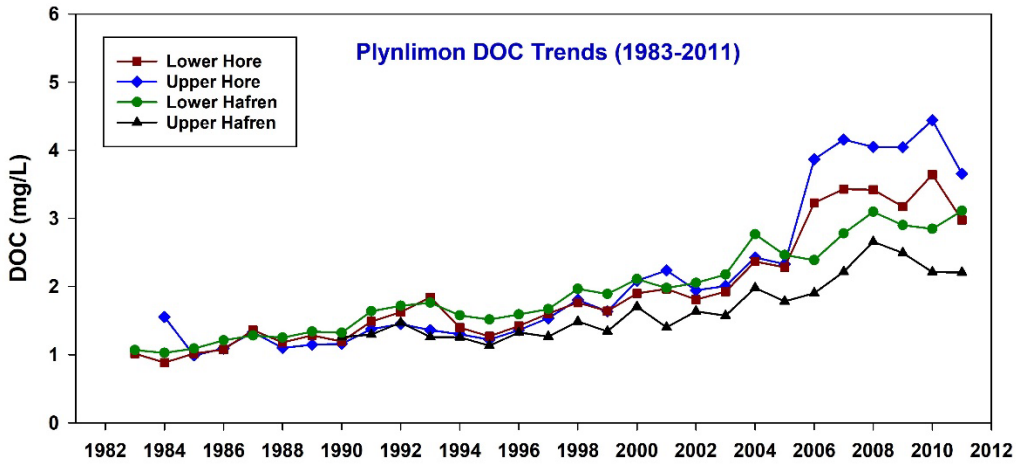


Figure 5.2: Plynlimon DOC Trends (1983-2011) with annual mean DOC (Neal et al., 2013 and Norris et al., 2017)

The trend of DOC concentrations in the upper Severn catchments is shown in Figure 5.2, showing a gradually increasing trend since the late 1990s with high peak concentrations in 2009 (Table 5.1). The monitoring infrastructure was established initially to investigate the impacts of upland conifer plantations on the hydrological cycle in the 1960s. With growing awareness and concern about acid deposition and the effect of forest management practices on river chemistry, hydro-chemical measurements have generated a 30-year uninterrupted record from weekly to monthly which includes pH, alkalinity, nutrients, major cations, anions, trace metals, dissolved organic carbon and dissolved organic nitrogen. The Plynlimon catchment experiment is one of the longest-running

studies in Europe, producing extensive environmental data (Neal et al., 2011; Robinson et al., 2013) with several decades of high-quality climate, flow and water quality data. As an extensive data set, it has helped to improve the understandings of many aspects of hydrology, water chemistry and the consequences of changing the land cover in the catchment (Robinson et al., 2013).

Table 5.1: Plynlimon Mean, Maximum and Minimum DOC concentrations mg/L (1983-2011)

	Upper Hore	Lower Hore	Upper Hafren	Lower Hafren
Mean	2.06	1.87	1.68	1.90
Maximum	21.30	14.00	9.65	11.90
Minimum	0.20	0.20	0.20	0.20
Standard deviation (σ)	1.72	1.30	1.27	1.35
Mean + 1 σ	3.78	3.17	2.94	3.25

5.3 Materials and Methods

5.3.1 Model data

Table 5.2 shows the data used in this study for model application with temporal and spatial coverage. The meteorological data, daily precipitation and temperature, were obtained from the UK Met office (Met Office, 2012). Records of continuous daily water discharge at the Hore and the Hafren were obtained from the National River Flow Archive (NRFA). Discharge records after 2009 were obtained from the Plynlimon experimental site in the UK Centre for Ecology and Hydrology (CEH). Weekly water quality data from the Plynlimon experimental site were also provided by CEH (Neal et al., 2013, Norris et al., 2017).

Table 5.2: Data used in this study for model setup and calibration

Type	Source	Time coverage	Spatial coverage
Land use	Centre for Ecology and Hydrology (UK land cover map)	2007	All the catchment
Meteorology (precipitation and temperature)	Met office	1995-2013	All the catchment, and used for the model calibration and validation
Flows	National River Flow Archive (1995-2009) Centre for Ecology and Hydrology(post-2009)	1995-2013 (Different for all the stations)	Hore catchment (reach 1, 2) Hafren catchment (reach 2,3)
Water quality	Centre for Ecology and Hydrology	1983-2012 (Different for all the stations)	Reach 1,2 for both the Hore and Hafren catchment

5.3.2 The Integrated Catchments Model (INCA-C)

The INCA-C model was employed to reproduce and simulate carbon dynamics in the Plynlimon catchment. INCA-C is a dynamic, semi-distributed catchment scale process-based model of DOC concentrations and fluxes in surface waters (Futter et al., 2007). The INCA model was initially developed to simulate nitrogen (Whitehead et al., 1998a; Whitehead et al., 1998b) and phosphorus (Wade et al., 2002b) at a catchment scale. Several sub-models were later added, including carbon (Futter et al., 2007), sediment transport and soil erosion (Lazar et al., 2010), pathogens (Whitehead et al., 2016), an organic contaminant model (Lu et al., 2016), and microplastics (Nizzetto et al., 2016).

The INCA-C model simulates daily DOC concentrations, flux and water flow in streams. Fluxes and concentrations of DOC are simulated by solving mass balance equations for terrestrial processes while simultaneously solving flow equations. INCA-C requires a daily time series of soil moisture deficit (SMD), hydrological effective rainfall (HER), air temperature and precipitation as inputs. The latter two input datasets were estimated using

a semi-distributed rainfall-runoff model, PERSiST (Precipitation, Evapotranspiration and Runoff Simulator for Solute Transport model, Futter et al., 2014). PERSiST simulates water fluxes at the catchment scale, with a series of user-defined buckets indicating different hydrological processes such as snowmelt, surface runoff generation, soil water storage, groundwater and streamflow. It is particularly designed to provide input time series for the INCA models. In addition, spatial data describing catchment land cover and soil properties, the location of point sources, abstraction and effluent discharge are required to run INCA-C. INCA-C then provides daily time series of flow, DOC and DIC(dissolved inorganic carbon) concentrations at each reach boundary, as well as profiles along with descriptive statistics of these variables at selected sites. INCA-C is calibrated using the time series of observed and modelled daily discharge and point estimates of in-stream DOC concentrations to minimise the differences between observed and modelled values.

5.3.3 Highlighted processes for acidic deposition and enzymatic latch

A detailed description and model equations of INCA-C terrestrial and in-stream carbon processing routines, hydrological routing, and climate controls are well documented by Futter et al. (2007). The initial version of the model has been further modified, and this section will present the highlighted processes related to the analysis in this study.

In the INCA-C model, all carbon procession in soils and surface waters are modelled as a series of first-order differential equations. All rate coefficients in soil layers are dependent on moisture saturation status and soil temperature. INCA-C simulates in-soil temperature by using observed air temperature. The rate coefficient for soil moisture effect is a linear function of soil moisture content. For example, in-soil carbon

transformation operates at a maximum rate when the simulated SMD is at zero, and cease when the simulated SMD is greater than SMD_{max}. SMD_{max} is a calibrated threshold of the maximum soil moisture deficit at which carbon transformations may occur.

The following simplified equations explain the effect of sulphate concentrations and the enzymatic latch on organic carbon sorption and desorption in soil solution. The modification of the sulphate effect to INCA-C has been initially introduced by Futter et al. (2009) and successfully applied to a boreal catchment in Finland. The enzymatic latch process is an innovative additional application in this study. The same processes occur in the organic and mineral soil horizons with different rate constants.

As mentioned above, the change in mass of DOC is controlled as a combination of soil moisture (k_M, d^{-1}), soil temperature (k_T, d^{-1}) effects and the mass of organic carbon sorbed ($c_2 DOC,\ kg\ C\ d^{-1}$) and desorbed ($c_1 SOC,\ kg\ C\ d^{-1}$). The base rates of the sorption(c_2) and desorption(c_1) coefficients are estimated during the model calibration. To simulate soil water sulphate concentrations ($[SO_4^{2-}]$), atmospheric SO_4^{2-} deposition is used as a surrogate in soil solution. Equation 5.1 and 5.2 are simplified equations of the changes in the mass of DOC and SOC (solid organic carbon) with functions of soil chemistry controls on organic carbon sorption and desorption rates.

Equation 5.1

$$\frac{dDOC}{dt} = k_M k_T (c_1 SOC - (c_2 + b_1 [SO_4^{2-}]^{b_2}) DOC)$$

Equation 5.2

$$\frac{dSOC}{dt} = k_M k_T \left((c_2 + b_1 [SO_4^{2-}]^{b_2}) DOC - c_1 SOC \right)$$

The sulphate effect ($b_1[SO_4^{2-}]^{b_2}$) is added on the rate at which DOC is transformed to SOC. At high sulphate concentrations, the transformation from DOC to SOC will be more rapid. Increasing sulphate concentrations should increase the rate at which DOC is transformed to SOC, and hence lower the mass of DOC in solution (Equation 5.1). The equation for the change in mass of SOC is effectively the reverse of that for DOC (Equation 5.2). When b_1 is at zero, the effect of $[SO_4^{2-}]$ is turned off in simulation where organic matter changes from the dissolved to the solid phase.

The "enzymatic latch" mechanism (Freeman et al., 2001b) has been proposed as a mechanism to explain an increase of DOC concentrations observed following a severe drought in peatlands (Worrall and Burt, 2004). The enzymatic latch mechanism hypothesises that the oxygen limitation encourages storing wetland carbon by indirectly decreasing the hydrolytic enzymes' activities which decompose organic matter. Thus, organic carbon is stored and not released during drought, but the subsequent re-wetting process will rapidly increase DOC concentrations in soil waters. The mechanism is simulated in INCA-C as an increase in the rate of SOC to DOC transformation in both organic and mineral soil horizons, and the effect only occurs following a drought

event.The enzymatic latch effect is triggered by a drought event by increasing the rate at which SOC is transformed to DOC. Drought is defined in INCA-C when simulated SMD is exceeding a critical threshold (l_2, mm). The maximum effect of the enzymatic latch is immediately following a drought event and gradually declines to zero. The decay in enzymatic latch effect is simulated as a linear function of the number of days (n) since the drought threshold was crossed (t_{Crit}) divided by the duration of the latching effect in days ($365 \times l_0$). l_0 indicates a return period (years) of the enzymatic latch. In the event that another drought event occurs in which the SMD exceeds the critical threshold (l_2), the number of days since crossing the drought threshold (t_{Crit}) is reset to 0. The enzymatic latch multiplier (l_{Mult}) is defined in Equation 5.3. It should be initialised to a value of 0.

Equation 5.3

$$l_{Mult} = \begin{cases} \left(1 - \frac{t_{Crit}}{365 \times l_0}\right), & SMD_{t-n} > l_2 \ \& \ t_{crit} \leq 365 \times l_0 \\ 0, & all\ other\ conditions \end{cases}$$

Since both sulphate effect and enzymatic latch effect are associated with the same process, the changes in the mass of DOC and SOC, Equation 5.1 and Equation 5.2 can be updated accordingly to Equation 5.4 and Equation 5.5 respectively. The rate of production of DOC from SOC is controlled by the desorption(c_1) coefficient and the enzymatic latch effect ($l_{Mult} l_1$), where l_1 (d^{-1}) is the enzymatic latch rate coefficient. The rate of DOC transformation to SOC is controlled by the sorption(c_2) coefficient and the sulphate effect ($b_1[SO_4^{2-}]^{b_2}$). The range of parameters is achieved during the calibration process.

Equation 5.4

$$\frac{dDOC}{dt} = k_M k_T((c_1 + l_{Mult} l_1)SOC - (c_2 + b_1[SO_4^{2-}]^{b_2})DOC)$$

Equation 5.5

$$\frac{dSOC}{dt} = k_M k_T\Big((c_2 + b_1[SO_4^{2-}]^{b_2})DOC - (c_1 + l_{Mult} l_1)SOC\Big)$$

5.3.4 Model parameterisation and Generalised sensitivity analysis

5.3.4.1 The calibration procedure

The INCA-C model has been applied to the whole upper Severn Catchment at Plynlimon using a long run of data from 1995-2013 to calibrate and validate the model. The period was split in two: 2000-2013 for calibration, and 1995-1999 for validation. The period of model evaluation was set according to the availability of daily meteorological inputs. Given that the two catchments have quite a different land uses, separate INCA-C models were set up for the two catchments to compare catchment responses under a changing climate. Reach boundaries were drawn at Plynlimon experimental monitoring stations where long-term flow and water quality data were available. Manual calibrations were carefully conducted in which land phase hydrology and carbon processing parameters were allowed to vary so as to observe the correspondence between modelled and observed stream flows and DOC. The initial values of parameters and those ranges were found from the literature (Futter et al., 2007). The manual calibration was repeated and conducted until the Nash-Sutcliffe(NS) statistics (Nash and Sutcliffe, 1970) showed a

reasonable agreement between simulated and observed values of flow and DOC. The parameter sets derived from manual calibrations were used as the starting point for a Monte Carlo analysis in which selected carbon processing and hydraulic parameters which could affect modelled DOC concentrations were allowed to vary by ± 20%. The Monte Carlo simulation produced an ensemble of 10,000 model runs, and those results were assessed to find the best performing parameter sets. Model performance was assessed using the sum of Nash-Sutcliffe statistics representing correspondence between modelled and observed DOC in upper and lower Hafren and upper and lower Hore, respectively. Best performing model runs were selected and used for further analysis.

5.3.4.2 General Sensitivity Analysis

A General Sensitivity Analysis (Hornberger and Spear, 1980; Spear and Hornberger, 1980) using Monte Carlo Simulations was applied in order to investigate model complexity and understand parameter influence on model simulations. The method is also conversely to inform those parameters that appear to exert little influence on model results as well. The main strategy is to incorporate uncertainty in the model simulation by specifying the parameter values from probability distribution functions, rather than point values. The Monte Carlo simulations are employed with chosen parameter values from the specified distributions, which should reflect the feasible parameter ranges. The results of Monte Carlo simulations are classified into those that are considered behavioural and non-behaviour with respect to the pre-defined criteria set. The definition of behaviour and non-behaviour is problem-dependent, and the criteria can be for general trends or extreme events. The each of model results from the Monte Carlo parameter sets is compared with the observed values via the pre-defined behaviour criteria algorithm (in this study,

increasing trends in DOC concentrations) in order to determine the occurrence or non-occurrence of the required behaviour. Then, the parameter sets will be classified to the one associated with the occurrence of behaviour (B) and one with the non-behaviour(N). Similar methodologies have been applied to provide a probabilistic procedure for model calibration and to identify critical parameter uncertainties in phytoplankton models (Whitehead and Hornberger, 1984), in a model application of phosphorus and macrophyte dynamics in River Kennet (Wade et al., 2002d), and in new model development of dissolved oxygen, Q^2 (Cox and Whitehead, 2005).

The Hornberger and Spear's generalised sensitivity analysis looks for the difference between the behavioural and non-behavioural sets for each parameter. It does so by comparing the cumulative distribution of that parameter in each set. Where there is a strong difference between the two distributions for a parameter, it may be concluded that the simulations are sensitive to those parameters. If the two distributions are very similar, it may be concluded that the simulations are not very sensitive to those parameters. A quantitative measure of the difference between two distributions can be calculated using the nonparametric Kolmogorov-Smirnov d statistic ($d_{m,n}$). The Kolmogorov-Smirnov two sample test is computed as following Equation 5.6:

Equation 5.6

$$d_{m,n} = sup_x|S_n(x) - S_m(x)|$$

where S_n and S_m are the sample distribution functions corresponding to the behaviour (m) and non-behaviour (n) values of a given model parameter and this statistic is the same as

that used in the Spear and Hornberger (1980). The value of $d_{m,n}$, d statistic, can be used as an index of relative difference for model parameters. The highest $d_{m,n}$ value among all parameter set is the most influent model parameter. This approach is a nonparametric method of sensitivity analysis in that it makes no prior assumptions about the variation or covariation of different parameter values, but only evaluates sets of parameter values in terms of their performance.

In this chapter, the aim of the behaviour analysis is specific to understand which particular parameters will have a significant influence on the rising trends of modelled DOC concentrations. The general sensitivity analysis was applied to the Hore and Hafren INCA carbon models, respectively. Based on a prior general sensitive analysis of the INCA Carbon model (Futter et al., 2007, Futter and de Wit, 2008), following carbon and non-carbon processing parameters were identified as the most significantly influential on model behaviours and considered to the analysis:

- In Forest cover, organic layer: DOC mineralisation rate, the rate of change in mass of DOC to SOC, the rate of change in mass of SOC to DOC, retention volume of organic layer, minimum organic layer flow
- In Forest cover, both organic and mineral layer: soil temperature rate multiplier, the rate of change in mass of SOC to DOC
- -In Peatland cover, organic layer: retention volume of organic layer, minimum organic layer flow
- In Peatland cover, mineral layer: the rate of change in mass of DOC to SOC
- In Peatland cover, both organic and mineral layer: the rate of change in mass of SOC to DOC, and maximum SMD at which carbon processing can occur.

In addition, parameters related to the enzymatic latch process and sulphate multipliers were also included in the analysis. The feasible space of model parameters was drawn and sampled randomly to generate 10,000 different parameter sets. The calibrated INCA-C model was then run with each of these parameter sets. The 2000 best performing model runs were retained for further sensitivity analysis. Model performance was assessed using the sum of the NS statistics representing correspondence between modelled and observed DOC in the Upper and Lower reaches for the Hafren and Hore.

5.3.5 Projected future climate

In this study, climate scenarios generated by the weather@home system (Massey et al., 2015, Guillod et al., 2018) were applied to explore the potential impacts of climate change on hydrology, the carbon in catchments. The modelling system- weather@home (hereafter, w@h) consists of a global climate model (GCM) with a nested regional climate model (RCM) driven by sea surface temperatures as well as other forcings. It enables the production of a large number of ensembles of weather events with the help of volunteers distributed computing across the world. In this model application, weather@home2 (hereafter w@h2, which includes an improved land surface scheme) was used to generate 30-year-long weather time series of rainfall and temperature projections for three different periods. The data set is specifically designed to support a risk-based approach for the study of extreme events such as drought and heavy precipitation, both of which require the spatial consistency of hydro-meteorological data sets (Guillod et al., 2018). The scenarios in each future time slice all follow the Representative Concentration Pathway 8.5 (RCP8.5) and sample the range of sea surface temperatures and sea ice changes from CMIP5 (Coupled Model Intercomparison Project Phase 5) models. A

number of daily and (or) monthly output variables are available from the RCM including temperature, precipitation, surface air humidity, mean sea level pressure, and potential evaporation. To assess the worst possible future condition of water quality, the RCP8.5 was selected as it is the most severe scenario among the current available future scenarios. Of particular relevance for water quality modelling is the changes in intense rainfall, which can affect leaching and drought, which influences chemical retention times and river flows. From a 25 x 25 km grid over Europe, 100-time series are available for each of time slices which represent baseline (BL, 1975-2004), near future (NF, 2020-2049), and far future (FF, 2070-2099) climatic scenarios. Outputs from grid cells include Plynlimon catchment in Wales were averages to generate input time series for water quality modelling application (Figure 5.3).

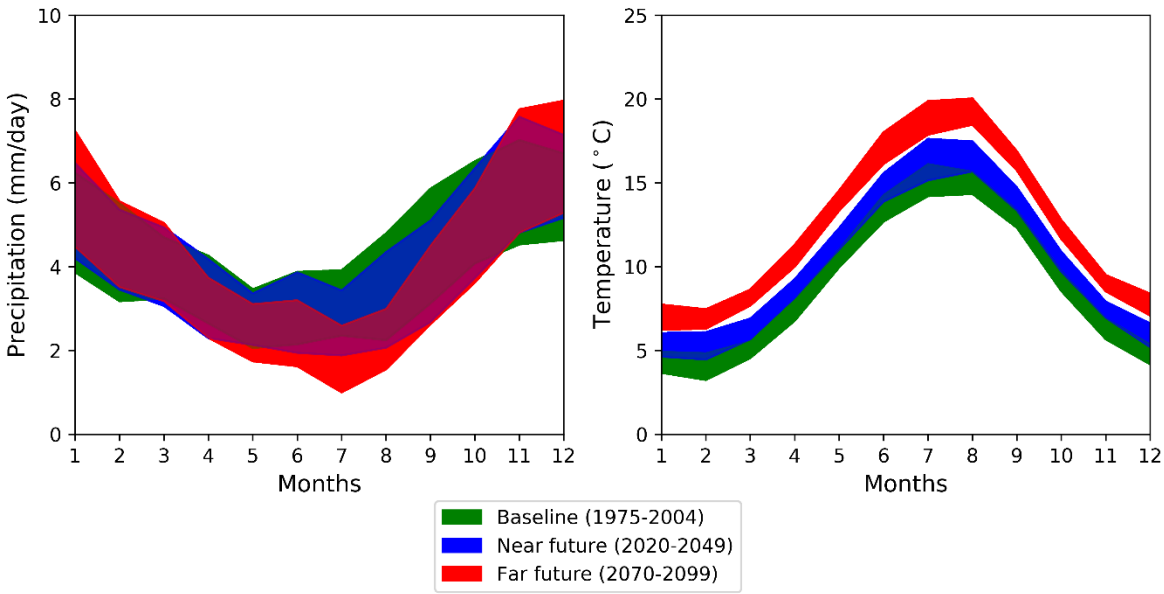


Figure 5.3: Precipitation and temperature projections (w@h2) for the Plynlimon catchment

The precipitation data was bias-corrected in order to reproduce the patterns and distributions of observed data better, and a linear approach was applied (Guillod et al., 2018). The bias correction method was conducted by using spatially-variable monthly bias-correction factors, obtained by the ratio of the modelled and observed monthly average precipitation. However, very little bias was observed from temperature, so temperature bias was not explicitly corrected. More details about weather@home2 data and bias-correction method can be found in Guillod et al. (2017) and Guillod et al. (2018). Figure 5.4 shows the comparison of bias-corrected precipitation (purple) and raw precipitation from w@h2 (grey) with observed precipitation. A dry bias during summer months on precipitation was well corrected.

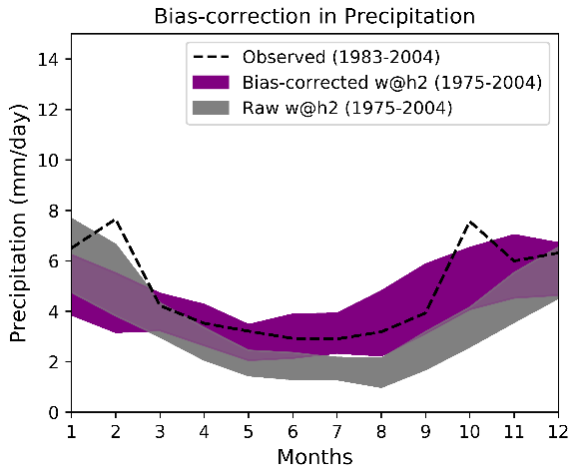


Figure 5.4: Comparison of Bias-corrected and raw w@h2 in precipitation in Plynlimon catchment

5.3.6 Sulphate deposition scenarios

Deposition scenarios from the Emissions Monitoring and Evaluation Programme (EMEP) were applied for this study to include the impacts of historical, present, and future acidic deposition. Historical sulphate deposition time series from 1970 to 2010 on a 50km x

50km grid resolution was taken from the output of the EMEP/MSC-W model, as well as the future projection on the depositions until 2030 was also provided under the Gothenburg Protocol Currently Legislated Emissions (CLE) scenario (Posch et al., 2007). The chemical transport model developed at the Meteorological Synthesizing Centre - West (MSC-W) is called the EMEP/MSC-W model. After 2030, the deposition is assumed to remain constant until 2090. More information on EMEP data can be found in Schöpp et al. (2003).

Reduction in acidic depositions is believed to have helped acidification recovery of European rivers and lakes (Skjelkvåle et al., 2003). As explained in the previous section, organic matter solubility in the INCA-C model is controlled by the concentrations of strong acid anions in soil solution (Futter et al., 2009; Futter et al., 2011). As soil solution data are not readily available, sulphate deposition data are assumed to be a reasonable surrogate for soil solution strong acid anions concentrations. Deposition for the forest land cover type was applied as an input to the modelling study. A smoothed sulphate deposition time series was created using estimates of average monthly wet sulphate deposition. The linearly interpolated monthly sulphate loadings were multiplied to daily precipitation depth to produce monthly average deposition values. These monthly deposition estimates were then assumed to be representative of the soil solution concentration throughout the soil profile. Estimated deposition showed that the peak was in the 1980s and rapidly declined since then. For the calibration, a single deposition series was provided. For the climate projections, 300 deposition time series were produced according to each input precipitation for each climate realisations for each of w@h2 time slices (baseline, near future and far future).

5.4 Results and Discussions

5.4.1 Model evaluation

The Monte Carlo simulation produced an ensemble of 10,000 model runs and those results were assessed based on observed values of flow and water quality (DOC) at reach 2 for both the Hore and Hafren catchments (Lower Hore and Lower Hafren). The best model parameter set was selected for the Hore and Hafren model and used for the rest of the study. A combination of model evaluation measures was used to determine the best performance model including (a) visual assessment of time series, (b) model performance statistics and (c) total DOC loads estimation, and selected model performance was described.

Figure 5.5 shows the calibration and validation results of flow and DOC concentrations at Lower Hafren and Lower Hore. It is seen that the models are successfully representing a temporal pattern of DOC concentration and flow. The model simulations did not always capture the magnitude of peaks in DOC concentration. However, the models were able to capture the timing of peaks and seasonal patterns of DOC concentrations.

Table 5.3 shows the performance indices of the INCA model in Plynlimon catchments (1995-2013). The performances were measured by Nash-Sutcliffe Efficiency (NSE), log NSE for streamflow and percent bias (PBIAS) for DOC concentration. Perfect matching was represented by the value of one for NSE. A negative value of NSE means that the average observed value is a better predictor than the simulated values of model simulation. NSE is one of the most popular indices for evaluating models, but it is sensitive to high or low extreme values. Therefore log-transformed discharge (log NSE) was evaluated to consider the variability balanced at low values (low flows). The NSE values of daily

simulated streamflow in both the Hore and Hafren catchments show acceptable ranges for a whole period of model evaluation (1995-2013), 0.68 and 0.66, respectively. The log NSE values were improved from the values of NSE in this study, which is 0.73 and 0.68, respectively, for the Hore and Hafren catchments. The simulated daily flow provided an acceptable reproduction of the observed flow, including a low flow period, particularly given the flow in Plynlimon is generally small.

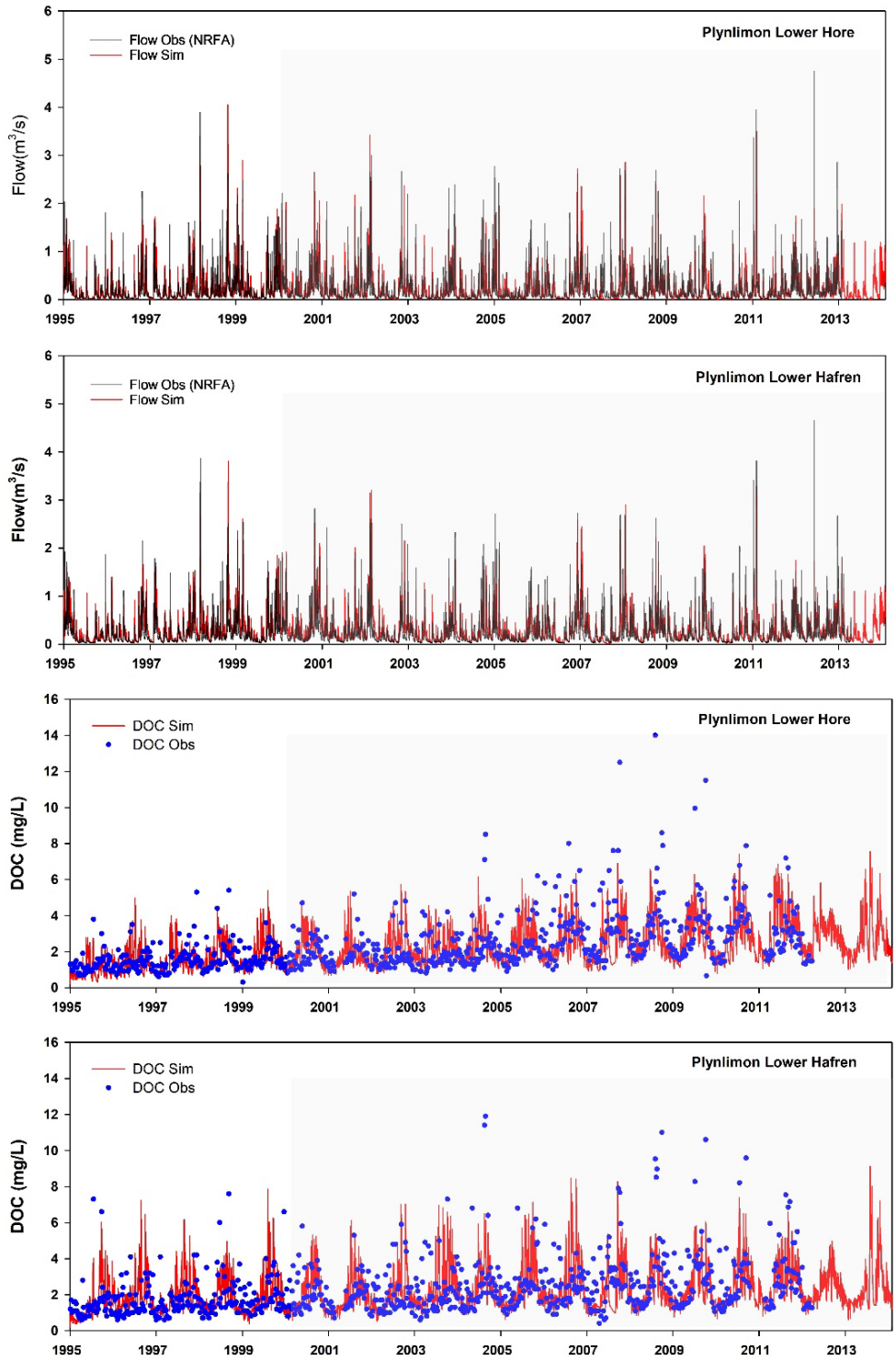


Figure 5.5: INCA-C calibration and validation results (Flow and DOC) at two locations in Plynlimon catchment. The grey-shaded area is the time period used for calibration. The unshaded area is used for validation.

Table 5.3: Performance indices of INCA -C calibration and validation (NSE: Nash and Sutcliffe index, PBIAS: Percent bias)

	2000 - 2013 calibration			1995 - 2000 validation		
	Flow		DOC	Flow		DOC
Reach	**NSE**	**logNSE**	**PBIAS**	**NSE**	**logNSE**	**PBIAS**
Lower Hore	0.68	0.69	8.94	0.67	0.80	4.96
Lower Hafren	0.67	0.68	8.93	0.61	0.72	-7.77

Concerning the DOC simulation, PBIAS is less than 10% bias for both catchments, and INCA-C results can be considered satisfactory. PBIAS measures the average tendency of the modelled values to be larger or smaller than observed counterparts. The optimum value is 0, and negative values indicate a model overestimation bias while positive values mean an underestimation of modelled values. As suggested by Moriasi et al. (2007), satisfactory simulation can be indicated by PBIAS from ±25% up to ±70% for water quality variables. The sulphate deposition data application was helpful to the model calibrations. Model calibrations without the sulphate deposition data were shown -42% and -16% of PBIAS values for the Hore and Hafren, respectively.

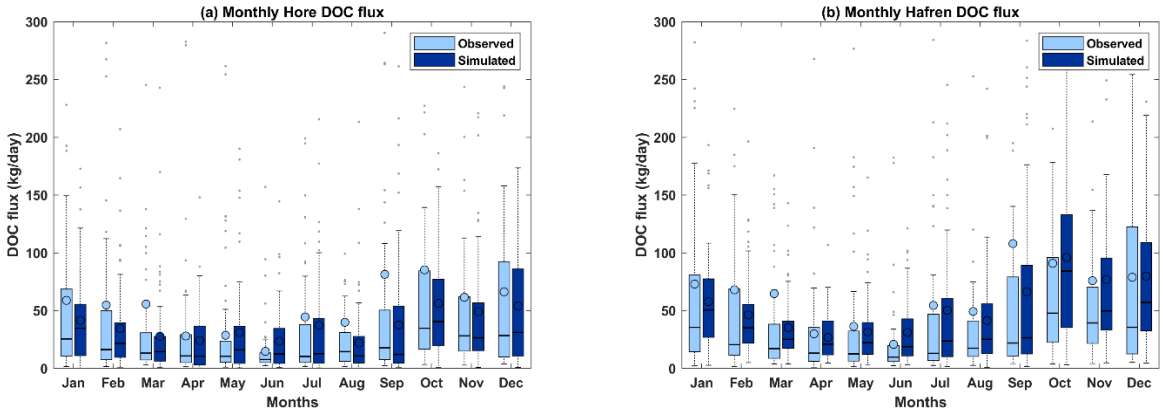


Figure 5.6: Simulated and Observed Monthly (in-stream) DOC load (kg/day) at Hore (a) and Hafren(b) catchments

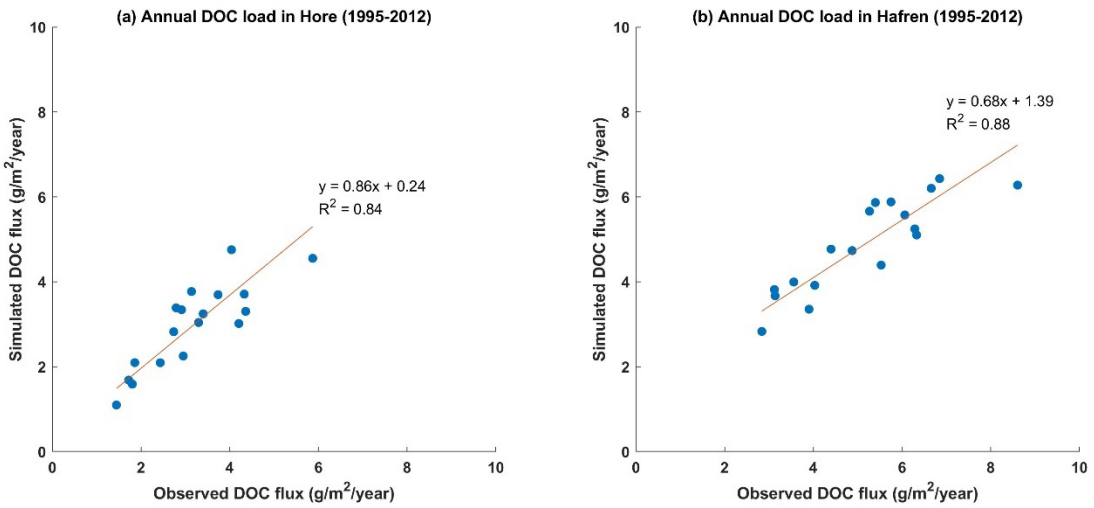


Figure 5.7: Simulated and Observed Annual DOC load at Hore(a) and Hafren(b) catchments

The monthly in-stream DOC flux (kg/day) was calculated to demonstrate whether the model reproduced similar seasonal loads. Figure 5.6 showed the simulated and observed monthly DOC loads, which yielded similar results and well-reproduced seasonal patterns. The precipitation-driven annual DOC fluxes showed a great agreement between observed and simulated DOC fluxes (Figure 5.7). The degree of fit in terms of fluxes load with an R^2 of 0.84 (for Hore catchments) and 0.88 (for Hafren) are acceptable.

The many measures used for evaluating model performance showed generally satisfactory simulations, in terms of reproduction of the seasonal patterns of both streamflows and DOC concentrations over a long period, given the uncertainty that characterises both model results and measurement data values. It suggests the model can be used with confidence in predicting future projections.

Model simulations were forced by a combination of gridded data for sulphate deposition and daily weather inputs and calibrated against point observations of streamflow and

water chemistry. The gridded data used here provided complete, gap-free time series, which is a prerequisite for INCA modelling. The EMEP gridded data are widely used when modelling future trajectories of recovery from acidification (e.g., Posch et al., 2007, Futter et al., 2009). Within a region, e.g., the Severn River catchment, long-term temporal variation in sulphate deposition is higher than spatial variation. Thus, it was considered appropriate to use gridded EMEP data as an input to the model. Ledesma and Futter (2017) showed that gridded climate products could give as good or better simulations of streamflow when compared to observational weather data. This finding, combined with the difficulties in obtaining gap-free observational weather data and the need for consistency when applying model parameterizations based on the present-day to future conditions motivated the use of gridded climate data throughout this study. Point observations of streamflow and water quality are also spatially-integrated values as they are the result of processes occurring throughout the catchment.

5.4.2 Sensitivity analysis -sensitive parameters

The criteria for behaviour were defined by examining the observed data and identifying a range of values for monthly mean and maximum DOC concentration from 1995 to 2013. Behaviour criteria were set to include an increasing trend of DOC concentration and reasonable ranges of monthly mean and maximum DOC concentrations for selected years and months. Summer months (August, July) were chosen and year of 1996, 2002, 2004 and 2008 were all selected to demonstrate increasing trends of DOC concentration. The top 2,000 model runs were retained to proceed with the general sensitivity analysis. Each simulation result consists of the parameter vector itself and the behavioural outcome, i.e. whether the particular parameter vector gave rise to the behaviour or not. The criteria

used to separate behavioural and non-behavioural parameter sets. Once the parameter set space was partitioned into two groups, the degree of difference between the two groups identified the model parameter influence. If the distribution of non-behavioural values of a model parameter does not separate from the behavioural one, this indicates that the parameter does not have a significant influence on model results. The Kolmogorov-Smirnov two-sample test was used to evaluate model parameter sensitivity. Following refinement of the parameter ranges, the 2,000 simulations for the Hore and the Hafren produce a mixture of behaviours (m) and non-behaviours (n) as expected.

In this study, 15 parameters for each type of land cover were applied to the sensitivity analysis, including parameters related to carbon processing, microclimate, hydraulic, enzymatic latch, and sulphate effects. As three land types (Forest, unimproved grassland, and peatland) were employed in this study, forty-five parameters in total were examined in the whole sensitivity analysis. The parameters and corresponding statistics ($d_{m,n}$) that are significantly above the 95% level are ranked in order of importance and listed in Table 5.4. Eleven out of forty-five parameters are significant at the 95% level or greater in the Hore catchment. Nine for the Hafren catchment are found influential, and the relatively large separations denoted by the K-S test $d_{m,n}$ explains that these parameters are important for obtaining the correct behaviour in a simulation. In this research, nine parameters contribute significantly to simulate an increasing trend of DOC concentrations in Plynlimon. Among the important parameters, the soil temperature response, sulphate effect (b_1) multiplier, and organic layer hydraulic parameters (residence time and retention volume) have a significant impact on both catchments. It is not a surprise that the temperature effect has ranked as the most important parameter to influence model

behaviour and DOC concentrations. In soil process, sorption and desorption rates of organic carbon are dependent on soil temperature and moisture status, which decide the total mass of DOC and SOC in soil layers and in-stream DOC concentrations. Parameters associated with the effect of sulphate and enzymatic latch revealed to the influential parameters in Plynlimon (Table 5.4).

Table 5.4: Kolmogorov-Smirnov Statistics (significant at 95% level or greater) in the GSA of parameters influencing model behaviour

Rank	Parameter	Land cover type	$d_{m,n}$
Hore catchment (m= 305, n= 1695)			
1	Organic layer DOC mineralisation	Unimproved grassland	0.25
2	Soil temperature response	Unimproved grassland	0.25
3	Organic layer retention volume	Unimproved grassland	0.25
4	Organic layer b_1, sulphate multiplier	Unimproved grassland	0.18
5	Soil temperature response	Peatland	0.12
6	Organic layer b_1, sulphate multiplier	Forest	0.11
7	Organic layer residence time	Unimproved grassland	0.10
8	Soil temperature response	Forest	0.10
9	Organic layer b_1, sulphate multiplier	Peatland	0.09
10	Organic layer residence time	Forest	0.09
11	Mineral layer retention volume	Forest	0.08
Hafren catchment (m=132, n=1868)			
1	Soil temperature response	Unimproved grassland	0.57
2	Organic layer enzymatic latch rate	Forest	0.25
3	Soil temperature response	Forest	0.25
4	Organic layer SOC to DOC	Peatland	0.22
5	Organic layer residence time	Unimproved grassland	0.19
6	Organic layer retention volume	Unimproved grassland	0.14
7	Organic layer residence time	Forest	0.14
8	Organic layer critical SMD threshold	Unimproved grassland	0.13
9	Organic layer b_1, sulphate multiplier	Peatland	0.13

5.4.3 Climate Change in Plynlimon

According to the results of w@h2 climate scenarios, Plynlimon precipitation is expected to show a slight increase in winter months and a clear decrease in summer months (Figure 5.3). The trends are gradual from the near future to the far future time period. In the far future, the average increase of December precipitation is about 15% whilst a 43% average decrease is expected in July. The total annual average precipitation at Plynlimon is expected to decrease by 3% in the near future and 6% in the far future. Projected changes in the seasonal patterns of precipitation are more significant than the total annual average precipitation. The average daily air temperature is projected to increase for all months for both future time periods (Figure 5.3). The mean temperature changes are from 9.01℃ as the baseline to 10.22℃ for the near future and to 12.28℃ for the far future scenario.

5.4.4 Simulated flows and DOC with projected future conditions

The calibrated INCA Carbon model was run with w@h2 climate scenarios and acidic deposition scenarios for the baseline, near future and far future for both Hore and Hafren catchments. The flows in Hore and Hafren show a clear seasonality, with low flows in summer months and high flows in winter months (Figure 5.8). The results of w@h2 driven by the INCA model, future climatic change will enhance this seasonal flow patterns. In particular, summer flows will decrease significantly, while changes in winter flows will be mild. This is probably because the projected lower summer rainfall and increased air temperatures in the Plynlimon area contribute to decrease flows, up to an average of 51% in July under the far future scenario. In winter months, projected flows are expected to increase the range of 1-5% in the near future, and 3-15% in the far future compared to the baseline. However, Plynlimon is a small stream with an average flow

with 0.234 m^3/s and 0.205 m^3/s for Hafren and Hore, respectively (average between 1973-2009). It is quite difficult to represent in the model simulation, so the range of each future time slices is overlapping with the range of observed flow period. However, the median lines are showing the degree of changes in each time period (Figure 5.8 (a) and (b)).

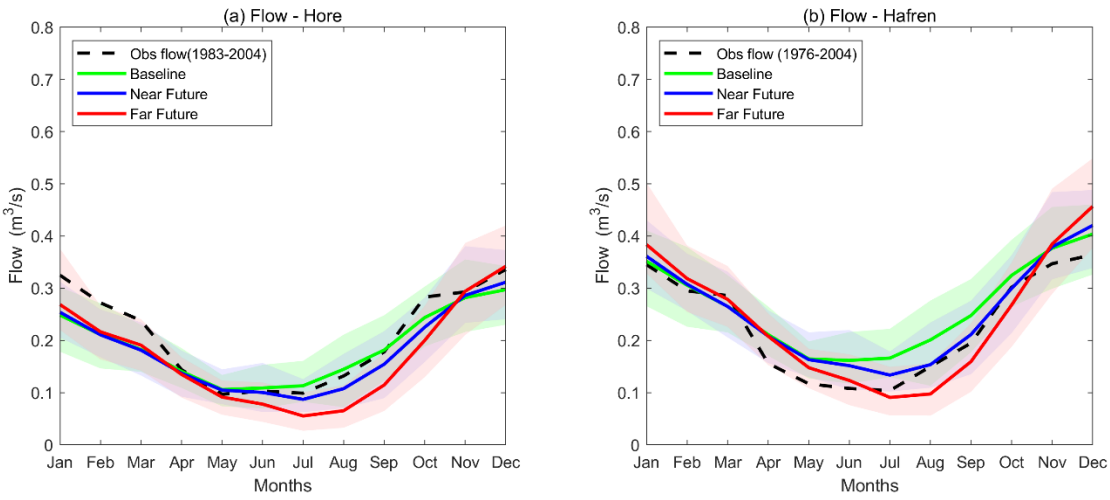


Figure 5.8: Impact of climate change on flow at Hore (a) and Hafren (b) catchments

The INCA model results show a substantial increase in DOC concentrations in the near future and the far future from the baseline for both catchments. The DOC concentrations in the far future are stabilised and slightly reduced in the summer months after the near future. Figure 5.9 shows the DOC monthly average and maximum concentrations driven by the w@h2 climate scenarios. For the Hore catchment, the simulated baseline values slightly overestimate DOC concentrations from April to July compared to the observed values (Figure 5.9 (a) and (c)). The rest of the months are represented in a reasonable range. It is worth noting that the time series of baseline condition started in 1975 and DOC monitoring has started since 1983, which may cause a slight discrepancy between

baseline and observed values. The Hafren catchment, observed values lie within the range of the simulated values of baseline (Figure 5.9 (b) and (d)).

In the near future for both catchments, there is a significant increase in later summer to autumn (August, September, and October). In the forest-dominant catchment at the lower Hore (Figure 5.9 (a)), there is a slight reduction in DOC concentration the summer months in far future compared to the near future. However, there is still a big surge from the baseline level. Hafren (Peatland-dominant) catchment shows a different trend (Figure 5.9 (b)). From September to November, there is an even greater increase in DOC in the far future than the near future while the summer months of DOC concentrations are slightly decreased.

Several droughts and associated low flows in summer months and subsequent re-wetting in early autumn in peatlands could destabilise peatland carbon stocks and enhance in-stream DOC production rates. This process is greatly affected by the combination of temperature and water availability in pore water as well. The w@h2 analyses projected less summer precipitation with warmer temperatures, which seems to cause an enzymatic latch effect in peatland soils. In the model simulations, the soil moisture deficit is a surrogate measure of soil dryness, and it is representative of the difference between the water currently in the soil and the water-holding capacity of the soil. The SMD is a key driver of INCA-C results as in-soil all carbon transformation rates are assumed to be soil moisture dependent. Additionally, in the INCA-C model application presented here, SMD is compared to a drought threshold below which an enzymatic latch effect is triggered. The SMD threshold of 31mm was applied for this study in Plynlimon, and if SMD values were greater than 31mm depth, the enzymatic latch effect was triggered. The threshold

was achieved from the calibration process, and it is consistent value with a measurement from Plynlimon experiment site (Hudson, 1988).

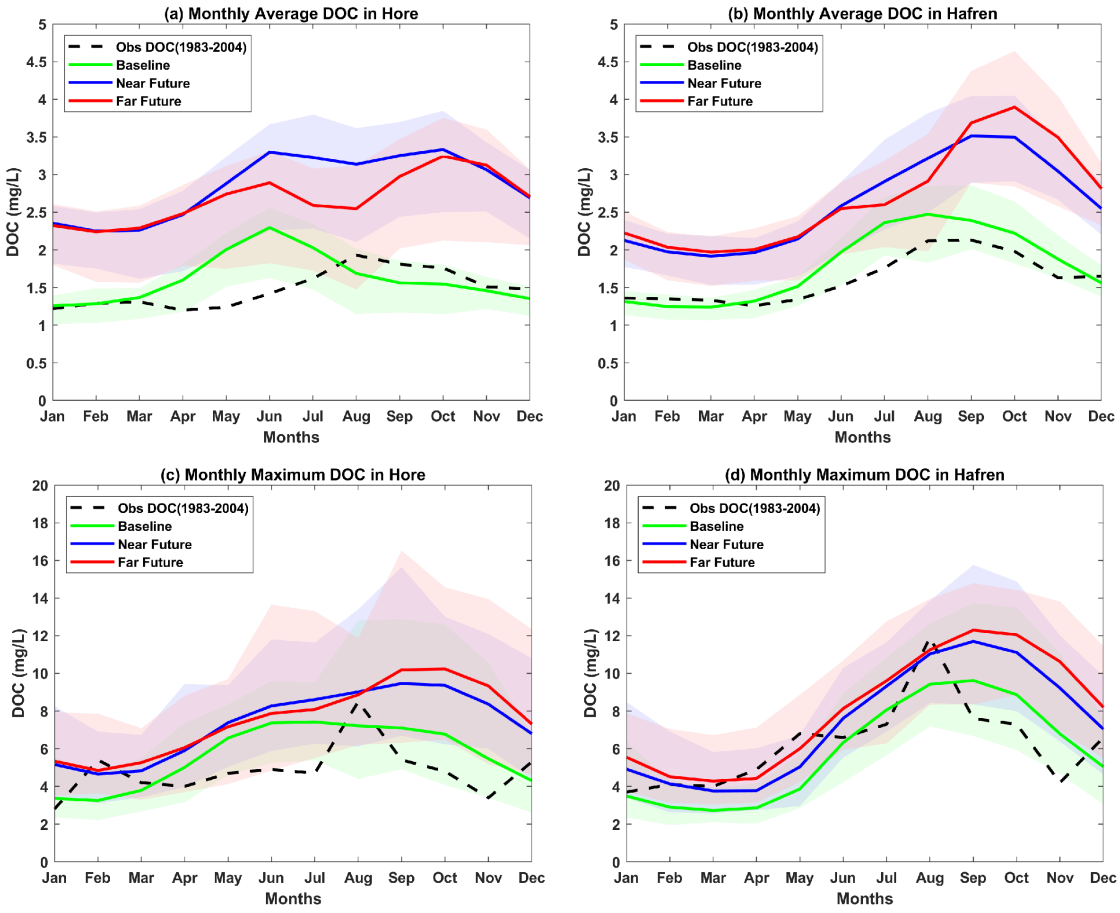


Figure 5.9: Impact of climate change and acidic deposition (up to 2030) on the Upper Severn catchments; the simulated mean monthly DOC concentrations at the Hore (a) and Hafren (b), the simulated monthly maximum DOC concentrations at the Hore (c) and Hafren (d)

Figure 5.10 shows the changes in the frequency and duration of soil moisture droughts (when SMD > 31mm) for the two future scenarios that the enzymatic latch effect was triggered. The w@h2 future scenarios suggest a substantial increase in the duration of each soil moisture droughts (Figure 5.10 (b)) than its frequency. It can be explained that during the drought, organic carbon is not mobilised. However, subsequent re-wetting after

a long and severe drought, DOC production rates are accelerated from the peat. When the peatland soils become saturated and are mobilised, they flush DOC into the stream. Therefore post-drought surge of DOC concentrations become visible in autumn months (Figure 5.9 (b)).

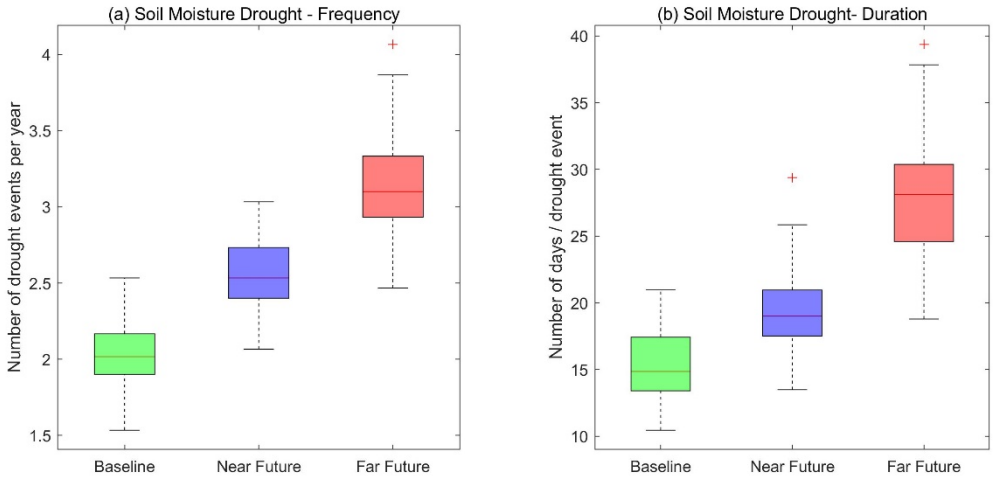


Figure 5.10: Changes in frequency (a) and duration (b) of soil moisture droughts in Plynlimon

Maximum DOC concentrations show a substantial increase in the wet season, while a small increase in the summer months is projected at around 20% (Figure 5.9 (c) and (d)). The projected sulphate deposition continues to decline until 2030 and remains constant after. The projected changes have inter-annual variability, but no intra-annual variability. Therefore, the sulphate decline could affect the stabilisation of DOC concentrations in the far future period. However, the changes in seasonal variability of DOC concentrations are most likely driven by climatic changes and enzymatic latch processes during droughts. While this chapter has only presented results for the Plynlimon in Wales, enzymatic latch simulations in peatlands can also be extended to understand the other peatlands catchments in the UK.

5.5 Conclusions

In this study, the impact of climate change and acidic deposition on flow and water quality was analysed for an uplands catchment, Plynlimon, in Wales, UK. This chapter focused on the dynamics of droughts and sulphate deposition in peatlands by including those mechanisms in water quality modelling. Our study is the first to model enzymatic latch mechanisms in the UK catchments to explain a complex biochemical process in a multi-parameterised process-based model. The main findings are focused on assessing whether the rising trends of DOC concentrations would be continued or stabilised in the future. Taken across from our future climate, sulphate scenarios and the identified enzymatic latch parameters, the projected DOC concentrations in Plynlimon will continue to increase in the near future and then will be stabilised in the far future. However, in the far future, the seasonal patterns of DOC concentrations will change, with a post-drought DOC surge in autumn months. Finally, this chapter and modelling has provided a basis for time series analysis of DOC concentrations in the River Severn. The subsequent transfer of DOC rich waters into the Thames is considered in **Chapter 6**.

6 The Severn - Thames Transfer: Dynamic water quality modelling assessment

6.1 Introduction

This chapter introduces how the Severn-Thames Transfer scheme would work, if it is implemented in the future water resources system, and what its expected performance would be, especially under extreme droughts. Water resources management planners are increasingly aware of the effects that climate change and population growth will have on the current, conventional water resources management plan (Borgomeo et al., 2014; Hall et al., 2019b). This requires an understanding of the vulnerability and sustainability of the current water supply options, and the development of new water resources options to cope with more intense, longer, and more frequent droughts.

From a water industry perspective, higher level of pollutions in the source of water supply are huge concern and it is directly link to their operation costs. If the quality of sources (such as rivers, reservoirs or groundwaters) is poor, water utilities might decide to cease a withdrawal of water temporary, or the water cannot be adequately treated or cause additional treatment costs. For the sustainable water supply, achieving a desirable ecological goal is imperative. For example, the implementation of EU Water Framework Directive set the thresholds to achieve acceptable water quality and ecological status for rivers, lakes, wetlands, and groundwater. In the UK, phosphorus has been considered to be the key parameter controlling ecology status (Jarvi et al., 2002); thus phosphorus control is determined to be the main management strategy to avoid the growth of algal blooms and maintain healthy ecological status (Crossman et al., 2013; Whitehead et al., 2013). In addition, higher nitrogen concentration may affect the productivity and

biodiversity in freshwater systems (Barker et al., 2008). As established in **Chapter 5**, higher DOC concentrations in drinking water can cause human health issues and the extra treatment costs. It affects watercolour, transparency, and pH, which all directly link to stream ecology and increases in DOC represent a change in the quality and ecology of streams, thereby creating difficulties from a Water Framework Directive (WFD) perspective (Whitehead et al., 2006).

In order to incorporate the wide range of drought intensification into this analysis, a new regional climate model, weather@home2 (w@h2) (Guillod et al., 2018), that delivers a large ensemble of spatially coherent synthetic drought simulations is employed in the analysis. Therefore, this research highlights a water quantity-quality modelling approach to explore a specific water management option, the Severn- Thames Transfer during future droughts.

This chapter aims to provide an innovative modelling representation of the hypothetical inter-basin water transfer scheme, the Severn-Thames Transfer, under extreme spatially extensive drought events. This chapter introduces how the Severn-Thames Transfer scheme would work if implemented in the future water resources system and its expected performance and impacts on the specified water quality variables and aquatic environments.

The specific objectives are:

1) to identify future hydrological drought events and how the characteristics of droughts and coincident droughts between two catchments would change under projected climate change;

2) to identify the hypothetical operation rules of the Severn-Thames Transfer scheme by investigating water availability in the River Severn for a transfer during drought periods in the River Thames;
3) to assess the potential impacts of water transfer on water quality and explore trade-offs between additional water supply and water quality degradation in a given time frame.

In this study, the inter-basin transfer option via a pipeline was used as the case, with the ultimate aim of providing a rigorous evidence base for the suggested modelling strategy.

6.2 Methods

6.2.1 Modelling framework

The Severn-Thames Transfer scheme is a hypothetical transfer scheme and does not yet exist; hence a modelling approach is essential. The methods presented here seek to assess the potential effectiveness and feasibility associated with the Severn-Thames Transfer option. Figure 6.1 provide a map of the Severn-Thames Transfer study area and locations of proposed abstraction and discharge points. Figure 6.2 shows a flowchart of the combined modelling framework used in this analysis.

Chapter 4 introduced the extensive modelling application on river flows and water quality in both the Thames and Severn, and the same modelling setup has been used for this analysis. The hydrological, water quality and water resources modelling are driven by synthetic drought event sets, w@h2 (Guillod et al., 2018). Long and continuous time series were generated for the UK over three periods: the recent past (1975–2004 for a baseline), the near future (2020–2049), and the far future (2070–2099). A hundred-time

series are available for each time period and represent 100 different trajectories of weather patterns. According to the hydrological model, the projected future river flows predict the drought characteristics and coincidence in two catchments. Then, hydrological modelling and water system simulations predict the status that triggers water use restrictions and water shortages for each future climate realisation. In this case, the reservoir storage levels are the status of the system that will impose water use restrictions and potentially trigger water transfers. Based on the output from each simulation, the hypothetical operation rules of the Severn-Thames Transfer are set, which specify the timing and volume of transfer. Then, the expected performance and water quality impacts of the Severn-Thames Transfer are assessed.

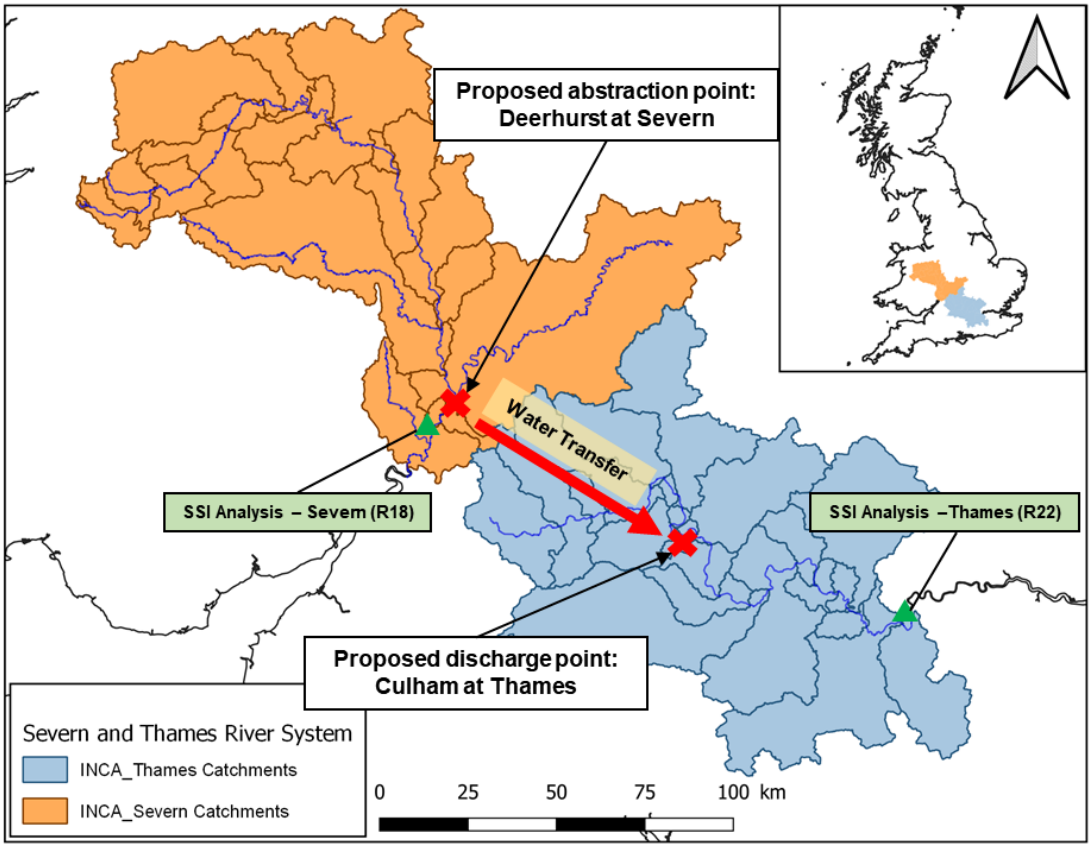


Figure 6.1: A map of the Severn-Thames Transfer area

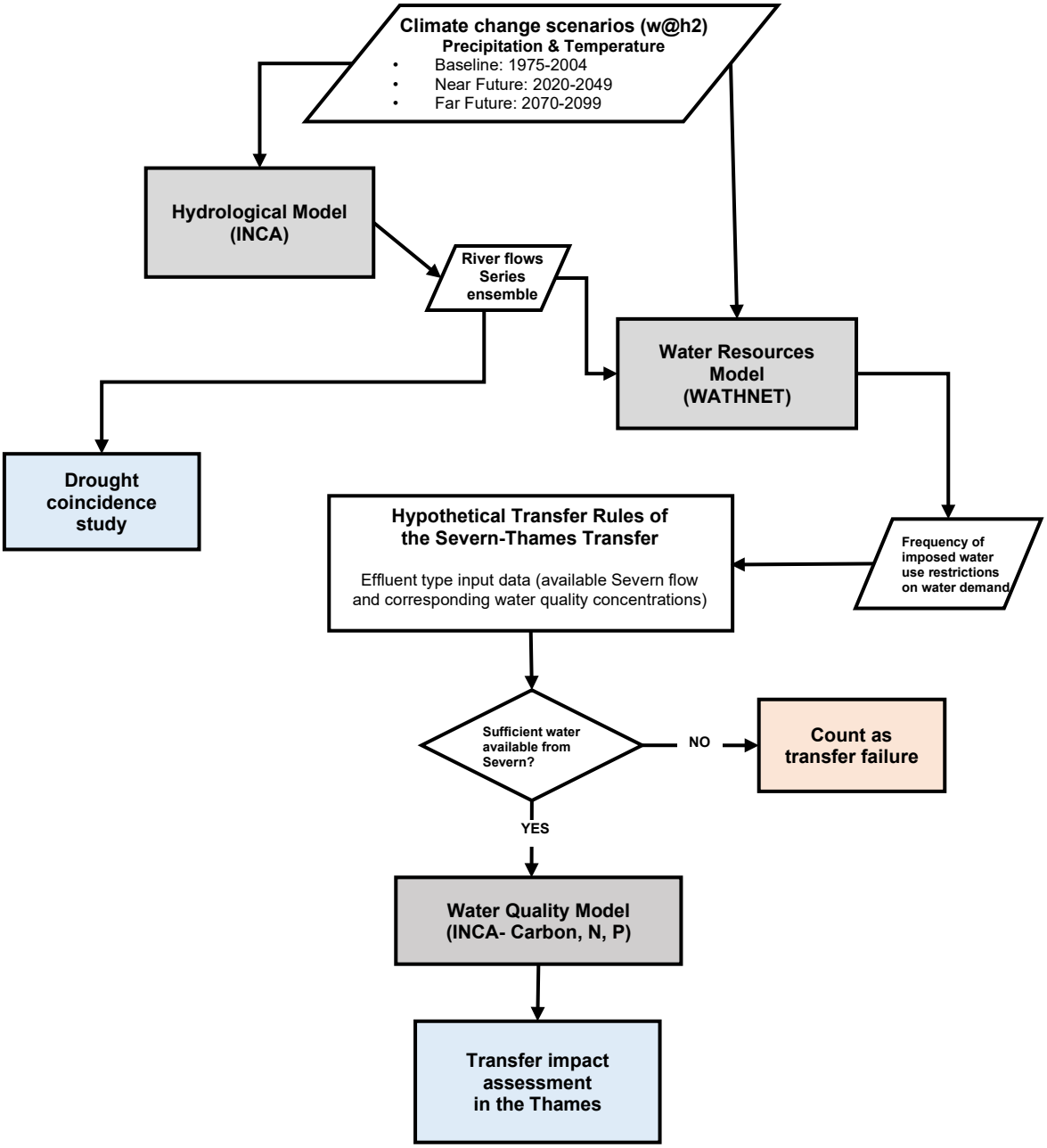


Figure 6.2: Flowchart of the combined modelling framework used in this analysis

6.2.2 Hydrological droughts and coincidence of droughts

Drought takes various forms depending on which part of the hydrological cycle it impacts most strongly. Hydrological drought is associated with the effects of a period of precipitation deficiency (meteorological drought) on surface or subsurface water supply and is usually followed by meteorological and soil moisture droughts (Leng et al., 2015; Van Loon, 2015; Van Loon and Van Lanen, 2012). The characteristics of hydrological droughts, such as frequency and intensity, are often defined on a water basin scale.

Investigating the impacts of hydrological droughts in the two catchments is crucial for the case study. For example, investigating how often flows in the Thames are in drought but flows in the Severn are not would be an important consideration for the appraisal of the Severn-Thames Transfer option. On the other hand, if both catchments are increasing experiencing droughts at the same time, the Severn-Thames Transfer scheme would be discouraged. To assess the feasibility of Severn-Thames Transfer, a drought index, the Standardised Streamflow Index (SSI), was employed. SSI is computed by aggregating streamflow data for specified monthly time scales (e.g. 1, 3, 6, 12 or 24 months) before fitting the aggregated streamflow data to a probability distribution function to obtain the SSI value in each month. There is no popular probability distribution function fitted to SSI calculation, although a few probability functions, such as log-normal and Tweedie, have been applied in previous studies (Shukla and Wood, 2008, Barker et al., 2016). In this study, the 1-month SSI was obtained by fitting a log probability density function, following Shukla and Wood (2008). The analysis was carried out using simulated baseline and future flows of the River Thames (at Kingston, reach 22) and the River Severn (at Haw Bridge, reach 18) obtained through the INCA hydrological model and

w@h2 climate scenarios (Figure 6.1). A drought event was defined as a period of the time when the drought index (SSI) fell below -1.0, representing the drop of the streamflow by one standard deviation below average streamflow conditions. Three metrics were used to examine drought characteristics in the River Severn and River Thames: (1) frequency—the number of drought events over a defined period (30 years, as in the baseline, near future and far future scenarios of w@h2); (2) intensity—the average SSI value for a drought event; and (3) duration—the length of each drought event. The results will be shown in Section 6.3.1.

6.2.3 London water supply system and potential operation rules for the transfer

The water supply for London comes from many sources but is primarily abstracted from the River Thames, upstream of Kingston (Teddington weir). The abstracted water is stored in raw water reservoirs before being sent to water treatment systems and supply pipes. The raw water reservoirs in London provide storage for use in dry periods when restrictions on abstraction from the source are imposed. The amount of water that can be abstracted from the river depends on the percentage of total raw water storages in London. The minimum flows at the Teddington weir required to maintain a healthy aquatic environment are protected by the Lower Thames Operating Agreement (LTOA), which sets environmental flow requirements based on the storage level in London's reservoir. The Lower Thames Control Diagram (LTCD) is the operating tool within the LTOA, which provides the day-to-day rules to regulate abstraction from the River Thames at Kingston, subject to environmental flow requirements (Figure 6.3). As reservoir levels fall, environmental flow requirements are gradually reduced from 800 to 300 Ml/d, allowing for continued abstraction during a drought period. At the same time, water use

restrictions are imposed to aid demand reduction (Table 6.1) when reservoir levels drop below the thresholds defined in the LTCD.

The runoff values generated with the INCA hydrological model are used as inputs to the WATHNET model (Kuczera, 1992). A water resources model, WATHNET, has been set up to represent London's urban water supply system. WATHNET was selected because of (1) its efficient computation time, (2) the flexible scripting feature, which enables to introduce any rules and constraints, and (3) its capacity of multi-objective optimisation and handling optionality. The model has been successfully applied in many water resources planning application (Mortazavi-Naeini et al., 2014; 2015). In the model simulation, a water shortage occurs every time the reservoir level falls below the reservoir level thresholds defined in the LTCD. When the reservoir level thresholds are breached, demand restrictions and reductions are imposed. Demand restrictions are lifted when the storage levels have recovered to above the threshold levels. For each model run, daily storage level, level of services and supply shortages were recorded. Household water demand was considered to be constant through the year and across all simulations, and taken from population and per capita consumption data of 2014, provided by Thames Water (Thames Water, 2014).

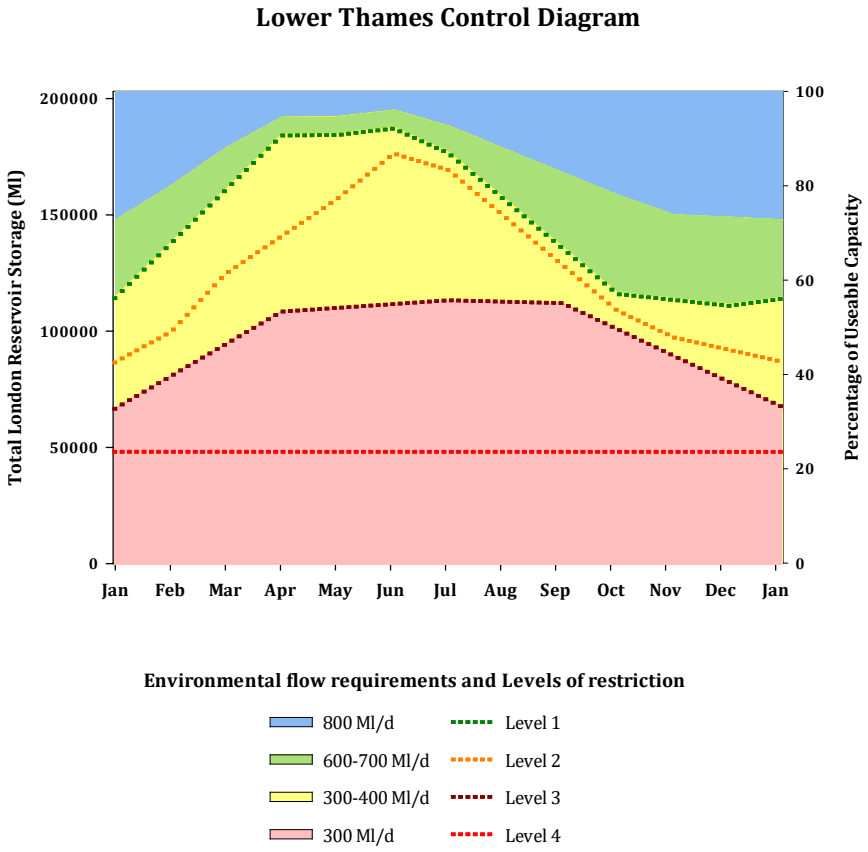


Figure 6.3: Lower Thames Control Diagram with storage control curves, levels of restriction and environmental flow requirements (redrawn from Thames Water, 2014)

Table 6.1: Four levels of water use restrictions in London (Thames Water, 2018)

Target Levels of Service (LoS)	Frequency of occurrence	Water use restrictions	Expected demand reduction (cumulative)
Level 1	1 year in 5 on average	Intensive water saving media campaign	2.2%
Level 2	1 year in 10 on average	Sprinkler/unattended hosepipe ban, enhanced media campaign	9.1%
Level 3	1 year in 20 on average	Temporary Use Ban (formerly hosepipe ban), an Ordinary Drought Order	13.3%
Level 4	Never (1 in 100 years)	Extreme restrictions such as standpipes and rota cuts in supply, an Emergency Drought Order	31.3%

As explained, the timing of the imposition of restrictions is dependent on reservoir level. The storage value on the LTCD determines the associated level of restriction on the month (or the day). Thus, the LTCD diagram can be a useful metric for deciding when the transfer will be triggered. Thus, reservoir drawdown periods are used to assess and decide the timing of water transfers, and we focus on the Level 3 restrictions on water use, as these have the most significant impact on water users and the economy. For each reservoir drawdown period, the first day when a reservoir level is below the threshold is recorded as an onset of a recession, and the last day of the reservoir drawdown period (when the reservoir level is above the threshold) is recorded as a termination. The duration of each reservoir drawdown period is also recorded, and any drawdown periods less than 10 days long are excluded from this analysis. Figure 6.4 shows the frequency of onset and termination months of reservoir drawdown with Level 3 thresholds.

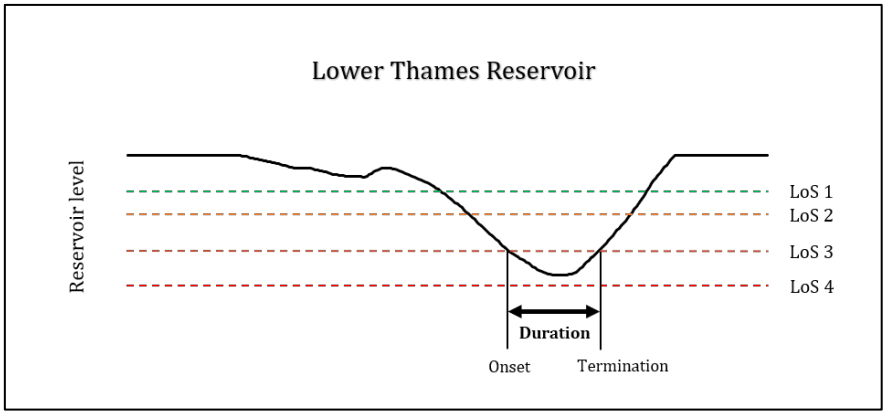


Figure 6.4: Reservoir drawdown period definition parameters (example Level of Service (LoS) 3 threshold)

6.3 Results and Discussions

6.3.1 Hydrological droughts and coincident droughts

Time series of the drought indices were calculated over the Thames and Severn catchments using projected future flows. Three different parameters: duration, intensity and frequency, were used to examine the hydrological drought characteristics. As the w@h2 climate dataset consists of 100 realisations in each time slice, the mean drought characteristics of a 30-year window, and the most severe drought event (by intensity) in each ensemble member, are used for discussion in this section. The changes in drought characteristics for the near future and the far future periods with respect to the current period (baseline) were compared to describe the trends of hydrological droughts in the future.

Both catchments will experience increased hydrological drought duration and frequency, and, to a lesser extent, increased hydrological drought intensity. Figure 6.5 shows the changes in drought frequency, mean duration, and mean intensity for two future projections. Each dot represents a realisation of w@h2 ensemble member, which indicate the mean value for the 30-year period. The size of the dots is scaled by the frequency of drought, that is, the occurrence over the 30-year period. Larger dots indicate more frequent events with corresponding drought intensity and duration. The error bar is for the one standard deviation to show the model ranges. In general, the Severn catchment (Figure 6.5(b)) shows greater changes than the Thames catchment (Figure 6.5(a)) in the three drought parameters. The two catchments will experience a gradual increase in drought frequency and duration, and the increase in drought frequency is projected to be more notable than drought duration. Changes in mean drought intensity are projected to

increase to a minor degree (up to 2%) in the Thames, while the Severn shows a slightly bigger shift towards an increase in the severity of droughts by up to 4% and 9%, in the near future and far future, respectively.

The changes in hydrological drought frequency generally reflect changes in precipitation patterns. As similar results are shown in the future flow projections (**Chapter 4**), higher temperatures could also further contribute to drying in summer and autumn in the two catchments areas, leading to the large increase in hydrological drought frequency in both areas. Figure 6.6 clearly shows the changes in the frequency of hydrological droughts for the two future scenarios, with an increase of 100% in the Thames and of 157% in the Severn in the far future.

The spatially concurrent hydrological droughts in the two catchments are analysed with the calculated SSI values, and are shown in Figure 6.5(c) and Figure 6.6(c). More frequent coincident droughts are expected with the longer durations in the two catchments for the two future projections, although no significant trend is observed in mean drought severity. In the far future period, the coincident drought would happen almost every year with on average three months of duration (Figure 6.6(c)). This would make the Severn-Thames Transfer option less sustainable. It implies that the Severn might not be able to supply water to the Thames, as both catchments would experience the hydrological drought at the same time.

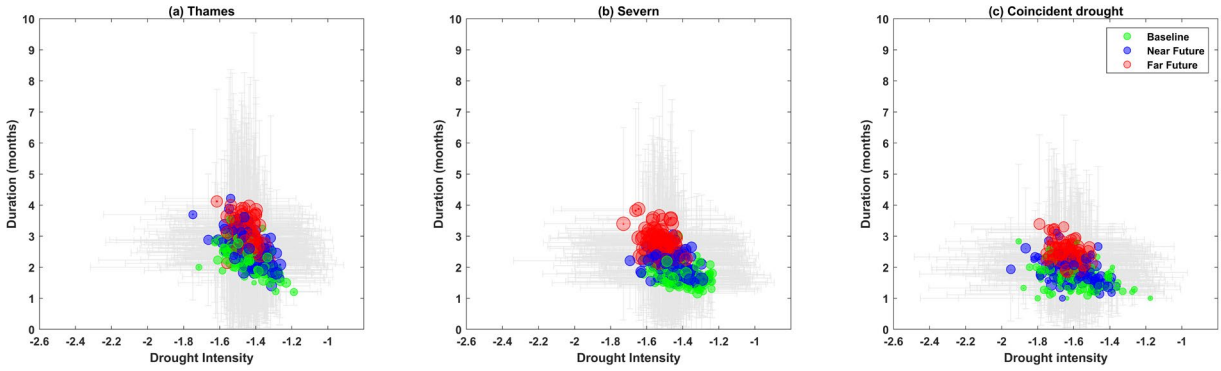


Figure 6.5: Changes in mean hydrological drought characteristics in the River Thames, the River Severn and the coincident drought in two catchments. The size of dots represents the frequency of drought, so larger dots indicate more frequent drought events.

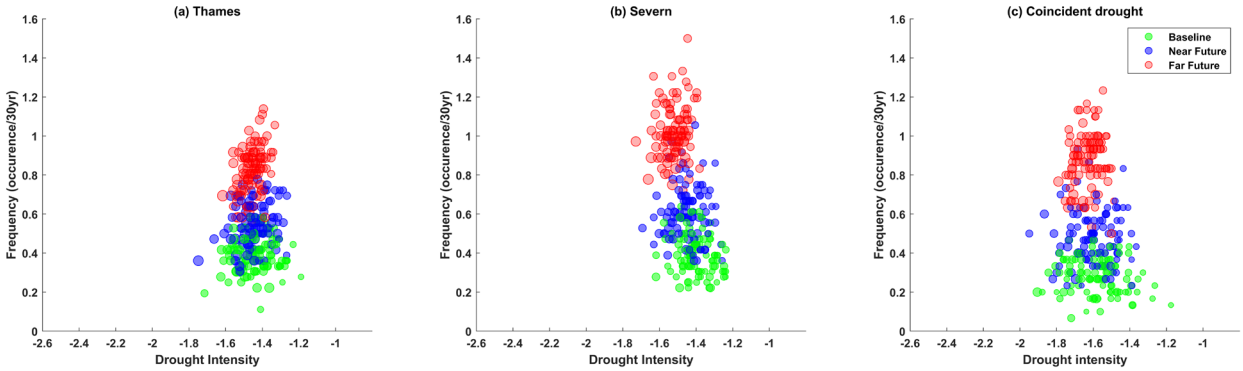


Figure 6.6: Similar plot to Figure 6.5, but changes in y-axis to frequency of drought and the size of dots indicate duration of drought.

The extreme hydrological droughts, especially those with high intensity and/or extended duration, are generally of most concern in drought management. For the extreme drought events, 100 droughts from each time slice were extracted to compare how their characteristics would change in the two future scenarios compared with the baseline condition. The intensity of drought was chosen as the criteria to select a representative drought, as the water resources system in London is more sensitive to drought deficit than drought duration in general (Borgomeo et al., 2015). Figure 6.7 shows the range of hydrological drought characteristics of the selected extreme drought events. The lines in the plot are the maximum and minimum values of drought parameters, and colours in dots

indicate different time slices (green: baseline; blue: near future; and red: far future). Black dots indicate a recent historical drought in the UK (the 1976 drought, within the baseline time period) and it has been marked for comparison with the magnitude of historical observation.

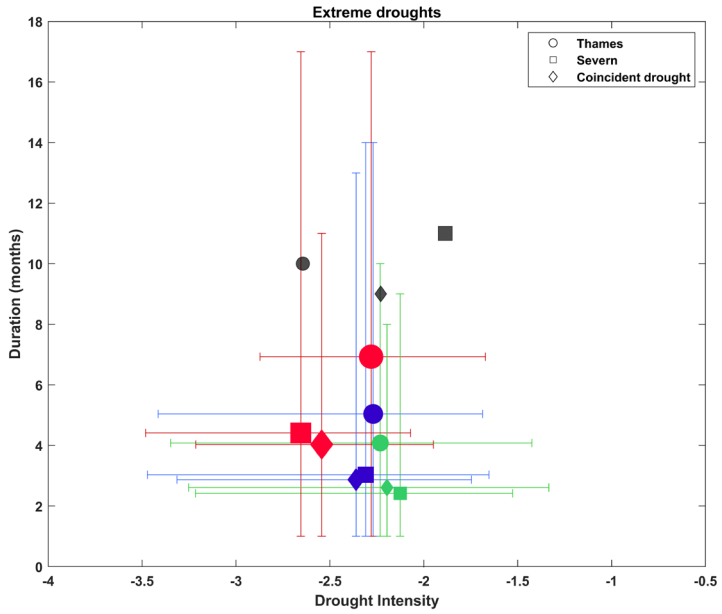


Figure 6.7: The range of hydrological drought characteristics of the extreme drought events. (green: baseline; blue: near future; red: far future; and black: historical drought in 1976 in the UK)

The most severe hydrological droughts are projected to increase in the two catchments with different characteristics. The Severn will experience more intense droughts, whereas the Thames is expected to experience longer droughts. The mean intensity of the droughts on the Severn shows a big shift towards to the lower SSI value (i.e., the most negative drought intensity represents the most severe event), whereas droughts on the Thames change little. The most severe event on the Thames shows the extended durations for the future scenarios, which is up to 70% longer in the far future than baseline. The most

severe coincident droughts are also expected to be more frequent and intense, as well as of longer duration. By the 2050s, extreme coincident drought would happen once every two years, with an average intensity of -2.4, which is categorised as extreme drought based on the classification of McKee et al. (1993).

The number of droughts is counted in each ensemble model simulation when the Thames experiences hydrological drought while the Severn does not. For those drought events, the Severn-Thames Transfer option is potentially useful, as surplus water could be transferred to the Thames. Figure 6.8 shows the distribution of the number of droughts in each ensemble member. The y-axis is the count of droughts, out of 100. The mean number of those specific droughts is projected to increase in the future. The mean number of droughts for the baseline period is 5.4, increasing to 7.8 for the near future and 10.8 for the far future. Even though the more severe droughts will occur on the Severn in the future, there still will be room to encourage the Severn-Thames Transfer scheme.

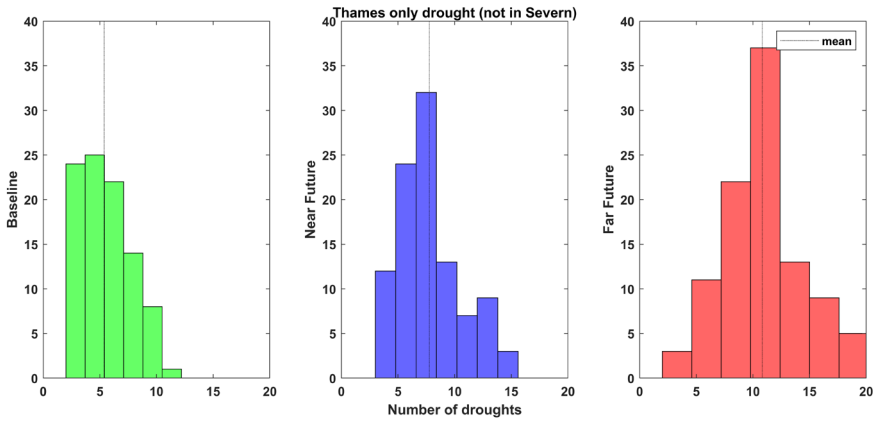


Figure 6.8: The number of occasions (y-axis) when the Thames is in drought while the Severn is not (green: baseline, blue: near future, red: far future)

6.3.2 Potential water transfer operation rules

6.3.2.1 Supply shortages in London, and Severn surplus flows

Monthly supply shortages in London were estimated, to identify the magnitude of shortages in each month and to estimate the demand on the possible Severn-Thames Transfer option. The Thames water resources system model was built in WATHNET, which was run on a daily time step for the study periods (baseline, near future and far future) with the ensemble of river flow time series generated by the INCA hydrology model. Monthly total supply shortages (shortfalls) for three 30-year periods (baseline, near future and far future) are analysed with the WATHNET model output and shown in Figure 6.9.

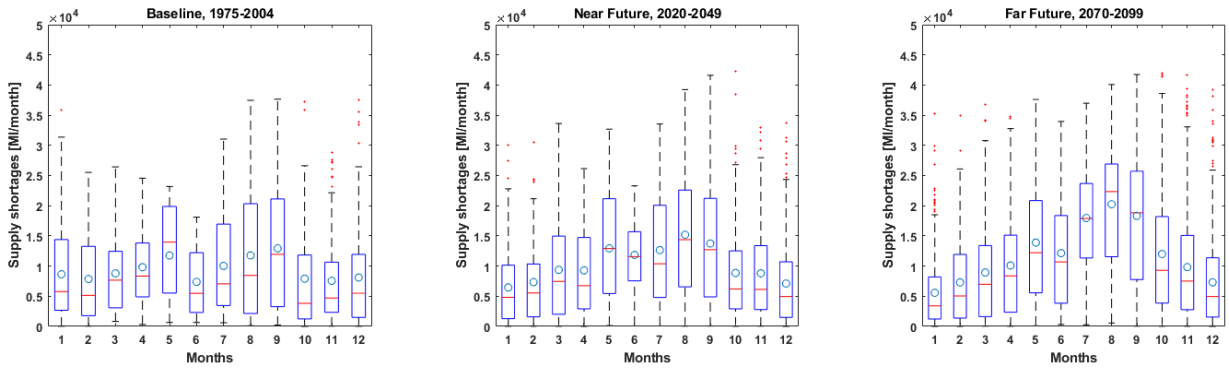


Figure 6.9: Projected monthly supply shortages in London

The average shortages per month vary from 7,000 to 12,000 Ml (megalitres) in the baseline, from 6,000 to 15,000 Ml in near future, and from 5,500 to 20,000 Ml in the far future. September and August have always been ranked top, having the highest shortages for all three time series. For the two future scenarios, the period of time May to September is ranked as the top five months with big supply shortages. Total shortfalls in summer

months (May to October) are projected to increase in the future up to 78% in July, whereas supply in winter months (December to February) are projected to decrease over time up to 35% in January. More than 60% of increases in shortages are projected in June, July and August in the far future scenario, compared with the baseline scenario.

As the unsupported transfer option (no storage option available) largely relies on surplus water availability in the Severn when the transfer scheme is triggered, an estimation of the projected yield of surplus flows is necessary. The environment flow requirements for the Severn, which is called 'Hands off Flow' (HoF) (Environment Agency, 2013b), must be taken into account as well. Thus, the HoF constraint at Deerhurst[3] on the Severn (a potential abstraction point for the STT) was applied to the calculation of surplus flows in the Severn. Figure 6.10 shows the monthly surplus flows on the River Severn with the strict environmental flow condition applied (HoF2).

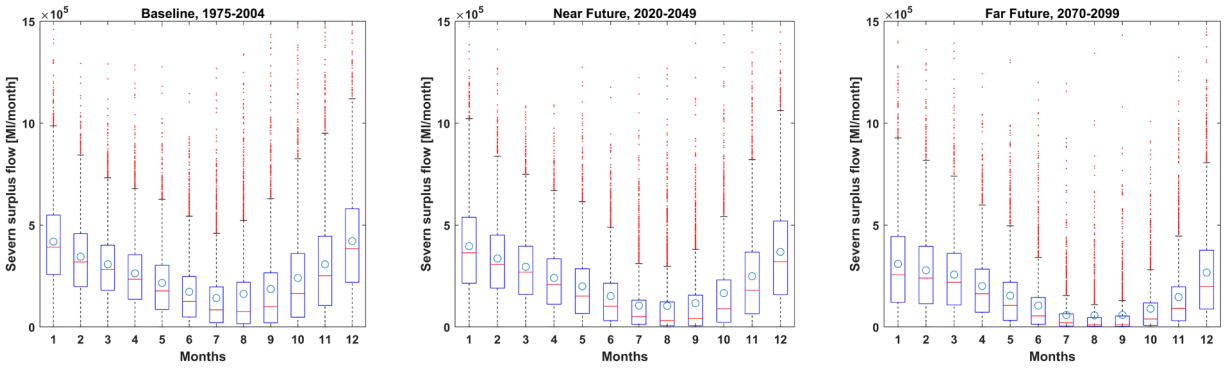


Figure 6.10: Projected Severn surplus flows at HoF2 conditions

[3] There are two HoF conditions at the Deerhurst in Severn, HoF1 and HoF2: (1) HoF1 is at 1800 Ml/d when abstraction would be required to cease; and (2) HoF2 is at 2490 Ml/d when any abstraction would be limited to 240 Ml/d (Environment Agency, 2013).

In general, the River Severn has sufficient surplus flows in all months to make a transfer possible. The availability varies across the year, and the potential yield would vary according to the drought event that triggered the Severn-Thames Transfer option. Although this study only considers the unsupported transfer option, strategic storage for surplus water during non-drought periods might be required to maximise the benefit of the Severn-Thames Transfer.

6.3.2.2 Suggested operation rules and expected performance of the transfer

In this case study, Level 3 (as defined in section 6.2.3) was defined as a threshold to trigger the Severn-Thames Transfer. This volume-based threshold approach would work on the same principles as the current existing strategic scheme and the progressive implementation of demand restrictions in London. This would mean that it is relatively unlikely that large volumes of transfer would be requested in spring or winter.

The timing and frequency of the potential transfers are estimated. The demand for the water transfer is projected to increase in the future. Water transfers are requested six times in the baseline (30 years) on average. The demand is likely to be doubled in the near future, and even tripled in the far future on average. Figure 6.11 shows the changes in the number of transfers requested by the Thames in the two future scenarios.

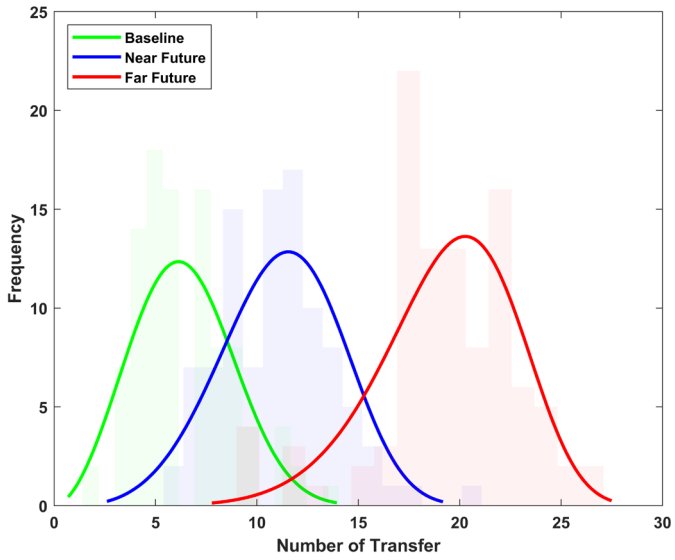


Figure 6.11: The changes in number of transfers requested by the Thames over the projected climate scenarios

The seasonal patterns in the onset and termination of transfer are also investigated. Figure 6.12 shows the frequency of onset and termination months of reservoir drawdown with Level 3 thresholds. In the baseline, around 30% of transfers are triggered in September, which ranked at the top month of onset. The frequency of September triggers increases in near and far future scenarios. The trends of monthly water shortage are reflected in the seasonal pattern, and the frequency of onset is estimated to increase by 15% in August in the far future. Reservoir levels are mostly recovered over winter (November and December). The trends of termination months are consistent over future time series. Average duration of each transfer case was 87 consecutive days on average in the baseline, increasing to 101 consecutive days for the near future and 125 consecutive days for the far future.

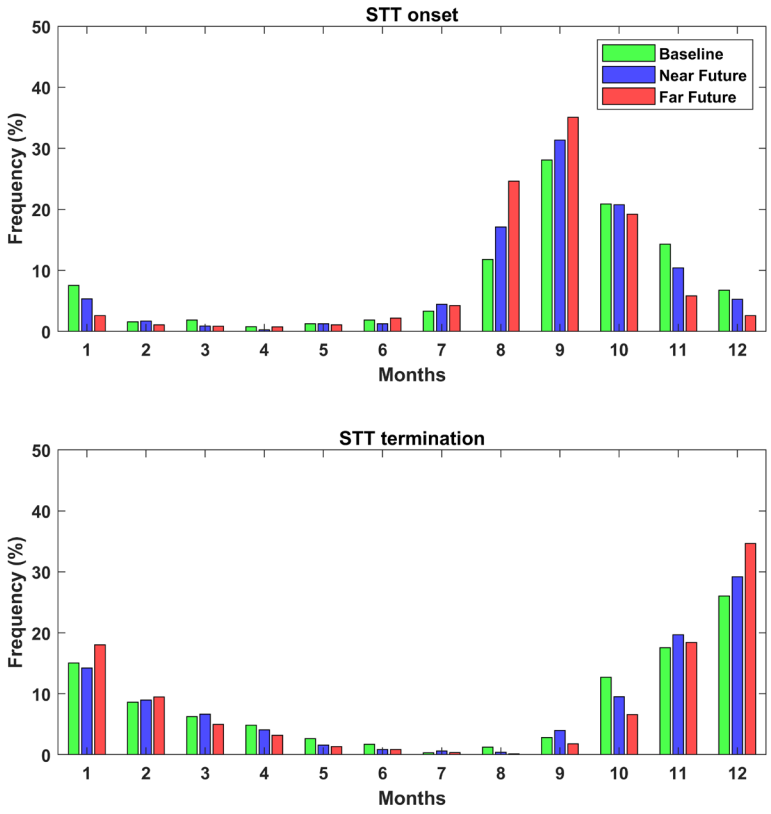


Figure 6.12: Frequency (%) of the Severn-Thames Transfer onset and termination months with LoS 3 threshold

Hypothetical operation rules are suggested based on the model results. Two operational rules for transfer volumes can be suggested based on the projected future scenarios: (1) Scenario A is to make a transfer of a requested amount from the Thames when a shortfall occurs, and (2) Scenario B is to make a transfer of a fixed amount (100 Ml/day) as soon as the reservoir levels fall below the Level 3 threshold level.

In other words, the window of transfer activation is decided by the different transfer trigger in scenario A and B, based on the degree of infrastructure failure. The Severn-Thames Transfer trigger will determine the exact onset location and the duration of a

transfer. Scenario B has a wider length of the window for transfer activation, as the trigger is the time when the reservoir level falls below Level 3, and a fixed amount will be supplied if available from the Severn. In the case of scenario A, the Severn-Thames Transfer trigger is the time when the shortfall occurs in the Thames. The water resources model results suggested that all occurrence of shortfalls was in Level 3 and Level 4 restrictions, but not necessarily all the time. Therefore, the length of the transfer window could be shorter than scenario B.

Table 6.2: The Severn-Thames Transfer performance estimation

		Scheme triggered (transfer success) days/30years	Success rate %	Demand (transferred volume) Ml/day	Success rate %
Scenario A: Requested amount	**Baseline (1975-2004)**	135 (68)	50	9277 (3525)	38
	Near Future (2020–2049)	296 (119)	40	12366 (3326)	27
	Far Future (2070–2099)	778 (228)	29	21936 (3612)	16
Scenario B: Fixed amount: 100 Ml/d	**Baseline (1975-2004)**	560 (302)	54	8729 (4655)	53
	Near Future (2020–2049)	1142 (530)	46	10130 (4648)	46
	Far Future (2070–2099)	2434 (892)	37	12526 (4526)	36

Based on the two scenarios, the Severn-Thames Transfer performance is estimated in the future (Table 6.2). In the far future, 29% and 37% of the time the water from the Severn is available in order to alleviate the shortages in the Thames, depending on the transfer scenarios. Scenario B supplies a greater amount of water in general with a higher success rate, as it has a longer transfer activation period with the earlier onset timing. In this case, the Severn-Thames Transfer can occur before both catchments experience a coincident drought. As discussed in the previous section, the Thames will experience more extended

and frequent drought. Thus, the number of days the transfer triggered, and transfer demand is expected to increase in the near future and far future in both scenarios. In the case of scenario A in the far future, the Severn-Thames Transfer is triggered 5.7 times more frequent than the baseline with the increased demand, which is 2.3 times more than the baseline. Scenario B has similar trends, showing that four times more frequent on the transfer onset with 40 % more than the baseline volumes in the far future. The rate of success is expected to cut off gradually in the far future compared to the baseline, although the absolute number of days in the Severn-Thames Transfer success are expected to increase. Interestingly, the absolute transferred volumes in both scenarios remain almost constant from the baseline to the far future as well, while the success rates are projected to decrease, which reflect the increase of demand.

6.3.3 Hypothetical transfer rules for water quality modelling

The hypothetical operation rules of the unsupported Severn-Thames Transfer were identified based on the current and future hydrological analysis of both catchments. Two different operation rules were suggested based on the transfer activation trigger and transfer volumes from Deerhurst (the River Severn) to a location in the River Thames (Figure 6.1). The suggested scenarios were: 1) Scenario A was introduced to make a transfer of a requested amount from the Thames when shortfalls occur, and 2) Scenario B was to make a transfer of a fixed amount (100 Ml/day) as soon as the reservoir levels in London fall below the Level of Service (LoS) 3 threshold. The 300 Ml/d and 500Ml/d transfers are also tested according to the recent Water Resource Management Plan 2019 (WRMP19) planning options report (Thames Water, 2020a). The additional capacities [300 Ml/d and 500 Ml/d] are following the operation rules of Scenario B. Therefore, this

study is designed to test four different scenarios to assess water quality impacts in the Thames after transferring raw waters. The options to be considered within this INCA modelling study are summarized in Table 6.3.

Table 6.3: The Severn - Thames Transfer schemes to be considered for INCA modelling

Scenario	Transfer capacity	Transfer trigger	Feasible WRMP19 option
A	Varies depends on the demand in Thames	Shortfalls in Thames WRZ	Not applicable
B	100 Ml/d	LoS 3 in London reservoirs	Yes
C	300 Ml/d	LoS 3 in London reservoirs	Yes
D	500 Ml/d	LoS 3 in London reservoirs	Yes

The modelling assessment of water transfer and impacts on water quality between the Severn and the Thames is designed following a quantitative approach for the purpose of the study. INCA models (carbon, nitrogen and phosphorus) for both catchments are available (**Chapter 4**), therefore water quality concentration time series from INCA models in Severn at specific abstraction point (Deerhurst) are taken as an effluent-type input (point-source) in the INCA models of the River Thames for each of model simulations (for w@h2 time series, the baseline, near future and far future timeline). The location of discharge of the suggested transfer is at Culham in the upper Thames according to the recent WRMP19 (Thames Water, 2020a). The exact locations are shown in Figure 6.1.

The transfer strategy here is hypothetical and experimental in design to incorporate into environmental modelling. Consequently, there are underlying assumptions as follow:

i. There is no time lag between the date of withdrawal in the Severn and the date of discharge in the Thames.
ii. No storage option is available in the receiving water body yet.
iii. Raw water transfer; no water treatment before discharging in the Thames.
iv. No water loss while transporting in pipes.
v. Natural catchment boundaries are applied for INCA modelling. Thus, there is a difference from Water Resources Zone (WRZ) in Severn (Trent is not included). However, the water demand of Thames is taken from the whole WRZ of Thames for the scenario development.
vi. The Thames will accept all transferred water from the Severn.

6.3.4 Water transfer scenario analysis

The following sections will discuss the potential impacts of the Severn-Thames Transfer on water quality variables (dissolved organic carbon (DOC), nitrate, total phosphorus, and suspended sediment) for future time slices. All water transfer scenarios (A-D) assume that abstractions on the River Severn at Deerhurst will occur, with the water being pumped through a pipeline directly to the River Thames to discharge at Culham (Figure 6.1). Results are assessed at two locations of the River Thames: 1) Culham, a location where the transferred water is being discharged directly to the river channel (mid-reach of the River Thames), and 2) Egham (near Runnymede), next to the reach of Datchet intake for water supply in London.

Firstly, the effect of water transfers on water quality in the Thames was analysed by computing diagonal pair comparison plots. The diagonal pair comparison plot provides simple and straightforward visual comparisons of water quality variables to give a quick overview of data. This is a scatterplot of the data pairs in an X-Y graph, where monthly paired water quality variables are deployed. In this case, water quality concentrations with no transfer option are X, and four transfer scenarios are Ys. The data point falls above, below or on the diagonal line, indicating degradation, improvement, or no change, respectively. Thus, if data points are dominated in the upper triangular areas of the plot, the water quality has been degraded by the proposed transfer options. Water quality improvement will be shown if data points occupy the lower triangular areas of the graph. The monthly percentage change in water quality under different transfer scenarios was quantified for the baseline and two future time series. 30-year average system responses are shown here.

Monthly average DOC, nitrate, total phosphorus, and suspended sediment concentrations of INCA water quality model results are used to assess the impact of four different scenarios at two locations in the Thames mainstream. The results and discussion for each of the water quality variables will follow. In the figures, the same colour tones are used to illustrate each of the transfer scenarios (Scenario A: red, Scenario B: blue, Scenario C: yellow, and Scenario D: green).

6.3.4.1 Dissolved organic carbon (DOC)

Figure 6.13 and Figure 6.14 show the diagonal pair comparison of DOC concentrations under four different transfer schemes, and Figure 6.15 and Figure 6.16 show the monthly average changes in DOC concentrations by different transfer schemes. At discharge points, water quality concentrations are scattered close to the diagonal lines, which indicate the expected impacts are not significant. However, the degree of impacts is gradually larger towards the near and far future, compared to the baseline. Larger volume transfers (Scenario C: 300 Ml/d, Scenario D: 500 Ml/d) presents the greater impacts and the maximum increase in DOC concentrations in Scenario D (500Ml/d transfer) was found to be a 112% increase in the far future (Figure 6.13 (c)). Improvement of DOC concentrations are expected in winter, and degradation is expected in summer. It can be confirmed by the results of percentage changes in months in Figure 6.15, where the percentage change with negative values means improvement of water quality, and percentage change with positive values means the degradation of water quality due to the transferred water.

As DOC concentrations in summer months at lower Severn is continuously higher than those in upper Thames (**Chapter 4, section 4.3.5.1**), a transfer is being responsible for the summer months when the transfer is made low flows in the Thames. While transferred water flows along with the channel, the dilution capacity is enough to minimise and have the improvement on DOC concentrations at the lower Thames (Figure 6.14).

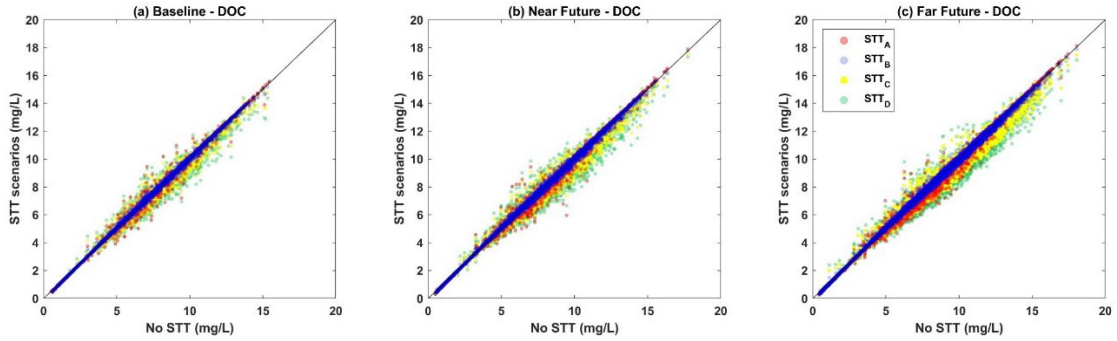


Figure 6.13: The results of transfer schemes: comparison of DOC concentrations at the discharge point (Culham, R9 of modelling setup) (hereafter, 'STT' indicates the Severn -Thames Transfer and 'No STT' means no implementation of transfers)

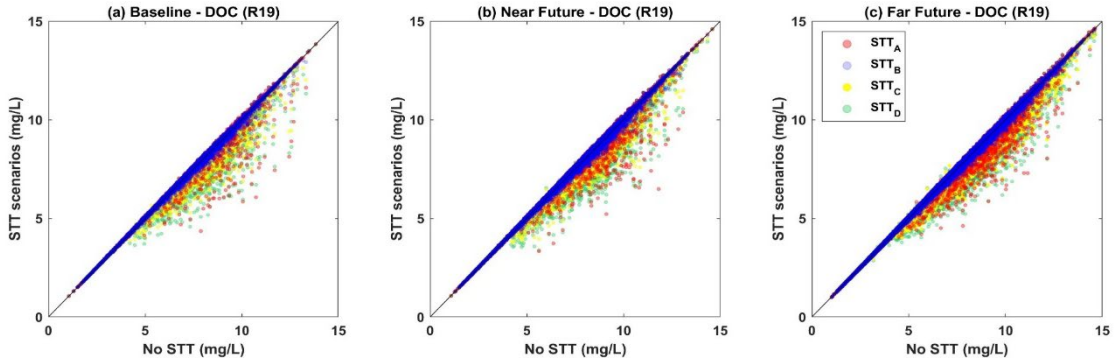


Figure 6.14: The results of transfer schemes: comparison of DOC concentrations at the lower Thames (Egham, R19 of modelling setup)

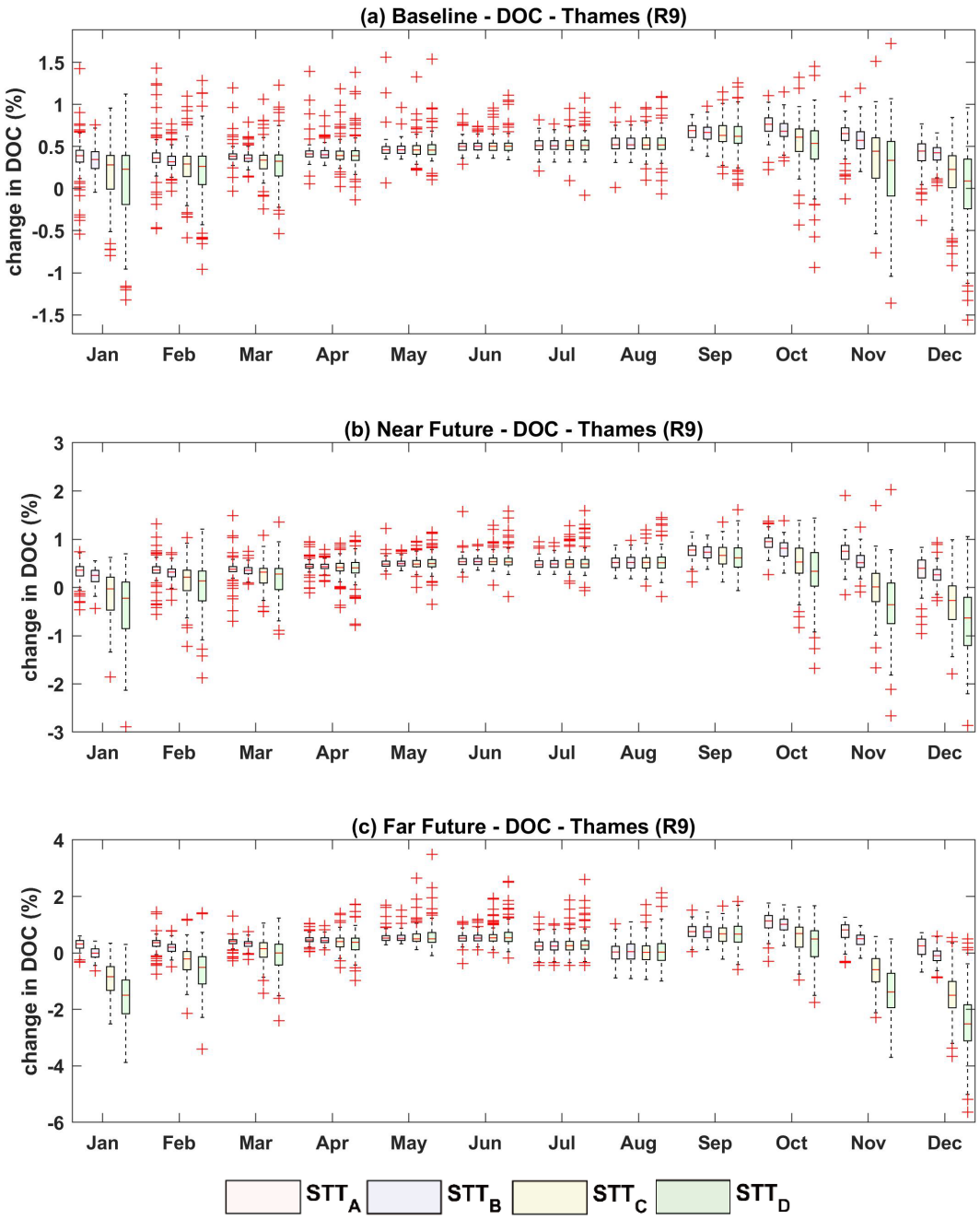


Figure 6.15: Monthly percentage change in DOC concentrations under different transfer scenarios at the discharge point (Culham, R9)

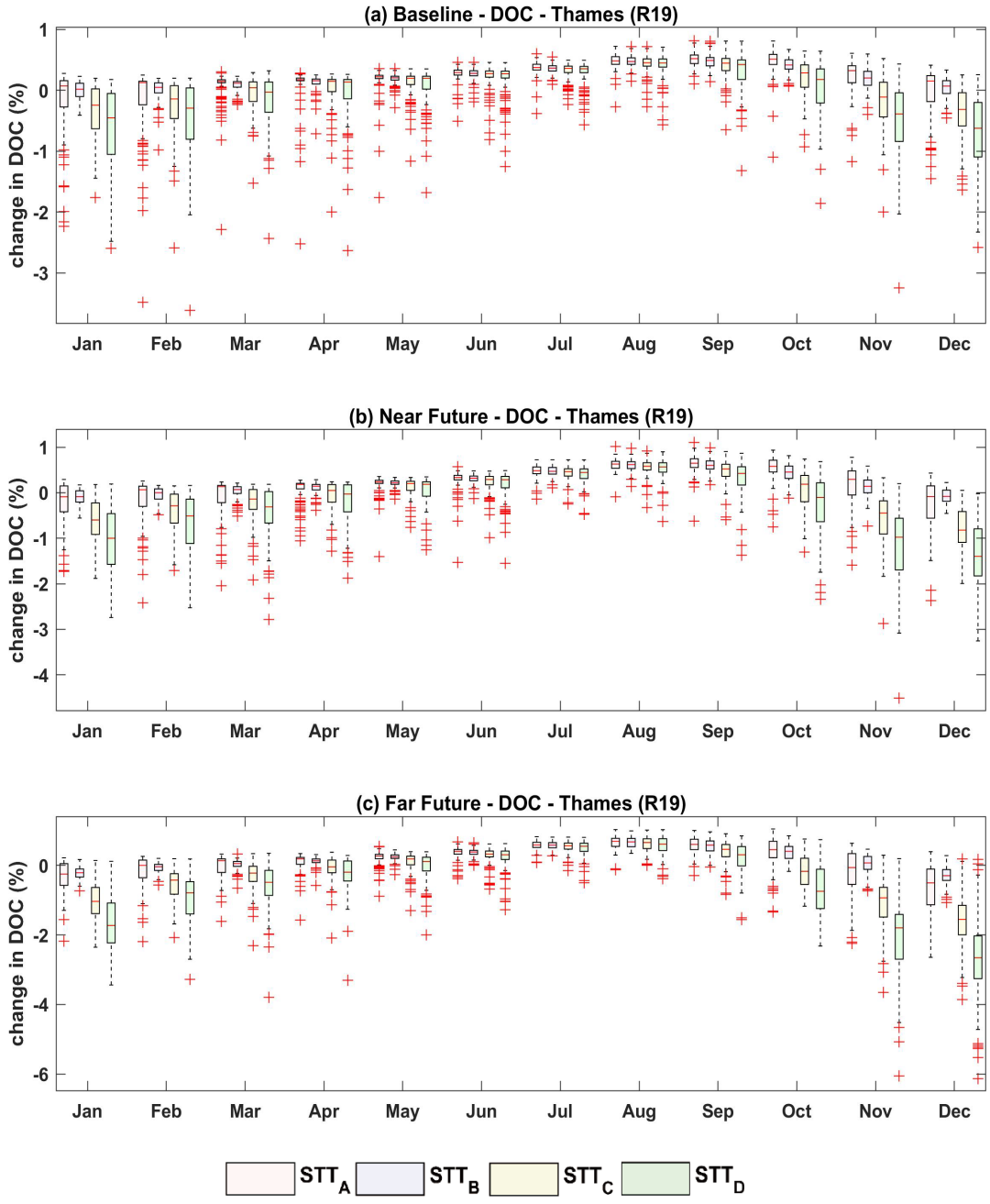


Figure 6.16: Monthly percentage change in DOC concentrations under different transfer scenarios at the lower Thames (Egham, R19)

6.3.4.2 Nitrogen

Nitrogen concentration is improved for all time slices, all transfer schemes, and all locations in the Thames. This is because the nitrogen concentration in the lower Severn is much lower compared to the Thames (**Chapter 4**). Thus, positive impacts are expected. Figure 6.17 and Figure 6.18 show the visual improvement on the nitrate concentrations at two locations of Thames. The improvement at the discharge points is greater, and those impacts are diluted along with river flow. A little improvement in water quality is observed at the lower Thames. Greater improvement of nitrate concentrations is expected in the autumn and winter months. This can be linked to the frequency of onset of transfers that around 30 % of transfers are triggered in September, and the transfer is continued throughout the winter months (refer to Section 6.3.2.2, Figure 6.12). From the Scenario D (500 Ml/d) at far future, up to 5 % of improvement in nitrate concentrations by the results of the monthly percentage change (Figure 6.19 (c)) and those impacts are reduced to around 3 % at the lower reach (Figure 6.20 (c)).

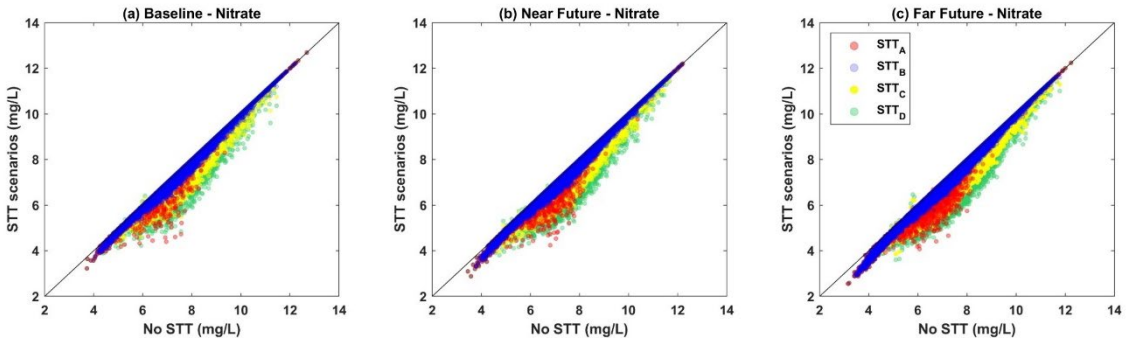


Figure 6.17: The results of transfer schemes: comparison of nitrate concentrations at the discharge point (Culham, R9)

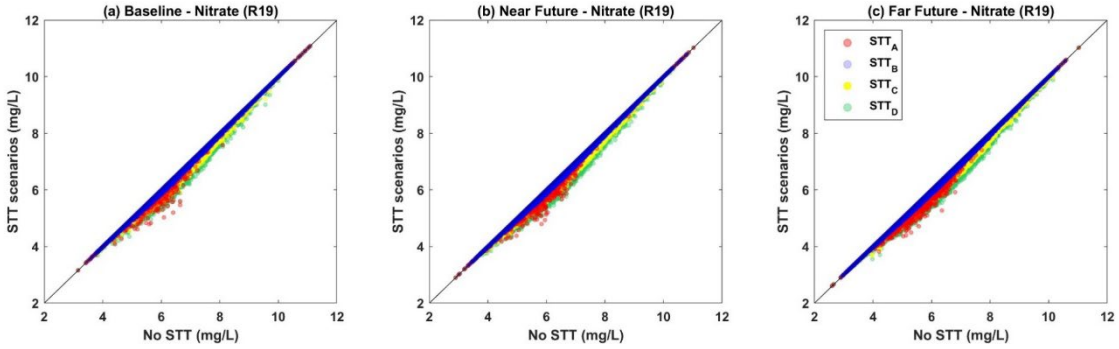


Figure 6.18: The results of transfer schemes: comparison of nitrate concentrations at the lower Thames (Egham, R19)

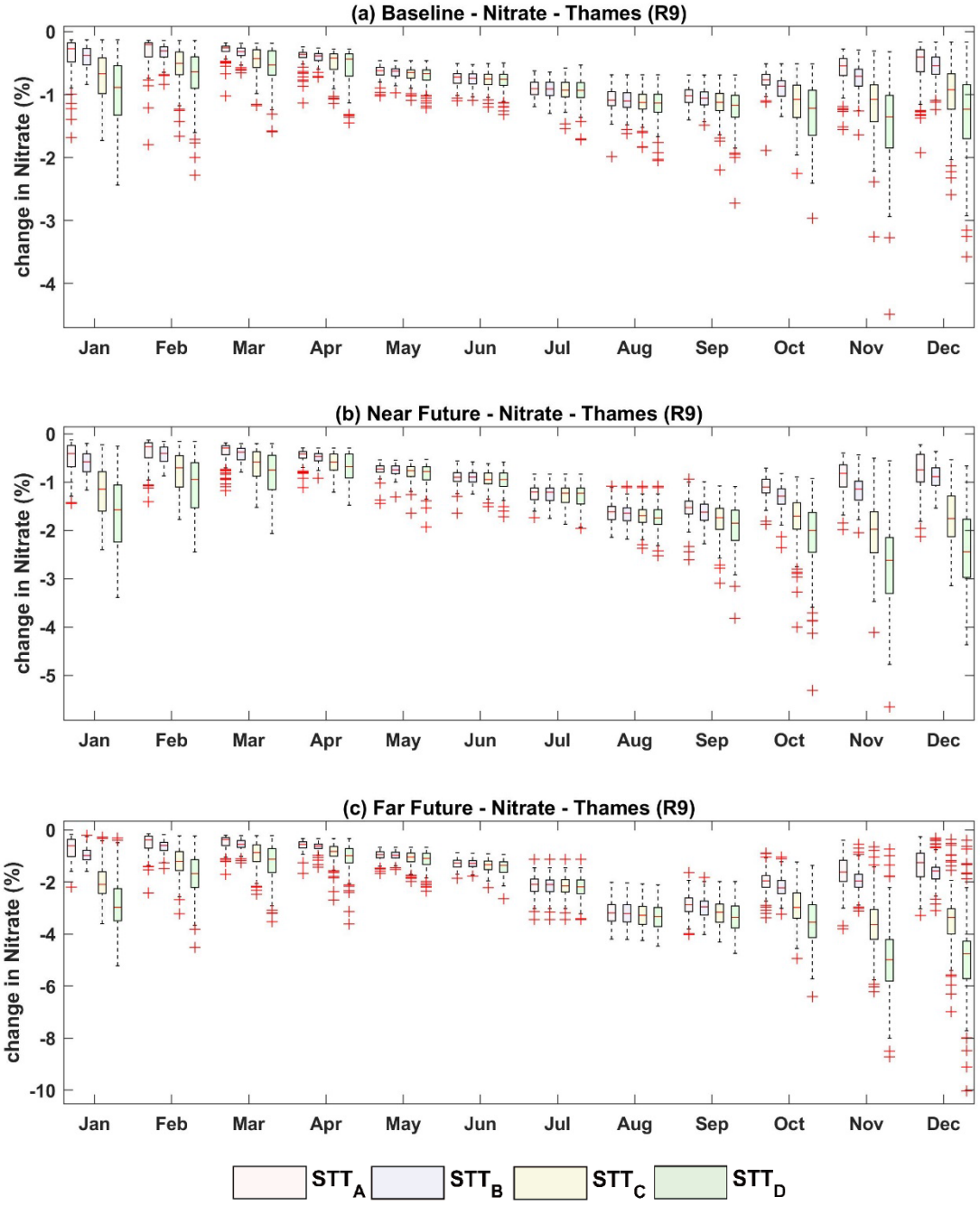


Figure 6.19: Monthly percentage change in Nitrate concentrations under different transfer scenarios at the discharge point (Culham, R9)

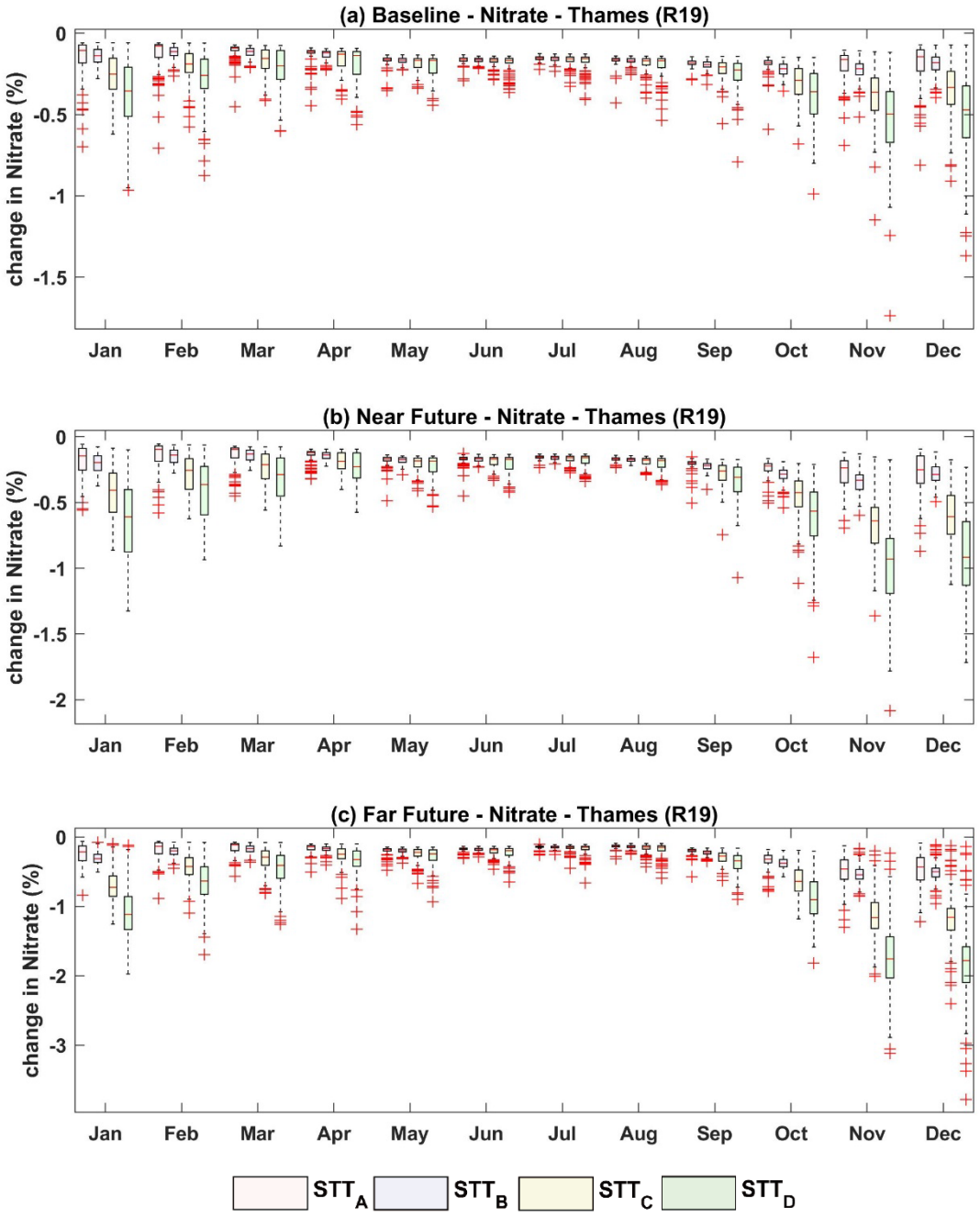


Figure 6.20: Monthly percentage change in Nitrate concentrations under different transfer scenarios at the lower Thames (Egham, R19)

6.3.4.3 Phosphorus and suspended sediment

The Severn-Thames Transfer shows the most significant impacts on total phosphorus concentrations than other water quality variables, and the degree of impacts are significant in the Thames. Total phosphorus concentrations will have a considerable increase, and larger volume transfers present greater impacts. The diagonal pair compares total phosphorus concentrations under four different transfer schemes (Figure 6.21 and Figure 6.22), and the seasonal changes in total phosphorus concentrations by different transfer schemes are shown in Figure 6.23 and Figure 6.24.

The pairs of water quality concentrations are occupied in the upper triangular area, which indicates the degradation of water quality. Figure 6.21 shows significant pollution spikes at the discharge location. As the annual average total phosphorus concentration is higher in lower Severn compared to the upper Thames (**Chapter 4, section 4.3.5.3**), some level of transfer responsibility was expected. However, the degree of the increase can be much more significant due to the timing of transfer and its volume. The level of pollution is gradually reduced along with the river channel, but still, the transfer is being responsible for the increased phosphorus concentrations at Datchet intake for water supply in London (Figure 6.22). The timing and trigger of the transfer become an important factor to be considered in this case. The difference between Scenario A and B was the transfer trigger. Scenario A was triggered when the London reservoir fails to supply water to the public (shortfalls), but Scenario B was trigged when the level of service reached 3 (LoS3) when demand restriction is being imposed on the public. In other words, Scenario A is transferring water most likely when the Thames experience more severe drought conditions than Scenario B. Thus, a transfer is becoming a semi-permanent point source

of phosphorus during droughts. Hence, the results show greater impacts of Scenario A than B (Scenario A in red and Scenario B in blue in Figure 6.21 and Figure 6.22).

Figure 6.23 and Figure 6.24 shows the percentage changes in total phosphorus concentrations in each month at the discharge point. In previous **Chapter 4 (**Figure 4.12**),** climate change will increase summer phosphorus concentrations in summer, and minimal impacts in winter are expected. However, the percentage change (positive) in phosphorus concentrations is shown the reverse seasonal trends of the phosphorus concentrations compared to the water body, which will consequently alter the temporal dynamics phosphorus.

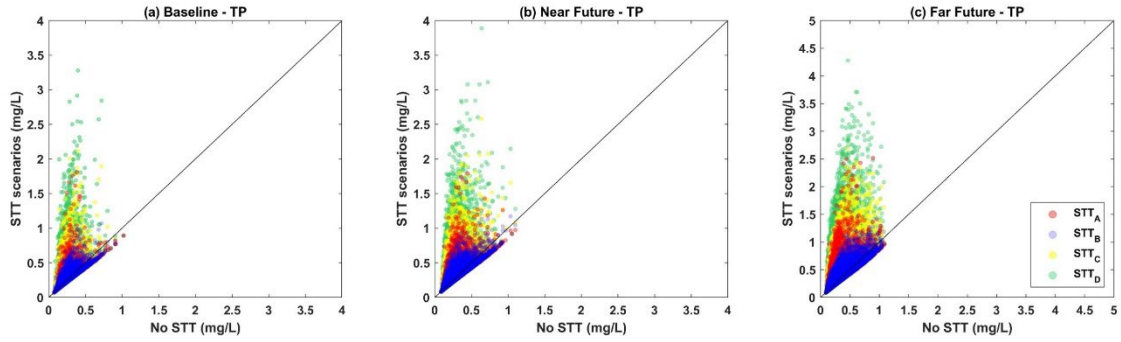


Figure 6.21: The results of transfer schemes: comparison of TP concentrations at the discharge point (Culham, R9)

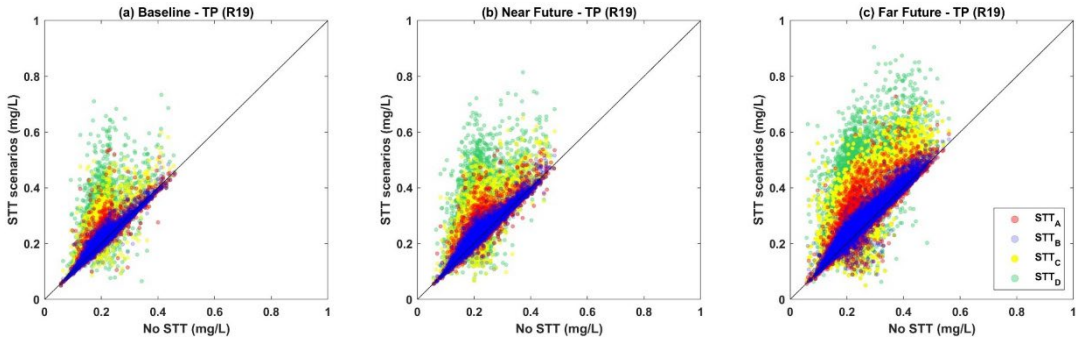


Figure 6.22: The results of transfer schemes: comparison of TP concentrations at the lower Thames (Egham, R19)

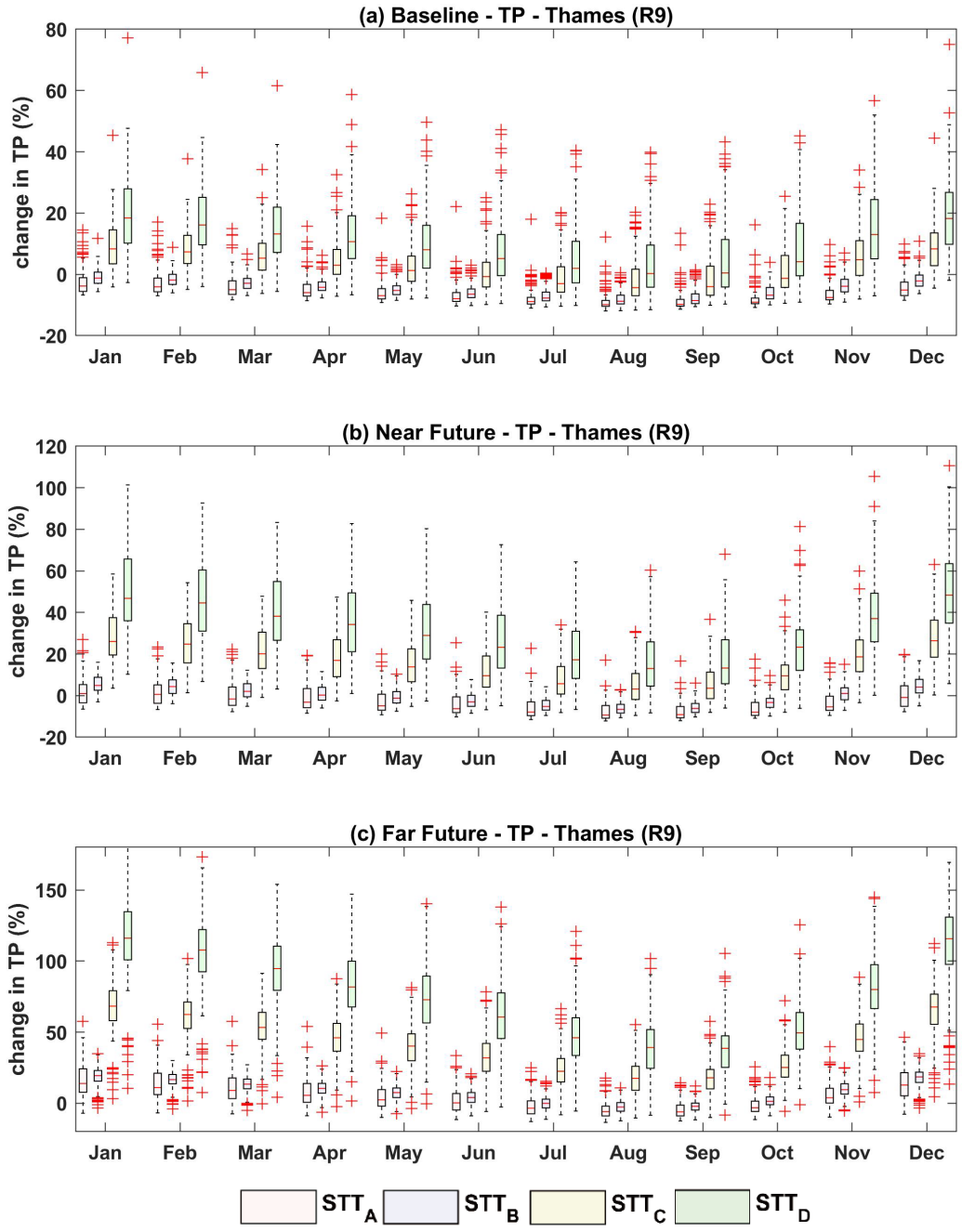


Figure 6.23: Monthly percentage change in TP concentrations under different transfer scenarios at the discharge point (Culham, R9)

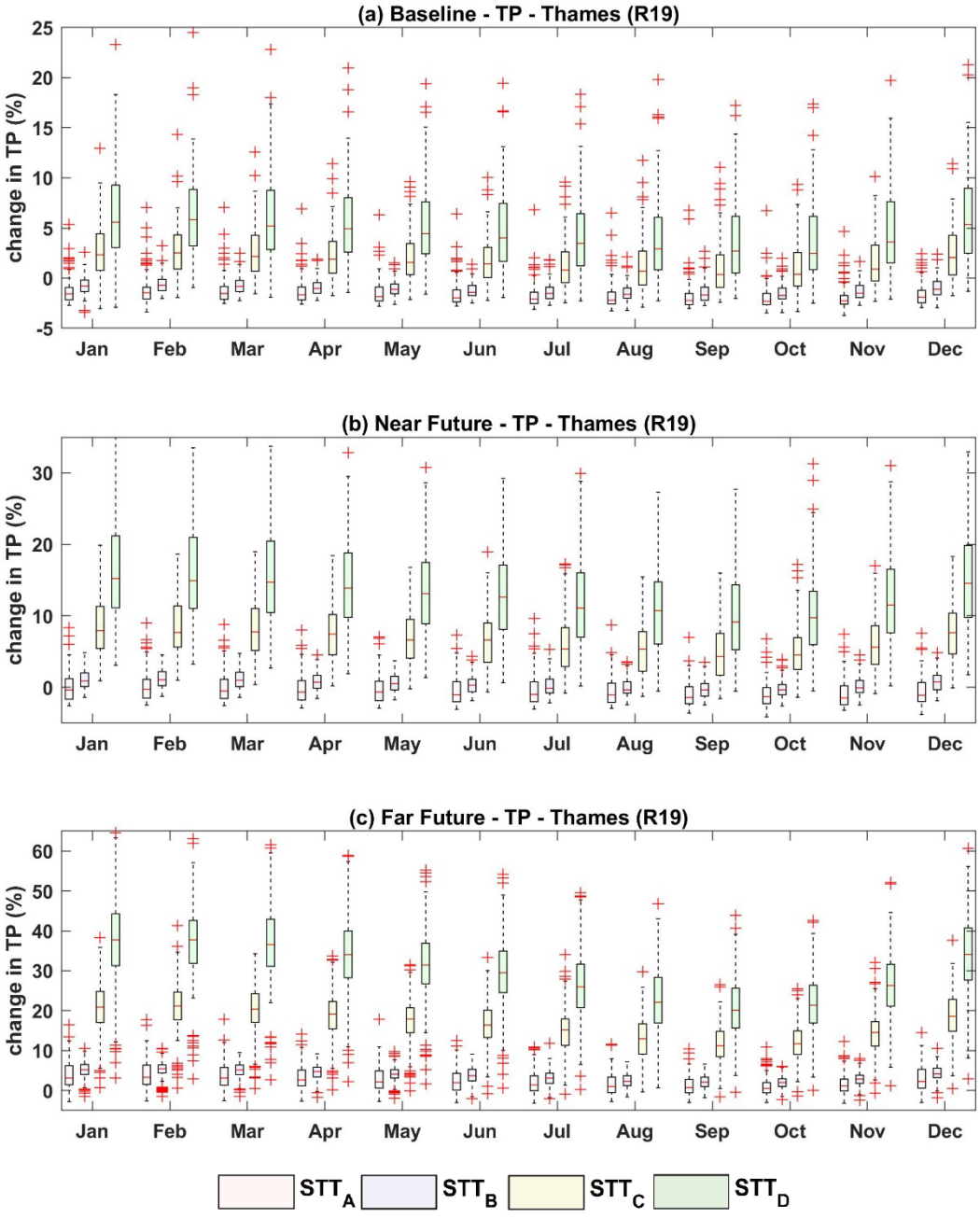


Figure 6.24:Monthly percentage change in TP concentrations under different transfer scenarios at the lower Thames (Egham, R19)

Figure 6.25 - Figure 6.26 show the diagonal pair comparison and Figure 6.27 - Figure 6.28 show the monthly average changes in suspended sediment concentrations by different transfer schemes. The minimal impacts are expected by transfers. The difference between suspended sediments and other water quality variables is that the impacts of transfers is increasing gradually along with the channel toward the lower reach (Figure 6.26). Although it is not clearly shown in the monthly changes (Figure 6.27 and Figure 6.28), the diagonal comparison plot displays the degree of changes from discharge point to lower Thames. Although minimal impacts are expected in suspended sediment concentrations, this variable is affected by the process of abstracting and discharging water in principle. Thus the opportunity for improvement in suspended sediments analysis exist here.

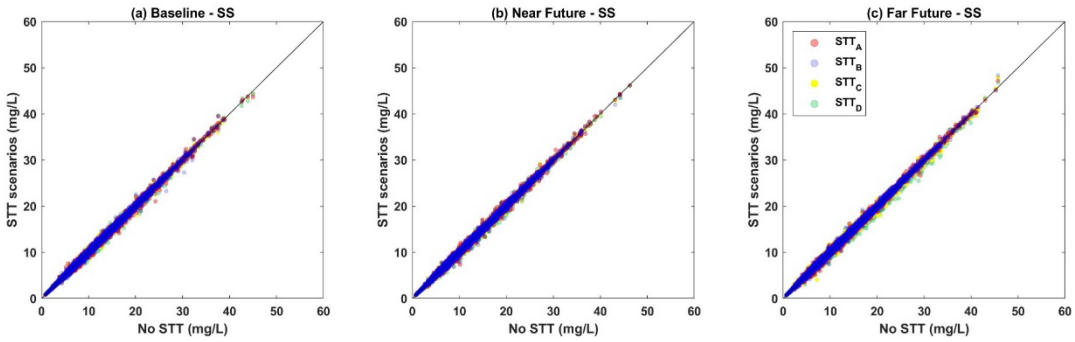


Figure 6.25: The results of transfer schemes: comparison of SS concentrations at the discharge point (Culham, R9)

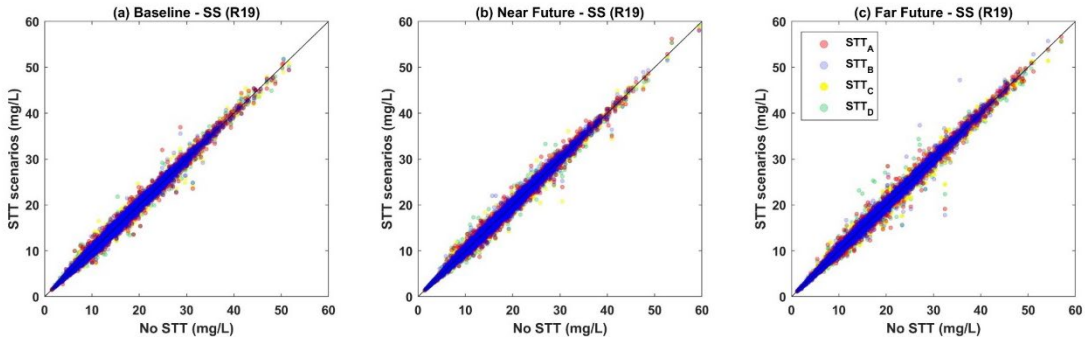


Figure 6.26: The results of transfer schemes: comparison of SS concentrations at the lower Thames (Egham, R19)

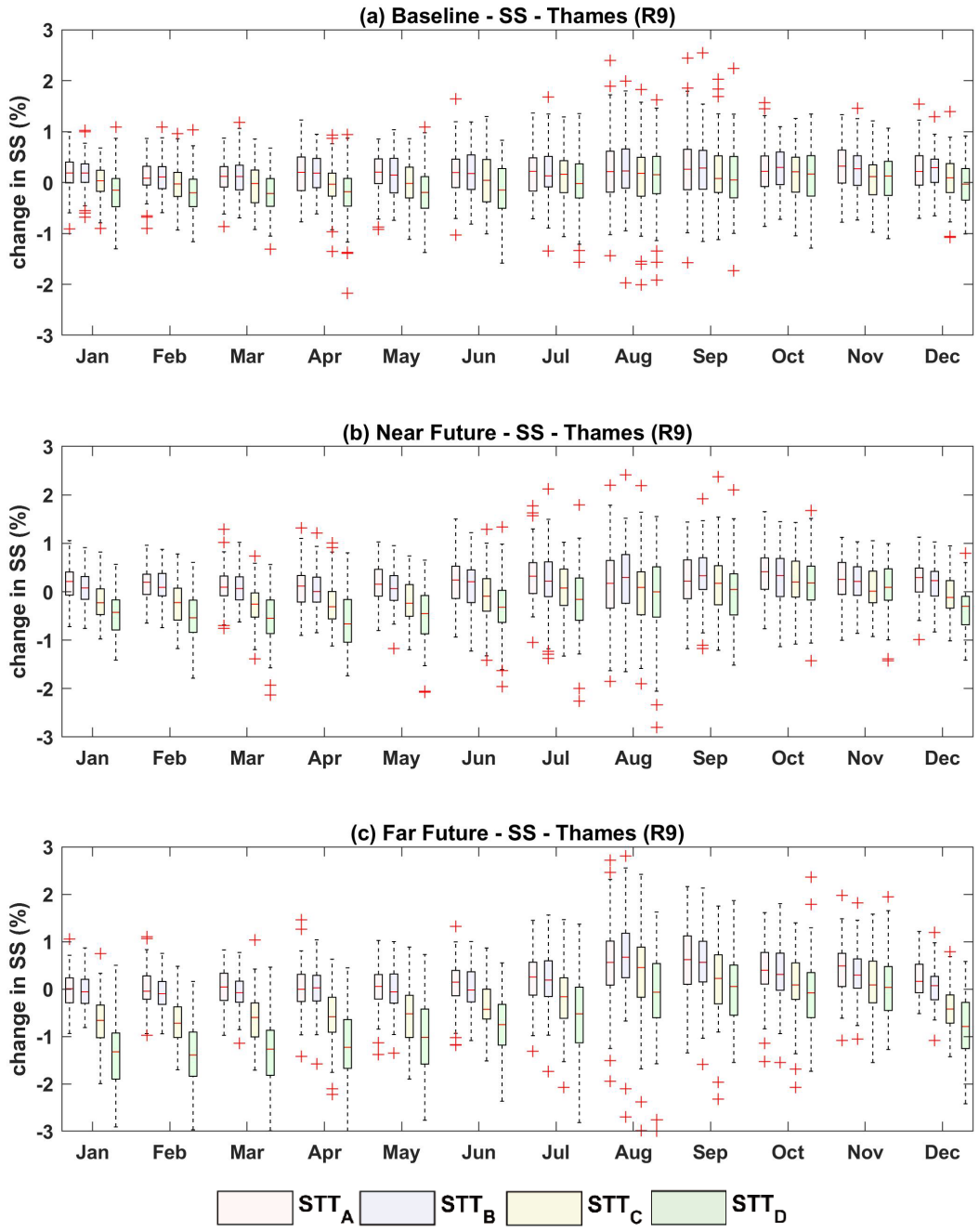


Figure 6.27: Monthly percentage change in SS concentrations under different transfer scenarios at the discharge point (Culham, R9)

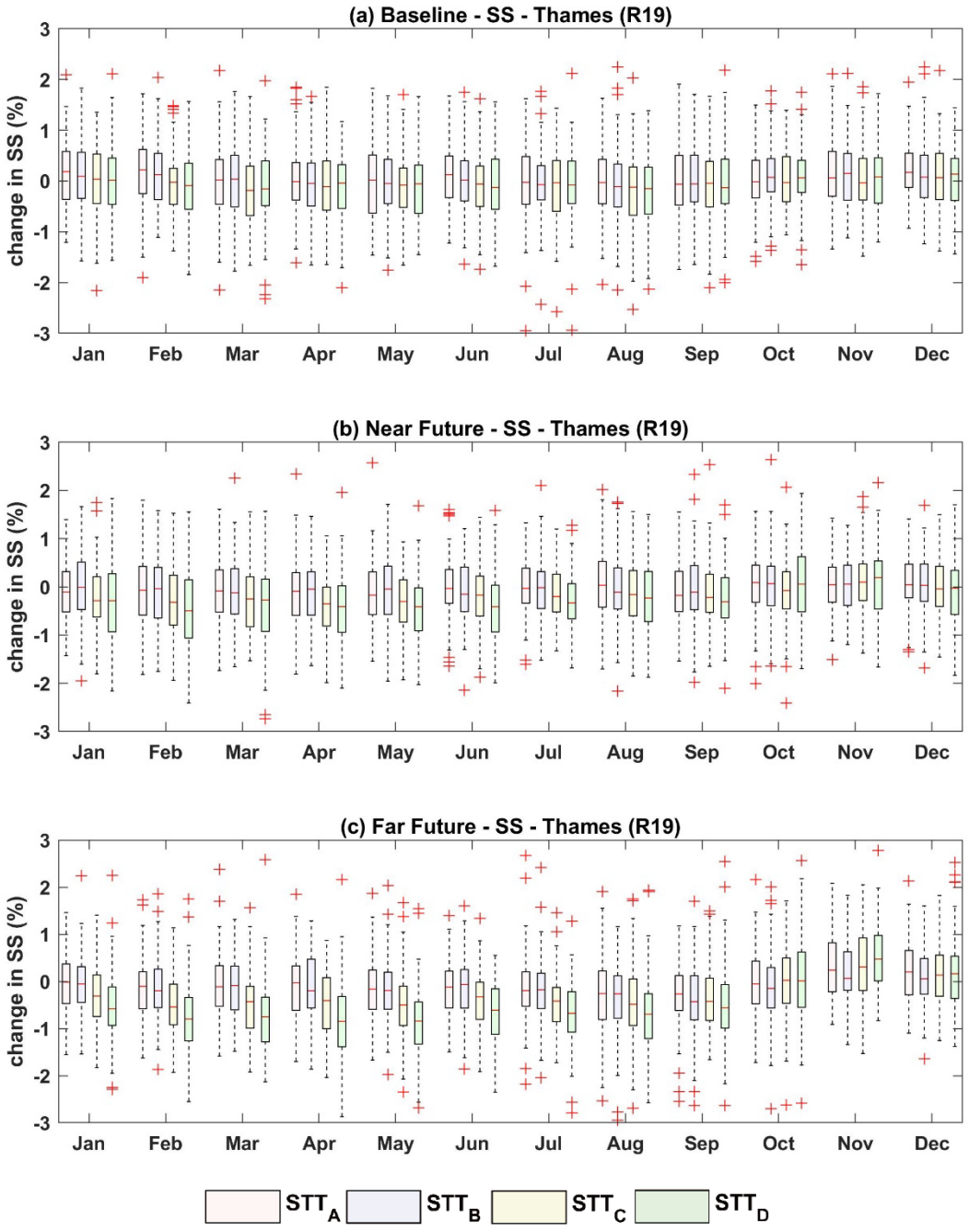


Figure 6.28: Monthly percentage change in SS concentrations under different transfer scenarios at the lower Thames (Egham, R19)

6.3.4.4 Possible risks on the water supply system

DOC concentrations greater than 5 mg/L in streams will complicate water treatment and may result in disinfection by-products to be formed in the amount of exceeding the standards (e.g. trihalomethanes) (Carpenter et al., 2013). In the Thames, the annual average DOC is higher than 5 mg/L, which is a burden in the system already. However, the transfer will further worsen DOC concentrations in the Thames when a larger volume of transfers is made. Scenario D (500 Ml/d capacity) transfer is planned as a joint operation with the Vyrnwy Reservoir redeployment of 200 Ml/d if implemented, which will increase the chance to transfer uplands higher DOC concentrations.

In terms of total phosphorus, the risk of exceeding the phosphorus threshold between the moderate and the poor river quality status defined by the EU Water Framework Directive is 0.25 mg/L. The concentrations must be less than 0.12 mg/L for 'good' and less than 0.05 mg/L for 'high' ecological status to be achieved. The following figures show the number of days per year total phosphorus concentrations above the defined thresholds (good status of 0.12 mg/L and poor status of 0.25mg/L) (Figure 6.29 and Figure 6.30). All transfer scenarios show the difficulties of maintaining good ecological status under the 0.12 mg/L threshold by exceeding this threshold most days in the year. Only 17% of water bodies in England meet good ecological status at this moment (Environment Agency, 2017), and the planned transfer will be a potential barrier to achieve good ecological status. In the near future, the number of days above the threshold of 0.25 mg/L in Scenario A and B is less than no transfer option. However, it can be seen that the larger volume transfer (from 300 Ml/d) will result in ecological status degradation and in the far future, all scenarios are responsible.

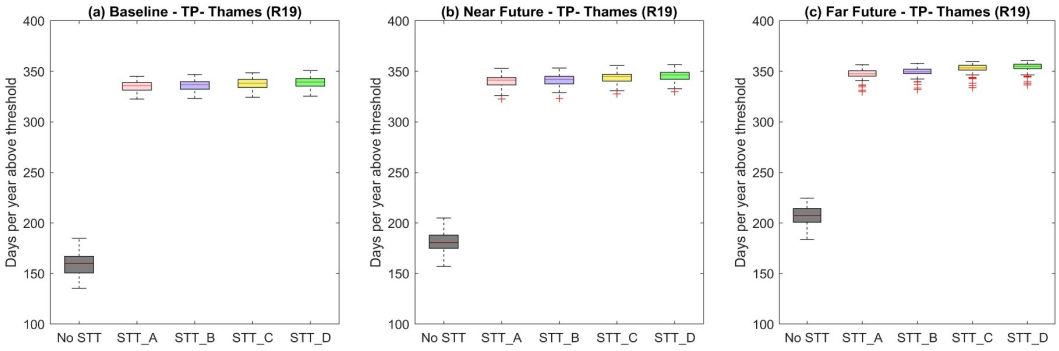


Figure 6.29 Number of days per year of phosphorus above 0.12 mg/L at the lower Thames (Egham R19)

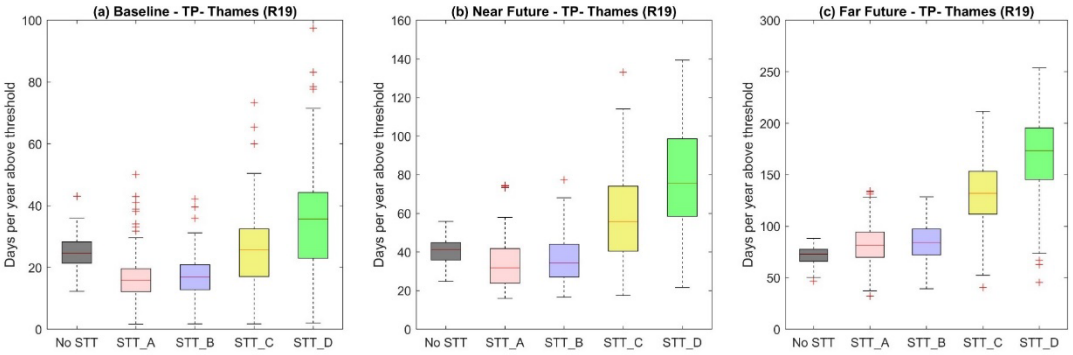


Figure 6.30: Number of days per year of phosphorus above 0.25 mg/L at the lower Thames (Egham, R19)

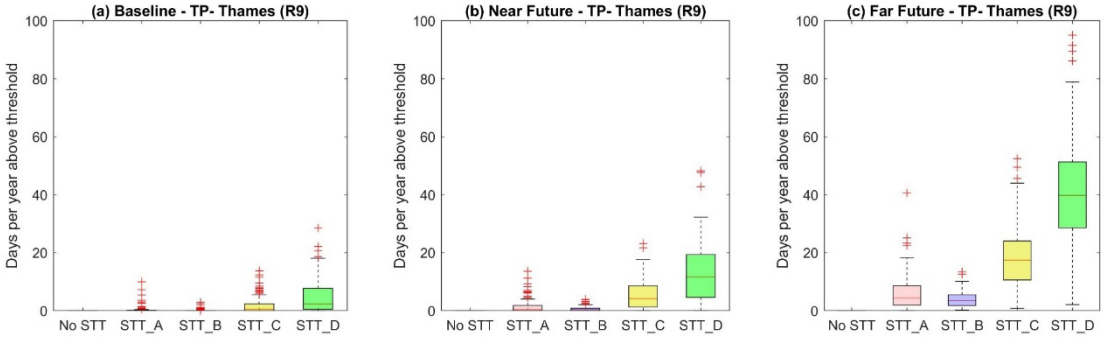


Figure 6.31: Number of days per year of phosphorus above 0.8 mg/L at the discharge points (Culham, R9)

Figure 6.31 shows the increasing number of days per year phosphorus concentration at the discharge point is above 0.8 mg/L, which Thames Water would cease water withdrawal from the river (Mortazavi-Naeini et al., 2019). Several mitigation efforts have

been made to improve the phosphorus pollutions in the British rivers, such as fertiliser application controls and the introduction of phosphorus stripping technique at the sewage treatment plants. The previous case studies in the Thames have shown by Bussi et al. (2017) and Mortazavi-Naeini et al. (2019), mitigation measures for phosphorus are effective to reduce the total phosphorus concentrations to compensate for negative impacts of climate change on river water quality. However, the proposed new transfer scheme may deliver additional burdens against the efforts in the Thames.

6.4 Conclusions

The projected intensification of hydrological droughts and climate change in two catchments in the UK means that conventional water resources options would face inevitable challenges in the water supply. This requires the improvement of our understanding of the vulnerability and sustainability of the current water supply options, and the development of new water resources options to cope with more intense, more extended, and more frequent droughts. This chapter presented the hydro-climatic projections coupled with water quality modelling in the Thames and Severn catchments to explore the potential water resources option, the Severn-Thames Transfer.

The transfer scheme is sensitive to the pattern and behaviour of droughts across the two catchments. The Severn will experience more intense droughts, whereas the Thames will expect longer droughts. Coincident droughts are projected to increase in frequency and duration in the two catchments. Although the benefits of the scheme are vulnerable to drought patterns, 29% and 37% of the time the water from the Severn is available in order to alleviate the shortages in the Thames, depending on the transfer scenarios in the far future. Interestingly, the absolute transferred volumes in both scenarios A and B remain

almost constant from the baseline to the far future as well, while the success rates are projected to decrease, which reflect the increase of demand in the Thames. Overall, the Severn-Thames Transfer scheme could provide a meaningful benefit to alleviate water shortages in the Thames if it is operated conjunctively with the London water supply system. As shown in two different scenarios A and B, the decision on the timing of the transfer is important to maximise the yield of the proposed scheme. Scenario B implies the need for a strategic reservoir to store the surplus water from the Severn.

The potential impacts of the Severn-Thames Transfer on water quality (DOC, nitrate, total phosphorus, and suspended sediment) in the Thames are assessed. Two additional Scenario C (300 Ml/d) and D (500 Ml/d) are added to explore a wider range of impacts. Large impacts on total phosphorus concentrations will be the concern of the proposed transfer scheme. Medium impact in DOC and minimal impacts on nitrate and suspended sediment are expected.

In this study, a 30- year average response presented here. An event by event or year by year analysis will be useful to inform how dynamically water quality responds to the threshold of water quality concentrations in the Severn to make a spike input to the Thames system to be used for the practical operation. Comparison between two reaches (at discharge points and the lower Thames) has an assumption that only transfer impacts are the source of changes in water quality. It is certainly a simplified version of supply-demand balance modelling to estimate hypothetical transfer options using hydrological modelling and water quality modelling as natural catchments of the River Severn and Thames are differ from actual water resources zones. It is worth noting that the multiple aspects, other than hydrological droughts, will impact the performance of the Severn-

Thames Transfer, including changes on existing water supply infrastructure, operation plans, demand growth or management scheme, land-use, catchment characteristics and politics. This analysis focused on providing the hydrological background and assessment of the potential water transfer options and those impacts to the recipient water quality, not on particular practical and operational details in terms of water loss through pipelines, and transport time from the abstraction point to the discharge point or any of infrastructure constructions.

7 Conclusion

7.1 Thesis Summary

Climate change will stimulate water-engineering responses in anticipation of frequent and severe extreme events such as droughts. Water stress in South East England is projected to increase over the next planning horizon, and water planners are continuously looking to develop drought management options with neighbouring water companies to transfer water cross basins to alleviate some of the water stress. The Severn-Thames Transfer is actively discussed as the preferred investment scheme for the respective catchments (Thames Water, 2020a). The principal aim of this doctoral thesis has been to provide a comprehensive modelling representation of the hypothetical inter-basin water transfer scheme under extreme and spatially extensive drought events to explore the possible occurrence of trade-offs between water quality and quantity within the proposed transfer scheme.

The following objectives were identified to achieve this aim: (1) to assess the impact of climate change on river flows and water quality in future droughts in the River Severn (donor river) and the River Thames (recipient river); (2) to explore the long-term carbon dynamics in the headwater of the River Severn and investigate the possible risk to the Severn-Thames Transfer; (3) to identify the hypothetical operation rules of the Severn-Thames Transfer scheme by investigating water availability in the River Severn for a transfer while the Thames experiences public water supply drought; and (4) to assess the impacts and risks on the water quality in the River Thames driven by the proposed transfer schemes.

Chapter 2 presented a critical discussion of water quality responses in droughts and how the catchment problems can be incorporated into the modelling framework. Together with **Chapter 2**, c**hapter 3** described how modelling tools and data were selected to provide inputs for the analytical chapters.

Chapter 4 focused on providing a thorough assessment of drought impacts on hydrology and water quality in the two study catchments. The water quality analysis focused on four water quality variables such as Dissolved Organic Carbon (DOC), nitrogen, suspended solids, and phosphorus. Nutrients (nitrogen and phosphorus) and suspended solids are also important water quality factors that pose serious threats to the water supply in the River Thames. High DOC concentrations have been observed in the rivers, which might mean additional treatment is required to remove DOC prior to connection with the water supply. According to the INCA model results presented, the concentration of DOC, nitrate, phosphorus, and suspended sediments is projected to vary due to the changing climate and drought in two future time series. The impact of the considered scenarios is to reduce the summer flows considerably, thus causing an overall decrease in summer DOC, nitrate, and suspended sediment and an increase in summer phosphorus in both catchments. However, the lower Severn shows different system responses to DOC and nitrate concentrations compared to others regions/reaches (the Thames and the upper Severn) and the degree of degradation in water quality in the lower Severn is a concern for when considering any proposed transfer between two catchments.

Chapter 5 focused on water quality issues for the donor river, the River Severn. The rising trends of DOC in the headwater of the River Severn have been observed over recent decades and with the proposed water transfer there is a chance that high DOC water might

be transferred across to the Thames. This might mean extra treatment is required to remove DOC prior to contributing to water supply. The main findings are focused on assessing whether the rising DOC concentration trends are likely to be continued or stabilised in the future. Taken across future climate, sulphate scenarios and the identified enzymatic latch parameters, the projected DOC concentrations in Plynlimon will continue to increase in the near future and then will be stabilised in the far future. However, in the far future, seasonal patterns of DOC concentrations will change, with a post-drought DOC surge in the autumn months.

Chapter 6 provided an innovative modelling representation of the hypothetical inter-basin water transfer scheme, the Severn-Thames Transfer, under extreme and spatially extensive drought events. Using projected future river flows (in **Chapter 4**), future hydrological drought events were identified to assess how the characteristics of droughts and coincident droughts between two catchments would change under the projected climate change scenarios. This chapter then included a demonstration of how the Severn-Thames Transfer scheme would work if it is implemented in future water resources systems, and what the expected performance would be. The proposed transfer scheme is sensitive to the pattern and behaviour of droughts across the two catchments. The Severn will experience more intense droughts, whereas the Thames will expect longer droughts. Coincident droughts are projected to increase in frequency and duration in the two catchments. In terms of water quality impacts, total phosphorus and DOC will be a concern and introduce a large burden on water supply systems and ecological conditions in the Thames. The timing and trigger for the transfer will become an important factor when minimising water quality risks, because a transfer can act as a semi-permanent point

source to the recipient river during droughts. Reduced nitrates and suspended sediment are expected to help improve water quality in the Thames as the water quality in lower Severn is generally better.

7.2 Research contributions

The principal contributions of the research presented in **Chapters 4**, **5** and **6** are summarised below:

i. A thorough assessment of future drought and climate change impacts on river flow and water quality in two major catchments in the UK. To date, this is the first integrated water quality modelling study of whole River Severn system.
ii. Catchment specific water quality issues are identified. This thesis also demonstrates the successful implementation of new parameter sets that simulate enzymatic latch processes in carbon dynamics.
iii. The innovative representation of an inter-basin transfer analysis, incorporating both water quality and quantity, and water resources system information together.
iv. The quantification of possible risks and impacts on water quality from the proposed new inter-basin scheme under severe future droughts in the Thames.

7.3 Limitations and future research

Several opportunities exist to improve on the research presented here. This section is intended to identify the limitations of the thesis and to suggest future research.

i. Land use change was assumed to be a constant over the next 100 years. This is obviously a huge limitation on the catchment modelling as effective land use can reduce the pressure on the water quality posed by climate change (Bussi et al., 2017). Future research is suggested to using combined scenarios with land use projections.

ii. The thesis covers four water quality variables, carbon, nitrogen, phosphorus, and suspended sediment. However, there are more contaminants of emerging concerns such as metals, persistent organic pollutants (known as POPs), metaldehyde, pathogens, pharmaceuticals etc. The pollution of metaldehyde in the River Severn is critical (Cooke et al., 2020), and this might cause considerable concern if it is transferred to the Thames. For example, water quality modelling of metaldehyde by using INCA model was successfully applied to the Thames basin (Lu et al., 2017) and this can be extended to the Severn catchment modelling.

iii. A more comprehensive consideration of the water resources system component could have the potential to extend this research. This thesis included consideration of basic water resource system information to develop a relatively simple transfer scenario. However, the water resources system is a complex system and more scenarios could be used to test a wider range of options. Recent work by Murgatroyd and Hall (2020) assessed the resilience of the Severn-Thames Transfer under severe future droughts and demand growth scenarios through the

application of water systems modelling. Their paper presented detail underpinning dynamic water system operations to evaluate the trade-offs in the new transfers, test multiple capacities and operation conditions in the two water companies, Thames Water and Severn Trent. Their main findings concur with this chapter's finding that the new transfer will bring increased resilience of water supplies in general, and the success of water transfer strategies and the drought resilience in these water resources zones will be fostered with supporting reservoirs (Murgatroyd and Hall, 2020) and those optimal locations from exporting and importing catchments (Dobson et al., 2020). Incorporating water quality modelling into the water resources modelling could provide rigorous management choices.

iv. Finally, further validations of the presented modelling framework may be required in other catchments in the UK or outside of the UK. Data availability can be a huge barrier. However, the INCA models have proven to be reliable in data scarce catchments, based on recent applications in Ethiopia (Bussi et al., 2021b) and the Mekong Delta (Whitehead et al., 2019b, Bussi et al., 2021a).

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