Impacts of Climate and Land Use Change on Groundwater Quality in England: A Scoping Study

Environmental Change, Adaptation and Resilience Programme
Open Report OR/22/076

BRITISH GEOLOGICAL SURVEY

ENVIRONMENTAL CHANGE, ADAPTATION AND RESILIENCE PROGRAMME

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PROJECT BACKGROUND

In 2019, the Environment Agency (EA) carried out a risk assessment following the government Climate Change Committee's methodology. This work highlighted groundwater quality as one area where the quality of the EA's plan was weak, and progress in managing the risk was poor. In the same year, the UK parliament declared a climate emergency, and the COP26 summit in 2021 has brought climate change and the need to adapt into even stronger focus. This prompted the need for further evidence gathering and in 2021 the EA commissioned BGS to undertake a scoping study to explore the impacts of climate and land use change on groundwater quality. The study had the following objectives: (1) To determine what key risks to groundwater quality may be associated with climate change, (2) what adaptation and mitigation measures may be needed, (3) how EA groundwater quality monitoring may need to change in the future associated with climate change and (4) what are the research and evidence gaps associated with the impacts of climate and land use change on groundwater quality. The study addressed the aims above through a number of desk-based activities which are detailed in this report.

LITERATURE REVIEW

A review of the findings of UKCP18 has illustrated the potential changes to the climate of England over the 21st century as context for changes in groundwater resources and quality. Air temperature, evapotranspiration and sea level are all predicted to increase throughout the 21st century. Whilst the direction of change in annual precipitation is unclear, wetter winters and drier summers are predicted, with greater magnitude extreme winter rainfall events. No published work has evaluated the impact of climate change based on UKCP18 data on groundwater recharge and levels. A review of previous studies using UKCP09 and other climate projections has shown limited consistency in the direction of change in long term average groundwater recharge and levels in England. There is some consistency in changes to seasonality in groundwater recharge and levels, with increased recharge and levels in winter, decreased recharge and levels in summer. There is limited evidence for changes in extremes (increasing high winter groundwater levels). A review of international literature related to climate change and groundwater quality has shown an overall worsening of groundwater quality over the next 50 – 80 years, although the trajectory of change for individual parameters is highly uncertain. Some parameters have a high level of confidence in a relationship with climate variables (e.g. shallow groundwater temperature and air temperature, sea level rise and salinity in coastal aquifers). However, for many components of climate change and water quality parameters, our understanding of relationships is near non-existent and speculative.

STAKEHOLDER PERSPECTIVES ON GROUNDWATER QUALITY INTO THE FUTURE

A workshop was held on "Groundwater Quality into the Future" as part of this study. The purpose of the workshop was to gather input from both Environment Agency and external stakeholders regarding the key issues related to future groundwater quality, and the priorities for adaptation, management and research. This workshop identified uncertainty in impacts of climate change on groundwater quality, the need for holistic approaches to management of groundwater in the terrestrial water cycle, and the need for continued monitoring as cross cutting themes. A number of focus areas were also identified: nutrients, emerging substances, changing rainfall characteristics, changing temperature, groundwater rebound, urban development and construction, changing salinity and groundwater ecosystems.

CASE STUDIES

The potential impacts of climate change on groundwater quality are illustrated through five case studies – Brighton, Chichester, Birmingham, Eden and Dove. The case study areas cover a range of different hydrogeological (Chalk, Permo-Triassic sandstone and Carboniferous Limestone), geographical (north, south, inland, coastal) and land use settings (rural, urban).

For each case study we discuss the hydrogeological conceptualisation and water quality issues of concern. We then present the results of UKCP18 (temperature, rainfall) and the derived products eFLaG (rainfall, evapotranspiration, groundwater recharge, groundwater levels) and GeoCoast (sea level rise), before providing a qualitative evaluation of the impacts of climate change on groundwater quality. Across all five case study areas, air temperatures are predicted to increase by up to 3°C. This could increase reaction rates for degradation of contaminants, but such increases may only be marginal. Increased sea levels are predicted to increase salinity in coastal aquifers. The direction of changes in long term average rainfall and recharge is uncertain, but the magnitude of changes is predicted to be small. There is generally a high confidence of increased rainfall and recharge seasonality and greater magnitude of extreme events in winter. This has the potential to result in spikes of pollutants, but this could also be offset by increased dilution. Land use change, and in the case of Birmingham, groundwater level recovery from historic over-abstraction, may have a greater impact on groundwater quality than changes in climate.

PRIORITISED RISKS TO GROUNDWATER QUALITY AND RECOMMENDATIONS

On the basis of the literature review, case studies and input from stakeholders, an initial prioritisation of the potential risks to groundwater quality associated with climate change has been made. The relatively small increases in temperatures and changes in long term average rainfall and recharge make these a low priority. The local nature of increases in sea level affecting coastal aquifers make these a medium priority. The high confidence in changes in rainfall and recharge seasonality and extremes and impact through changes to leaching, spikes and dilution make these a relatively high priority. The highest priority risk is land use change, whether induced by climate change or otherwise. Land use change may change contaminant sources and pathways, and is both highly uncertain and has a potentially high impact on both groundwater and other components of the terrestrial water cycle.

Building on the previous project tasks, a number of recommendations have been made regarding evidence gaps, monitoring approaches, regulation and adaptation measures. Specific recommendations are detailed in the table below and general recommendations are discussed herein. Further research is required to address the significant evidence gap related to how drivers of groundwater quality are likely to change in the future, and what the hydrogeological system response to changes in multiple, competing drivers may be. This is a large area of work and should be prioritised based on stakeholder needs. Subsequent work is required to consider the impacts of future changes in groundwater quality on downstream receptors, and what management strategies should be adopted. Recommendations for changes in groundwater quality monitoring detailed below are speculative at this stage given the high level of uncertainty associated with the impacts of climate change on groundwater quality.

A key recommendation from the workshop was for better integration of groundwater resources and quality in regulation, as well as better integration of groundwater as a whole within the terrestrial water cycle and urban planning. Given the uncertainty regarding the impacts of climate and land use change on groundwater quality, “no regrets” adaptation measures are most appropriate at this time. These measures, detailed below will address groundwater quality needs under current climate and land use and in any future. However, as “no regrets” measures address current groundwater quality issues, future issues which are not currently a concern (e.g. the next generation of emerging contaminants) will not be impacted by these approaches. This highlights the importance of addressing the evidence gaps above through targeted research projects. Detailed project proposals to address these gaps are beyond the scope of this report and should be co-produced between the Environment Agency, BGS and other stakeholders.

RECOMMENDATIONS SUMMARY

Evidence gaps
Changes in nutrient sources and pathways
Impacts of abstraction reductions on groundwater quality
Changing rainfall and recharge characteristics
Future emerging contaminants
Function of groundwater ecosystems
Toxicology of existing and emerging contaminants
Approaches to management of saline waters

Monitoring
Prioritise existing long term monitoring
New monitoring techniques (e.g. remote and automated sensing)
Improve GWQMN coverage including "good" status GWBs where quality may deteriorate
Review data periodically (c. 6 - 10 years)

Regulation/adaptation
Catchment nutrient budgets and targets
Reduction of agricultural nutrient losses
Rural land use change to increase water storage and reduce runoff
Flexible abstraction regimes
Greater enforcement of groundwater protection guidance and regulations
Use of real-time monitoring for contaminant spikes

1 Introduction

1.1 BACKGROUND

In 2019, the Environment Agency (EA) carried out a risk assessment following the government Climate Change Committee's methodology. This work highlighted groundwater quality as one area where the quality of the EA's plan was weak, and progress in managing the risk was poor. In the same year, the UK parliament declared a climate emergency, and the COP26 summit in 2021 has brought climate change and the need to adapt into even stronger focus. This prompted the need for further evidence gathering and in 2021 the EA commissioned BGS to undertake a scoping study to improve our understanding of the impacts of climate and land use change on groundwater quality in England.

The project aimed to understand the following:

1. What the key risks to groundwater quality may be associated with climate change?
2. What adaptation and mitigation measures may be needed?
3. How EA groundwater quality monitoring may need to change in the future associated with climate change?
4. What are the research and evidence gaps associated with the impacts of climate and land use change on groundwater quality?

1.1 PROJECT STRUCTURE AND REPORT OVERVIEW

The project addressed the aims above through a number of desk-based activities which are detailed in this report.

In Section 2, we first report the results of literature reviews related to (1) the impacts of climate change on physical meteorological and hydrogeological variables that may affect groundwater quality in England based on UKCP18 and (2) peer-reviewed research that evaluates the impacts of climate change on groundwater quality both in England and internationally. We also outline possible relationships between components of climate change and groundwater quality variables.

In section 3 we then consider stakeholder perspectives on groundwater quality into the future building on a workshop undertaken for this project. In section 4 we further explore the relationships between components of climate change and groundwater quality discussed in Section 2 in five exemplar case studies which cover a range of different hydrogeological and land use settings. For each case study, we detail the hydrogeological conceptualisation and groundwater quality issues of interest. We then present the results of the latest climate projections from UKCP18 and derived products such as the enhanced future FLow and Groundwater (eFLaG) project, before considering the possible impacts of these changes on groundwater quality.

Building on the potential relationships between climate change and groundwater quality in previous sections, section 5 explores potential changes required to the Environment Agency's groundwater quality monitoring network in response to climate change. Finally, in section 6 we detail recommendations from this project for evidence gaps, monitoring approaches, regulation and adaptation measures.

2 Literature Review

2.1 IMPACTS OF CLIMATE CHANGE ON PHYSICAL METEOROLOGICAL AND HYDROGEOLOGICAL VARIABLES IN ENGLAND

2.1.1 Overview of the approach

In this section we review the literature related to the impacts of climate change on the physical hydro-meteorological and hydrogeological variables in the UK. In 2018, the UK Met Office released the UK Climate Projections 2018 (UKCP18, Met Office (2018)). UKCP18 is the most up to date assessment of how the climate of the UK may change to 2100, and represents a significant advance from UKCP09 in terms of the climate science, observations and products available. UKCP18 provides information related to all the physical hydrometeorological variables and metrics detailed in the project proposal. These are:

1. Variables: temperature, precipitation, evaporation, sea level rise
2. Metrics: changes in long term average, seasonality, extremes

Consequently, in this review we initially focus on the results of UKCP18 in section 2.1.2. UKCP18 does not, however, provide information for the groundwater related variables (groundwater levels and groundwater recharge). We therefore reviewed the literature for specific studies related to impacts of climate change on groundwater levels and recharge. This is detailed in section 2.1.3.

It should be noted from the outset that the purpose of this literature review is to provide a conceptual basis for how changes in climate may affect the variables and metrics above, illustrated through the latest Met Office and peer reviewed literature. This will support the development of the conceptual understanding of how the likely future changes in these variables may affect groundwater quality. A detailed numerical assessment of the likely changes in these variables is beyond the scope of this task, and is addressed for the case study areas explicitly in section 3.

2.1.2 Physical hydro-meteorological variables

2.1.2.1 Temperature

Murphy et al. (2018) detail the results of UKCP18 projections for air temperature. Long term average temperatures increase everywhere but there is some spatial and temporal variability. The results are reported to be broadly consistent with UKCP09, with warmer (and wetter) winters, hotter (and drier) summers. There notable overlaps in the ranges of absolute future temperatures between UKCP09 and UKCP18, although seasonally there are some differences, with UKCP18 producing less warming than in winter in South East England than UKCP09 for a common scenario.

Seasonally, more warming is predicted in summer than winter, with a pronounced north-south contrast for the UK. This is illustrated in Figure 1 and Figure 2. Figure 1 shows changes in winter surface air temperature for 2061 relative to 1981-2000 under RCP8.5 derived from the 12-member Regional Climate Model (RCM) in UKCP18. Figure 2 shows the same but for summer air temperatures. It can be observed that summer air temperatures rise substantially more than winter air temperatures, and the change in summer air temperatures is more spatially variable with more warming in the south than north.

Recent research has explored changes in extremes of temperature in the UKCP18 projections. Kennedy-Asser et al. (2021) (Figure 3) show the rate of increase of the mean and 95th percentile summer maximum temperature above the global mean temperature from the UKCP18 global, regional and CPM projections, and for CMIP5. It can be observed that whilst both the UK mean and 95th percentile summer maximum temperature are increasing more than the global mean temperature, the 95th percentile summer maximum temperature appears to increase substantially quicker. Arnell and Freeman (2021) evaluate temperature extremes in UKCP18 regional projections from a human health perspective. Under all RCP scenarios, heatwaves and cold weather events are reported to increase and decrease in frequency, respectively. However, the change in cold weather events is smaller than for heatwaves. Similar to the projections of changes in seasonal mean temperatures (Figure 1 and Figure 2), a north-south divide is present in the temperature extremes. Smaller changes occur in the north than in the south both in terms of increases in heatwaves and decreases in cold weather events.

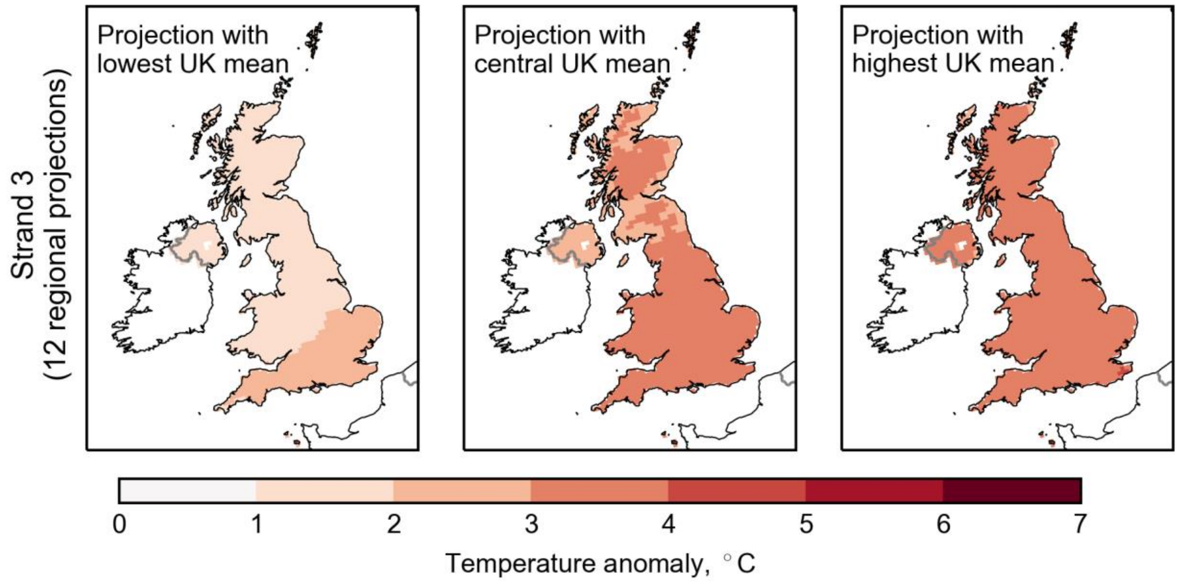

Figure 1 Maps of changes in winter surface air temperature for 2061-2080 relative to 1981-2000 under RCP8.5 from regional projections. Reproduced after Murphy et al. (2018). Contains public sector information licensed under the Open Government Licence v3.0.

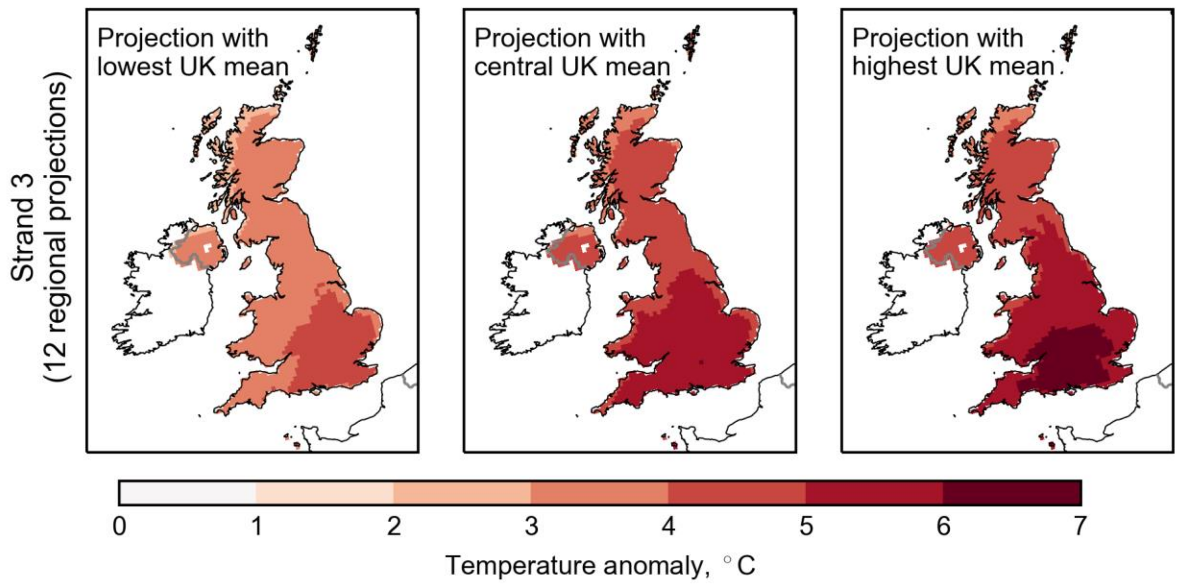

Figure 2 Maps of changes in summer surface air temperature for 2061-2080 relative to 1981-2000 under RCP8.5 from regional projections. Reproduced after Murphy et al. (2018). Contains public sector information licensed under the Open Government Licence v3.0.

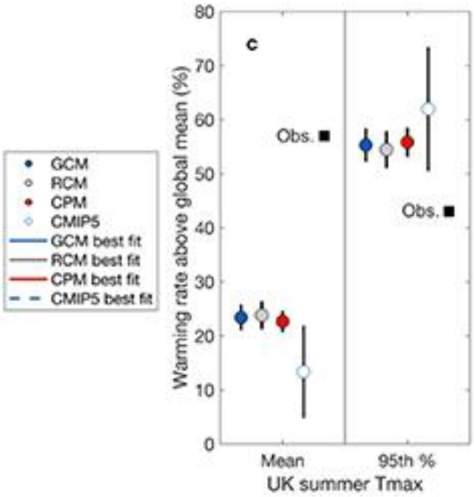

Figure 3 Warming rates for mean and 95th percentile summer Tmax above global mean surface temperature warming rate for UKCP18 GCM, RCM and CPM projections and CMIP5. Modified after Kennedy-Asser et al. (2021) and licenced under CC-BY 4.0.

2.1.2.2 Precipitation

UKCP18 projections for precipitation are detailed by Murphy et al. (2018). The results of UKCP18 are broadly consistent with UKCP09 in that drier summers and wetter winters are predicted, although natural variation means that there will be some dry winters and wet summers.

Figure 4 shows the spatial variability in the change in winter precipitation between 1981-2000 and 2061-2080 under RCP8.5 from the UKCP18 regional projections. Across England and for all regional projections winter precipitation increases. There is clear spatial variability in these increases, with the changes greatest in central and southern England and smallest in far northern England. Figure 5 shows the same information for summer precipitation. Whilst drier summers appear to occur across England, the greatest decreases in summer precipitation are predicted to occur in southern and southwest England, and the smallest change in northeast England.

Changes in extreme precipitation are reported by Kendon et al. (2019) based on the UKCP18 convection-permitting model (CPM) runs. An increased frequency and intensity of wet days in winter is reported, with bigger heavy daily rainfall events (change in 99th percentile). This is highlighted in Figure 6, which shows changes in the 99th percentile winter rainfall event across the UK for the CPM (top panels), the CPM re-gridded to the RCM grid (middle panels) and the regional projections (bottom panels). In both the CPM and RCM projections the magnitude of the 99th percentile winter precipitation event increases across the UK. In summer, Kendon et al. (2019) report decreases in the frequency of wet days but increases in precipitation intensity.

Changes in meteorological droughts in UKCP18 have been reported by Hanlon et al. (2021), who showed that the extent of changes to meteorological drought is affected by the extent of warming. Changes in the drought severity index (monthly anomalies of n month precipitation deficits) for different accumulation periods and levels of warming derived by Hanlon et al. (2021) using the UKCP18 regional projections are shown in Figure 7. No significant differences between the drought severity index for the observation period (1981-2000) and a 1.5 degrees rise was observed. Beyond 1.5 degrees, significant increases in the drought severity index were reported over all accumulation periods.

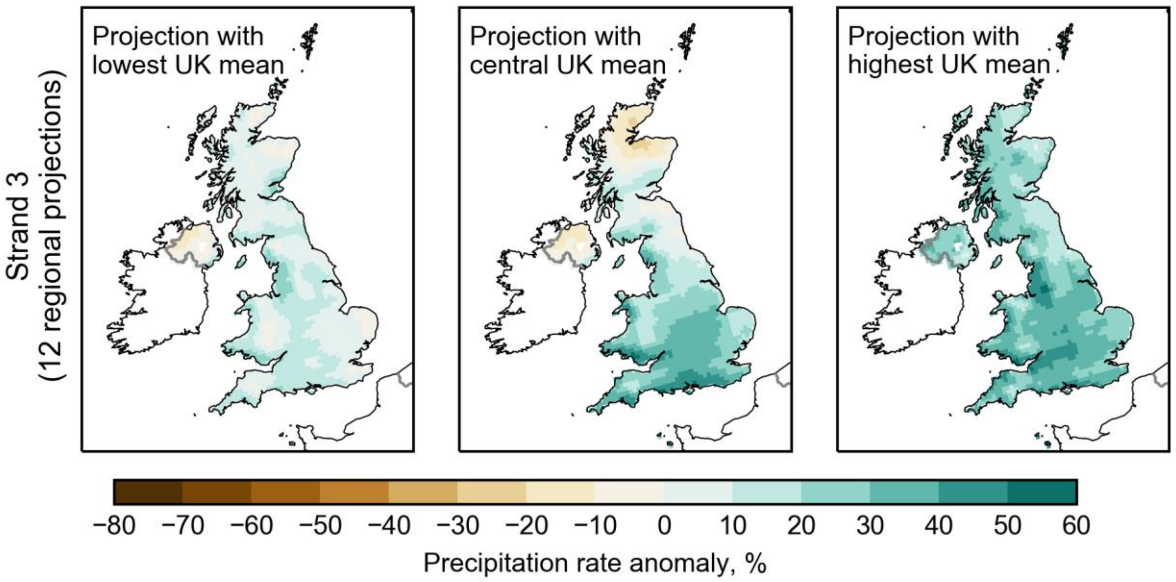

Figure 4 Maps of changes in winter precipitation for 2061-2080 relative to 1981-2000 under RCP8.5 from regional projections. Reproduced after Murphy et al. (2018). Contains public sector information licensed under the Open Government Licence v3.0.

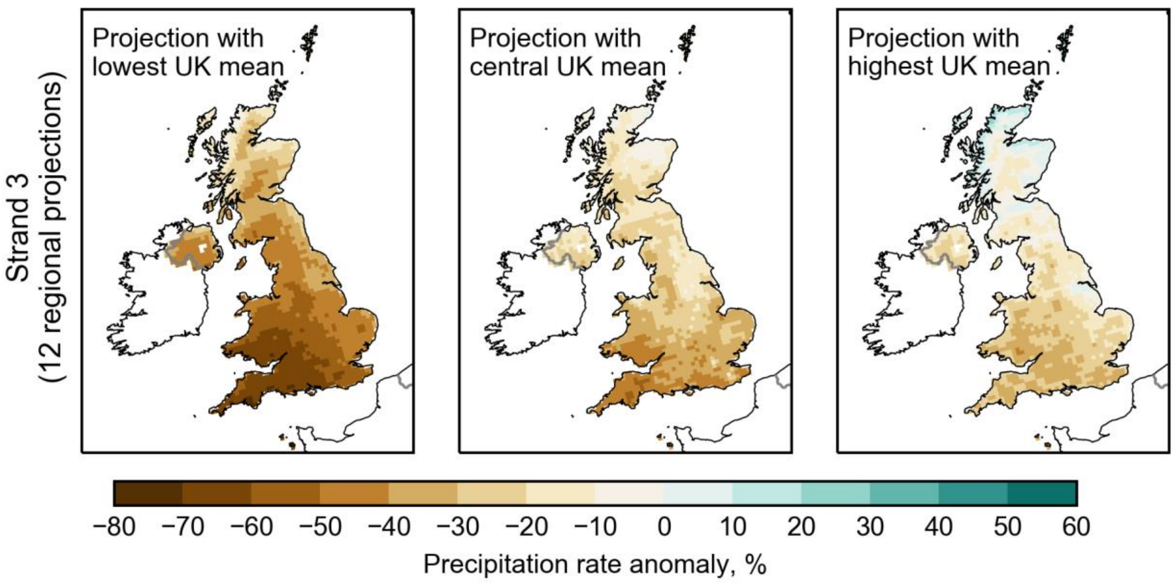

Figure 5 Maps of changes in summer precipitation for 2061-2080 relative to 1981-2000 under RCP8.5 from regional projections. Reproduced after Murphy et al. (2018). Contains public sector information licensed under the Open Government Licence v3.0.

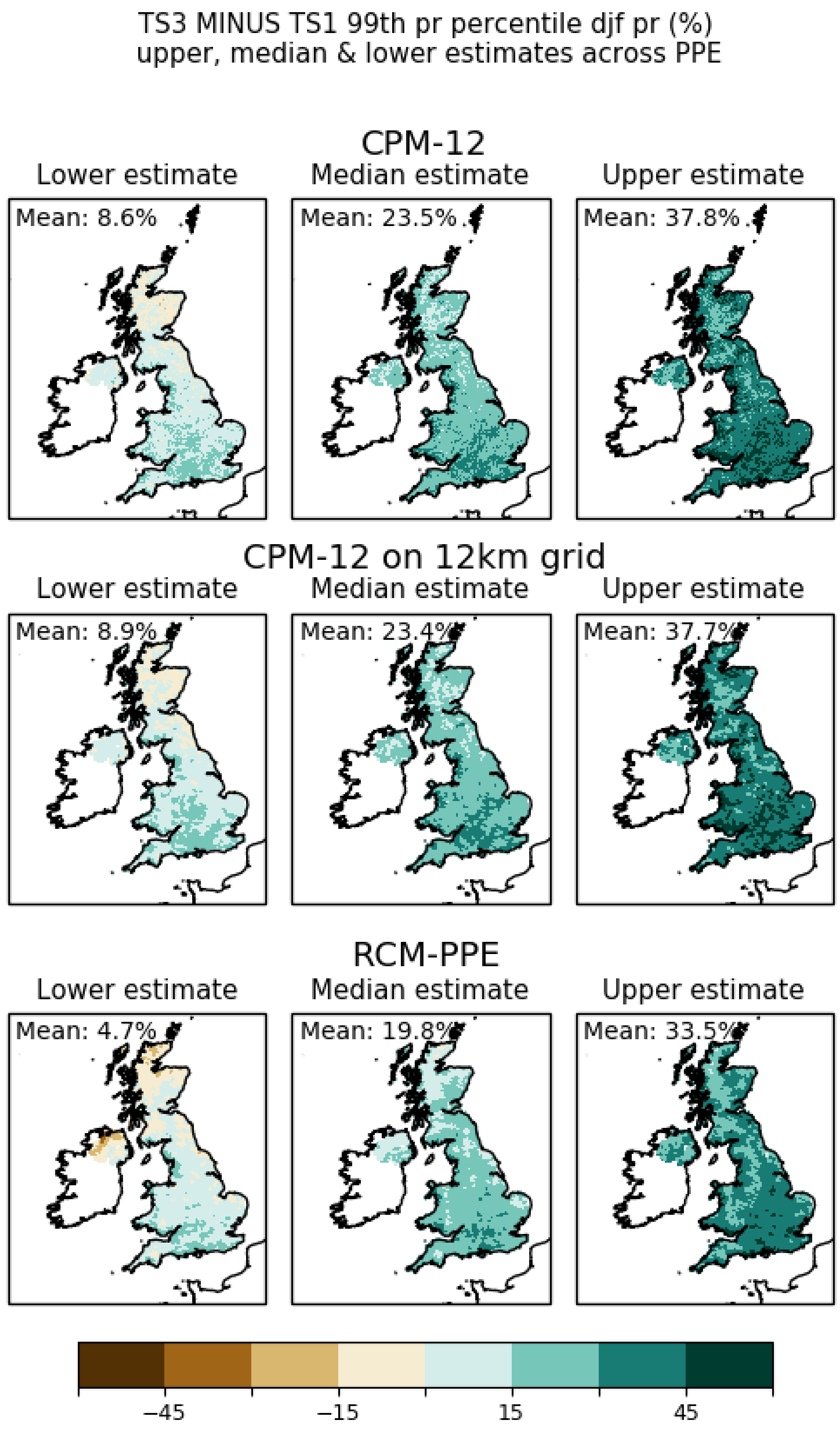

Figure 6 Change in the 99th percentile of daily mean precipitation in winter between the baseline (1981-2000) and future (2061-2080) periods. Reproduced after Kendon et al. (2019). Contains public sector information licensed under the Open Government Licence v3.0.

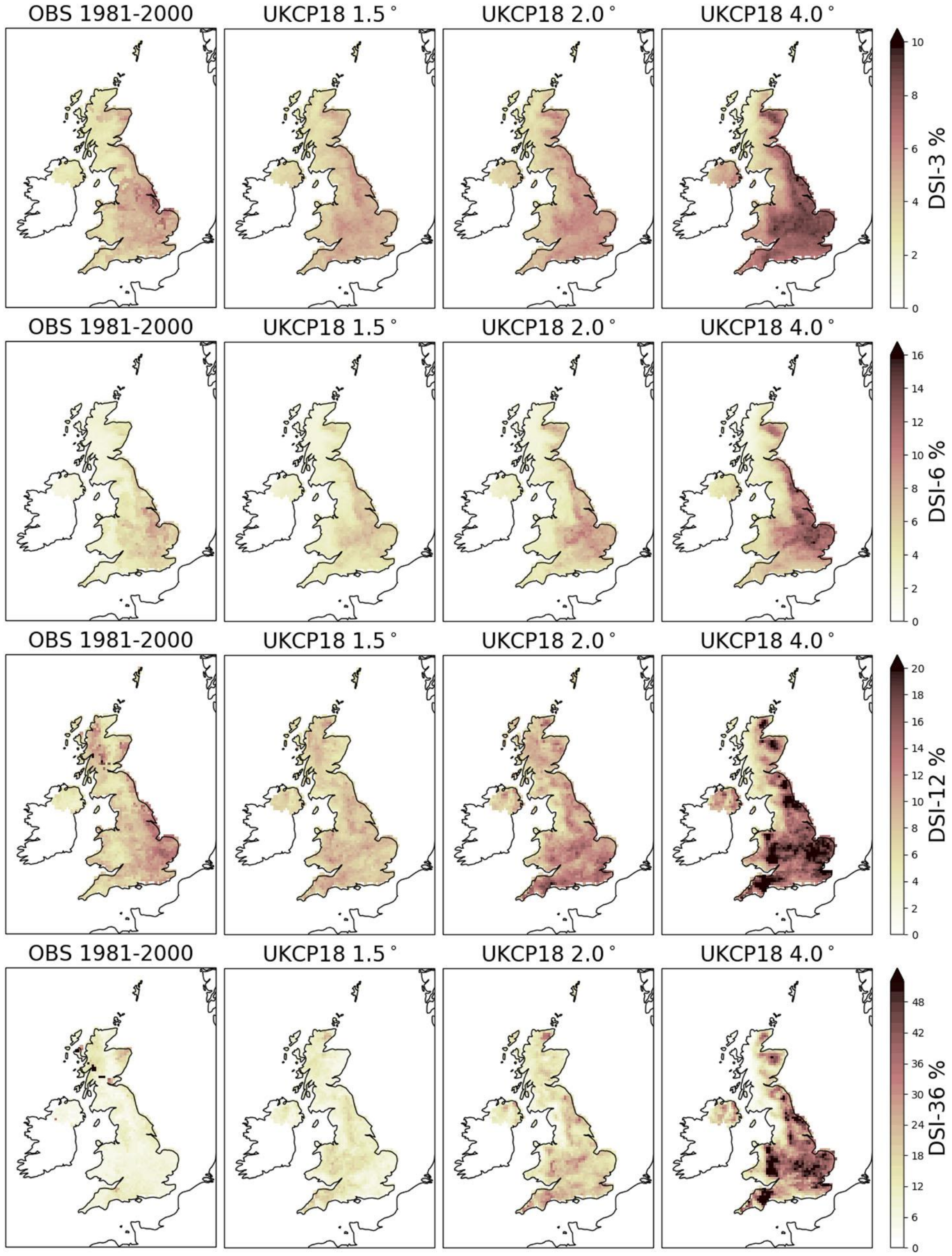

Figure 7 Change in drought severity index (monthly anomalies of n month precipitation deficits) Reproduced after Hanlon et al. (2021) and licenced under CC-BY 4.0.

2.1.2.3 Evapotranspiration

Murphy et al. (2018) and Pirret et al. (2020) provide an outline on the impacts of climate change on evapotranspiration based on the UKCP18 projections. Both canopy evaporation and soil evapotranspiration are projected to increase through much of the year, related to increased temperatures under climate change. Some seasonal variability is likely to occur, with reduced canopy evaporation and soil evapotranspiration in summer and early autumn due to reduced moisture availability. This occurs earlier in the year for canopy evaporation due to access to soil moisture storage for evapotranspiration. Robinson et al. (2021) provide potential evapotranspiration data based on the UKCP18 RCM data, however no interpretation of this data has been provided to date.

2.1.2.4 Sea level rise

The UKCP18 marine projections are summarised by Palmer et al. (2018). Sea level rise in UKCP18 is projected to be greater than in UKCP09 for similar emissions scenarios. An increase in the frequency and magnitude of extreme water levels around the UK coastline is predicted, resulting in an increase in coastal flood risk. The extent of sea level rises varies across the UK, with greater increases predicted in southern England than Scotland (Figure 8, right). Projections also vary substantially based on emissions scenario (Figure 8, left).

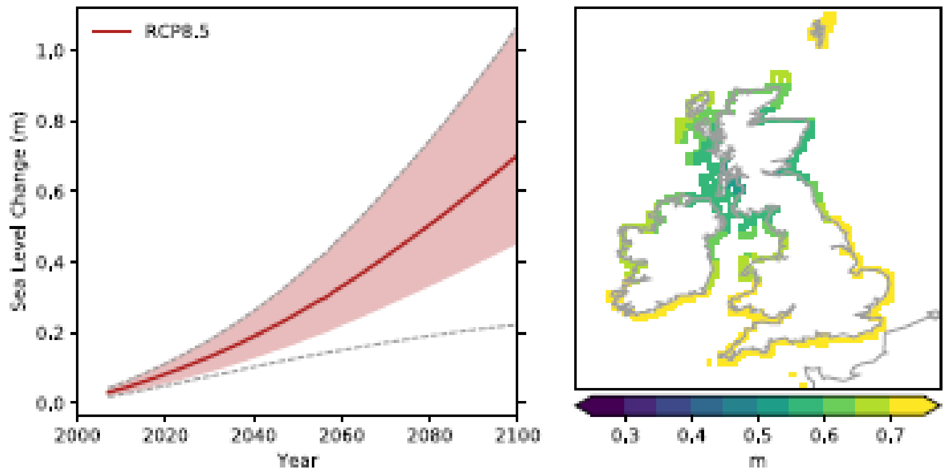

Figure 8: Left: Mean annual sea level change time series (left) for the UK in comparison to 1981-2000 under RCP8.5. Bold line is the median change and the shaded area the 5th – 95th percentile range. The dotted line shows the range across RCP2.6 to RCP8.5. Right: regional relative sea level change around the UK and Ireland coastline in 2100 for RCP8.5. Reproduced after Palmer et al. (2018). Contains public sector information licensed under the Open Government Licence v3.0.

2.1.2.5 SUMMARY OF CHANGES IN CLIMATE

Table 1 provides a summary of the changes in climate derived from the UKCP18 projections.

Table 1 Summary of changes in climate from UKCP18 reports

Variable	Long term average	Seasonality	Extremes
Temperature	Increases	Warmer winters and drier summers, biggest changes in S England, smallest in N England	Greater increases in extreme temperatures than mean temperatures, increasing frequency of heatwaves. Larger change in S England than north
Precipitation	Variable change in annual precipitation. Wetter winters, drier summers, biggest changes in S England, smallest in N England		Increasing intensity and frequency of wet days with bigger extreme daily rainfall events in winter. Decreases in wet day frequency but increases in intensity in summer. Significant increases in drought severity beyond 1.5 degrees warming
Evapotranspiration	Increases	Reduced ET in summer and early autumn due to reduced moisture availability	
Sea level rise	Increases	Increasing frequency and magnitude of extreme sea levels during winter storm surges	

2.1.3 Groundwater recharge and levels

2.1.3.1 Overview

Globally there are a number of recent reviews on climate change impacts on groundwater resources (Atawneh et al., 2021; Dragoni and Sukhija, 2008; Earman and Dettinger, 2011; Green et al., 2011; Taylor et al., 2013) and further work providing synopses of a number of these papers (Smerdon, 2017). Consequently it is beyond the scope of this report to detail these reviews, and the reader is referred to the references above for more information.

In this section we review the impacts of climate change on groundwater recharge and levels in the UK specifically. A recent review on this topic was undertaken by Jackson et al. (2015). We first provide a synopsis of this review, before reviewing in detail more recent research published after Jackson et al. (2015). Finally, we synthesise the findings of Jackson et al. (2015) and more recent literature to provide a "state of the science" regarding the impacts of climate change on groundwater recharge and levels in the UK.

2.1.3.2 Work to 2015

Jackson et al. (2015) provides the latest review of peer-reviewed studies evaluating the impacts of climate change on groundwater recharge and levels. Following a small number of studies before 2002, between 2002 and 2015 eight studies were reported in the peer reviewed literature, covering 12 sites predominantly in the Chalk of southern and eastern England. These studies are summarised in Table 2. Each of the studies cover a relatively small area of the UK and use different methodologies for use of GCM outputs (different GCMs, emissions scenarios and downscaling techniques) and approaches to quantifying impacts and only report local findings. Consequently, Jackson et al. (2015) concluded that it was difficult to directly compare results between these studies. The key results of these studies in relation to this research are as follows:

- There is uncertainty in the direction of change of both long term average recharge and groundwater levels. Groundwater recharge and levels have been predicted to either increase or decrease depending on (1) emissions scenarios (Yusoff et al., 2002), (2) the future timeslice considered (Bloomfield et al., 2003; Herrera-Pantoja and Hiscock, 2008; Herrera-Pantoja et al., 2012) and (3) the GCMs used (Jackson et al., 2011). GCM choice appears to affect the direction of change of groundwater levels less than recharge (Jackson et al., 2011).
- Multiple studies have predicted increases in winter recharge and decreases in summer recharge (Jackson et al., 2011; Yusoff et al., 2002)
- Changes in extremes have not been reported.

To address the challenge of interpreting the results of the divergent methodologies of these previous studies, Jackson et al. (2015) also presented results from the FutureFlows and Groundwater Levels (FFGWL) project (Prudhomme et al., 2013). FFGWL applied a consistent set of climate change driving data from UKCP09 to lumped parameter groundwater models for 24 boreholes across the Great Britain for the first time.

Table 3 shows a summary of the results of the groundwater level projections produced in the FFGWL project in the 2050s under a high emissions scenario. It can be observed that different components of climate change have different levels of uncertainty. There is relatively high confidence that extreme high winter groundwater levels (February 75th percentile) will increase (20/24 models show increases), and that extreme low summer groundwater levels (September 25th percentile) will decrease (21/24 models show decreases). There is lower confidence in the direction of change of seasonal and annual median groundwater levels, although a majority of models predict decreases in summer and annual median levels and increases in winter median levels. Jackson et al. (2015) also highlight the importance of local hydrogeological conditions in propagating climate signals, and that there appears to be a greater uncertainty associated with the UKCP09 ensemble than emissions scenarios in terms of changes in groundwater levels.

Table 2 Summary or groundwater and climate change studies reviewed by Jackson et al. (2015). Reproduced from Jackson et al. (2015) and licenced under CC-BY 3.0.

No.	Study	Region	Aquifer	Emission scenarios
1	Younger et al. (2002)	Northeast (Humber Estuary)	Chalk	Equilibrium GCM simulations using fixed CO_2 concentrations
2	Yusoff et al. (2002)	East Anglia (R. Ely)	Chalk	HadCM2 1. Medium-high 2. Medium-low
3	Bloomfield et al. (2003)	1. Southwest (Exeter) 2. East Anglia (Lincolnshire) 3. Southeast (Kent)	PT Sandstone Jurassic Limestone Chalk	UKCIP98 1. Medium-high
4	Herrera-Pantoja and Hiscock (2008)	1. South (Gatwick) 2. East Anglia (Coltishall) 3. West Scotland (Paisley)	Chalk Chalk Limestone	HadCM3 and CRU weather generator (CRU-WG) 1. High
5	Holman et al. (2009b)	East Anglia (Coltishall)	Chalk	UKCIP02 and CRU-WG 1. Low 2. High
6	Clarke and Sanitwong Na Ayutthaya (2010)	Northwest (Ainsdale)	Coastal sand dunes	0.6m sea level rise UKCIP02 and CRU-WG 1. Medium-high
7	Herrera-Pantoja et al. (2011)	East Anglia (Coltishall)	Chalk	UKCIP02 1. High
8	Jackson et al. (2011)	Malborough and Berkshire Downs and southwest Chilterns	Chalk	13 CMIP4 GCMs 1. Medium-high

Table 3 Summary of changes in groundwater levels produced by Prudhomme et al. (2013) and reported by Jackson et al. (2015).

Climate change component	**Metric**	**Increasing models**	**Decreasing models**
Long term average	Annual median	9	15
Seasonality	February median	16	8
	September median	6	18
“Extremes”	February 75th percentile	20	4
	September 25th percentile	2	21

2.1.3.3 Recent studies post 2015

Since the publication of Jackson et al. (2015), six peer reviewed studies have evaluated how climate change may affect groundwater recharge and levels in the UK. These are reviewed herein.

Jimenez-Martinez et al. (2016) report predictions of changes in groundwater flooding associated with climate change in the Chalk near Brighton. A transfer function approach was used to derive recharge and groundwater level time series from UKCP09 rainfall time series. This analysis suggested a reduction in annual groundwater recharge, but with increased seasonality (increased recharge and groundwater levels in winter, decreased recharge and groundwater levels in summer). An increased frequency of high groundwater level events is predicted, with groundwater-induced flooding c. four times more frequent by 2040–2069 and around seven times more frequent by 2070–2099 in comparison to 1961–1990. An increased probability of groundwater drought events in the 2050s and 2080s is also reported.

Ascott et al. (2019) evaluated the relative impact of changes in hydraulic conductivity with depth and climate change scenarios on estimations of pumping water levels and borehole yields during droughts. By applying 20 UKCP09 climate change scenarios, 11 different hydraulic conductivity depth profiles and six constant pumping rates, it was shown that during future drought events, the hydraulic properties of the aquifer exert a more significant control on lowest pumping water levels than changes in climate. It was concluded that both changes in climate and changes in hydraulic conductivity with depth below the lowest observed pumping water level should be taken into account in future assessments of the impacts of climate change on borehole yields during drought.

Bloomfield et al. (2019) present the first empirical evidence for changes in groundwater drought associated with anthropogenic warming, in the absence of long-term changes in precipitation. Using very long groundwater level time series for two sites in England, an increasing coincidence of groundwater droughts with precipitation droughts and hot periods in the early 21st century is identified. It was inferred that the nature of groundwater droughts is changing due to increases in evapotranspiration from the capillary fringe associated with anthropogenic warming. Given the UKCP18 projections reported in section 2.1.2.1, it seems likely that increases in temperature will affect future groundwater drought characteristics.

Yawson et al. (2019) used UKCP09 data with the AquaCrop model to estimate potential groundwater recharge from barley crop fields during the spring-summer growing season across 14 UK regions. Whilst the focus on the barley crop growing season limits the utility of this paper to quantifying changes in long term average and extremes, some conclusions can be drawn regarding summer recharge. Depending on the region of the UK, spring-summer groundwater recharge under barley crops was predicted to decrease by up to 38% or increase by 41%. Reductions in recharge were greatest in southern and eastern England and under the highest emissions scenario and furthest future time slice.

Hughes et al. (2021) report the application of the UKCP09 regional ensemble projections (also used in the FFGWL project) under a medium emissions scenario to a national scale gridded potential recharge model (Mansour et al., 2018) for the British Mainland. Both transient projections and results for the 2050s and 2080s were presented, focussing on River Basin Management Districts. It was shown that there is a general increase in annual potential recharge, particularly in the 2080s associated with increases in rainfall. Seasonally, there was a consistent trend of increased potential recharge in winter (driven by increased winter rainfall), decreased recharge in summer, and a mixed pattern in autumn and spring. Potential recharge was predicted to be concentrated in a smaller number of months in winter, with the summer period of low recharge extended by one to two months. No changes in extremes were reported.

Recently, Hannaford et al. (2022) presented the data and modelling methodology used in the enhanced future Flows and Groundwater (eFLaG) project. eFLaG applies data from the 12 member ensemble of regional projections produced in UKCP18 to 54 AquiMod (Mackay et al., 2014) boreholes and the national scale recharge model reported by Mansour et al. (2018). Hannaford et al. (submitted) do not report the predictions of the impacts of climate change on groundwater levels and recharge predicted in eFLaG, but these will be reported for the case studies in Task 4 of this project.

2.1.3.4 SUMMARY

Table 4 summarises the potential changes to groundwater recharge and levels due to climate change in England reported in the literature reviewed in this research. There is limited consistency in the direction of changes in long term average groundwater recharge and levels. At the national scale, Jackson et al. (2015) report that a majority of sites show long term decreases in groundwater levels in the 2050s. In contrast, Hughes et al. (2021) report that groundwater recharge is likely to increase across river basin districts in England in the 2050s. Divergence between these two national scale studies is likely to be due to (1) the different emissions scenarios ("medium" for Hughes et al. (2021), "high" for Jackson et al. (2015)), (2) different UKCP09 products used (regional projections by Hughes et al. (2021), probabilistic projections by Jackson et al. (2015)), (3) differences in scale (gridded national scale by Hughes et al. (2021), individual points and biased towards the Chalk in Jackson et al. (2015)). Similar methodological differences also result in inconsistent predictions of the direction of changes in future long term average groundwater recharge and levels from local scale studies (Bloomfield et al., 2003; Herrera-Pantoja and Hiscock, 2008; Herrera-Pantoja et al., 2012; Jackson et al., 2011; Jimenez-Martinez et al., 2016; Yusoff et al., 2002) or those for specific crop types (Yawson et al., 2019).

There appears to be some consistency in future changes to seasonal groundwater recharge and levels. National scale studies have reported increases in both recharge (Hughes et al., 2021) and groundwater levels (Jackson et al., 2015) in winter and decreases in recharge and groundwater levels in summer. Local scale studies also agree with this finding (Jackson et al., 2011; Jimenez-Martinez et al., 2016; Yusoff et al., 2002).

There is some limited evidence for future changes in groundwater level extremes. National scale studies (Jackson et al., 2015) show increases/decreases in the magnitude of relatively high (75th percentile) winter/low (25th percentile) summer groundwater levels. Local studies have reported predictions of increased frequency of extreme high and low groundwater level events in the 2050s and 2080s (Jimenez-Martinez et al., 2016). No studies have reported predictions of changes in extremes of groundwater recharge.

Table 4 Summary of potential changes to groundwater recharge and levels due to climate change in England

Variable	Long term average	Seasonality	Extremes
Groundwater recharge	Uncertain	Increased recharge in winter, decreased recharge in summer, shorter recharge window	Not reported
Groundwater levels		Increased levels in winter, decreased levels in summer	Increases in winter high levels, decreases in summer low levels, increased frequency of extreme high and low groundwater level events

2.2 IMPACTS OF CLIMATE CHANGE ON GROUNDWATER QUALITY

2.2.1 Introduction

Over the past few decades, most studies relating to climate change and groundwater have addressed processes that affect water resources or quantity (e.g. Taylor et al., 2013 and the references therein). By contrast, relatively few studies have examined climate change effects on groundwater quality. The quality of groundwater is determined by the chemical, physical and biological characteristics of the resource so consequently quality changes are expected to respond to variations in climate and anthropogenic actions because of the influence and interaction of recharge, discharge and land use on any given aquifer system. The protection and potential improvement of groundwater quality is a high priority for the environment with a clear need to maintain human and ecosystem health.

This literature review considers a broad range of recent publications that directly pertain to groundwater quality and the impact of climate change. The review starts with some brief summary statistics about the literature that is reviewed and has been divided into sections relating to 8 key themes which have become apparent. These are: general reviews; temperature; salinity; nitrate; organic carbon; organic contaminants; microbiology; and metals.

2.2.2 Methodology and Literature Summary

This literature review is based on an assessment of scientific publications which are included in the Web of Science Core Collection. The database has been searched using all the keywords “groundwater”, “quality” and “climate change” which returns 2360 results. These results have been screened for relevance to the search terms and reduced to 44 references. References come from 2006 to 2022. Figure 9 shows how the number of papers on this topic have increased over the past 15 years.

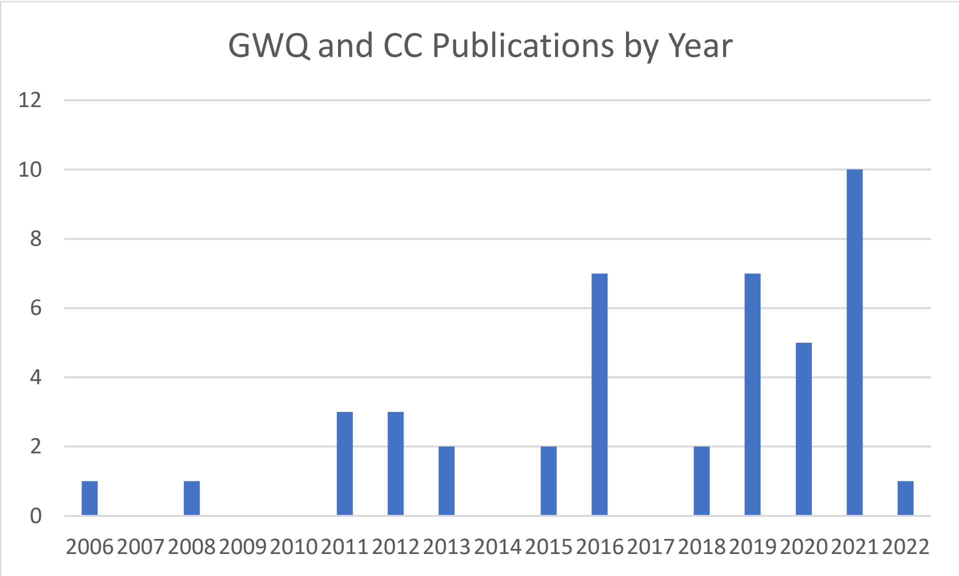

Figure 9 The evolution of publications on Groundwater Quality and Climate Change over 15 years.

The most highly cited is from a review written by Green et al (2011), attracting more than 500 hits in 10 years. In terms of publishing journals, the most popular choice for authors appears to be Science of the Total Environmental with 12 of the 44 publications coming from this journal. This is followed by Water (5) and the Journal of Hydrology (3).

The degree of global coverage for case studies of groundwater quality and climate change is shown in Figure 10. This shows there is a strong bias towards European based studies (in particular Italy and then the UK), followed by studies in Asia. No studies from South America have been identified.

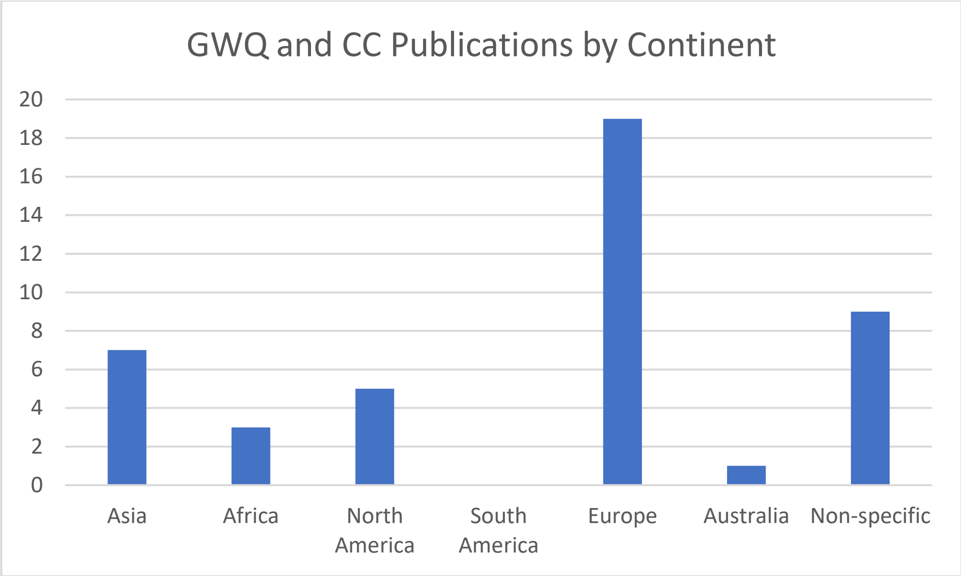

Figure 10 The distribution of Groundwater Quality and Climate Change case studies by continent.

2.2.3 Key Themes

2.2.3.1 General Reviews

Several reviews, or topic overviews have been published in the past two decades. Dragoni and Sukhija (2008) present a short review paper which largely focuses on groundwater resource issues. However, they do touch on the relationship between climatic change and groundwater quality. Here they suggest that water recharged during an arid period may have a higher concentration of salts and hence an elevated TDS, while during a wet period the converse may happen. The authors go on to mention the importance of long-term monitoring of water quality to understand the changes that are occurring.

In a comprehensive review linking climate change with groundwater, Green et al (2011) dedicate a subsection to some of the high-level impacts on water quality. The importance of salinization resulting from both sea level rise and over-pumping in coastal aquifers is noted. Similarly, from higher temperatures and increased rates of recharge it is suggested that biogeochemical reactions may be enhanced and transport of point and diffuse source contaminants may be enhanced. The importance of long-term monitoring at a wide spatial and temporal scale is also emphasized. In a book chapter linking climate change with groundwater from a book on Integrated Groundwater Management, Green (2016) discusses the same issues as in Green et al. (2011).

In North Dakota, USA, Li and Merchant (2013) investigated groundwater vulnerability to climate and land use change. The authors identified that most groundwater vulnerability modelling has been based on current hydrogeology and land use conditions but in fact groundwater vulnerability is strongly dependent on factors such depth to water table, recharge and land use conditions that all may change, possibly in an inter-related way, in response to future changes in climate and socio-economic conditions. They used a modelling framework which used three sets of models, including a modified version of DRASTIC, linked within a GIS environment to assess the impact of changing land use to growing bio-fuels under future climate scenarios. The results suggested that groundwater vulnerability would increase under a range of different scenarios.

Burri et al. (2019) consider a series of case studies to highlight increasing threats to groundwater quality in the Anthropocene. They highlight nitrogen, pesticides, pharmaceuticals, Non-aqueous phase liquids (NAPLs) and mineral extraction as major contemporary groundwater contaminants. They call for greater transdisciplinary research for a more comprehensive understanding of contamination dynamics and their effects on groundwater systems.

A study from Nigeria by Aladejana et al (2020) attempted to explore the link between climate change and groundwater quality for a shallow coastal aquifer. The study of 250 shallow wells explored the seasonal effect on redox-sensitive ions and metals. An increasing concentration of these ions and metals was observed in the dry season compared to the wet season. The paper suggests the need for strategic groundwater management policy and planning to ameliorate groundwater quality deterioration as a result of a changing climate.

Lasagna et al (2020) considered multiple ions, including NO_3, Cl, Fe and As, from shallow unconfined alluvial aquifer in the Piedmont plain and semi-confined or confined alluvial-pyroclastic aquifer from the Volturno-Regi Lagni plain in Italy. Using statistical methods, the authors found climate variables can produce sudden changes in geochemistry of shallow unconfined aquifers whereas semi-confined or confined aquifers react more slowly. Additionally, they suggest natural water quality is more affected by climate variations than anthropogenic contamination as a result of multiple environmental and anthropogenic factors.

In another Italian study examining climate variability, a series of case studies across Central Italy by Barbieri et al (2021) investigated the link between climate change and groundwater quality in regional carbonate aquifers. The study concluded that groundwater, compared to surface water, is more resilient to climate change although climate change can affect groundwater quality by reducing aquifer recharge and increasing anthropogenic pressures. A variation in groundwater chemistry associated with rainier years was identified especially for major element (in this case, Ca, Mg, Na, Cl, SO_4 and Cl) ratios. This was attributed to modified groundwater-surface water interaction times which could impact on water hardness with associated health effects.

Akhtar et al (2021) present a wide-ranging review looking at groundwater and surface water degradation that focuses on a number of ‘essential’ pollutants including pesticides, fertilisers and heavy metals, arising from anthropogenic activities and categorized these based on industrial applications, urban development and agricultural practices. They consider pollutant release due to climate change and equate this as one of the four main natural causes of contamination along with geological processes, natural disasters, and groundwater-surface water interaction.

2.2.3.2 Temperature

Temperature regulates key biological and chemical processes that affect the cycling of oxygen, carbon and other elements in soil and the underlying groundwater. As attested by the previous over-arching review papers, there are many factors that affect groundwater quality however there is very limited literature that considers temperature. Reidel (2019) analysed a large data set from a monitoring network in Germany covering over 2000 sampling sites at a density of ~1 location per 20 km^2. The author showed at the field scale, for naturally occurring temperature range between 5-20°C, temperature affects the quality of groundwater. Further, a 1°C rise in temperature is linked to a 4% decline in oxygen saturation and a pH drop of 0.02 due to CO_2 accumulation as a result of increased microbial activity and enhanced organic matter mineralisation. The results demonstrate, although not with a high degree of statistical confidence, that certain but central aspects of groundwater quality change due to warming ie pH and O_2 decrease whereas pCO_2, Mn and DOC increase. The author goes on to note this may have implications for water treatment for groundwater supplies, and some aquifers may become uninhabitable for groundwater biota.

2.2.3.3 Salinity

A number of publications have addressed this issue of coastal aquifer vulnerability to saltwater intrusion, particularly in relation to sea level rise as a result of climate change. Lyalomhe et al. (2015) used a Regional Risk Assessment (RRA) methodology based on numerical models to evaluate potential climate change-related impacts on alluvial coastal aquifers in the Esino River basin, Italy. The results showed climate change will show few impacts in this valley. Saltwater intrusion impact in future scenarios would be restricted to a few hundred meters close to the coastline and so have very limited effects on the Esino coastal aquifer.

Another study from Italy in a karstic coastal aquifer in Apulia used large-scale numerical models to understand the risk of groundwater quality degradation due to seawater intrusion (Polemio, 2016). Scenarios are run up to 2060 and show a serious worsening of groundwater salinization due to seawater intrusion as a result of decreasing rainfall and groundwater abstraction. In contrast to the study of Lyalomhe et al. (2015) results here showed that saline intrusion into the aquifer is between 2-3 km inland with concentrations greater than 5000 mg/L.

For a study from the Greek island of Crete which has seen extensive exploitation of groundwater for over 50 years and a degradation in groundwater quality from saline intrusion, Steiakakis et al. (2016) developed a regional groundwater flow model for this karstic system. The objective was to simulate the existing groundwater system and to evaluate the effects of combined impacts of groundwater exploitation and climate variability in the future. Also in contrast to the study of Lyalomhe et al. (2015) but in agreement with the study on karst by Polemio (2016) results here showed that saline intrusion into the aquifer appears to propagate at least 2.5 km far inland from the coast during summer. These studies seem to suggest that karstic systems are at greater risk of saline intrusion under a changing climate than more porous alluvial systems.

Saltwater intrusion and climate change in the Mekong Delta of southern Vietnam is discussed in a review article by Han et al. (2021). The authors identify a number of knowledge gaps for the basic characterisation of the region which need to be addressed before the problem can be fully assessed. These include application of environmental isotopes and borehole tests; intensive groundwater monitoring at multiple depths for salinity; development of coupled groundwater flow and salt transport models; and identification of the dominant factor causing saline intrusion. The authors suggest this approach so as to develop management strategies for dealing with this issue at the scale of the Delta. The approach is quite generic and could be applied to other areas of the World under such stresses.

An evaluation of how climate conditions affect groundwater quality in the Cape Flats aquifer, Cape Town, South Africa using a GIS-based modelling approach has been presented by Gintamo et al. (2021). The research found annual precipitation will increase until 2041 and then decrease until 2060. Also, an increase in precipitation was associated with an increase in the electrical conductivity (dominated by Cl) and that the groundwater electrical conductivity showed a linear positive correlation with the groundwater vulnerability index. The researchers use the WaterWorld model and recommend this for future use in a GIS environment.

Two papers address the issue of economic impacts of climate change and groundwater salinity on farmers as a result of decreasing precipitation. Akbari et al (2020) investigate the effects of temperature, precipitation and groundwater salinity changes on farmer's income risk in the Qazvin region of Iran. Taking scenarios up to 2050 the results showed that climate change and groundwater salinity have negative effects on impacts risks, in the most pessimistic scenario revenue risk will decrease by 11.2%. A study undertaken by Khan et al (2021) in Kohat, Pakistan, taking scenarios up to 2050 found a very similar revenue risk value of 11.4% as a result of groundwater salinity.

2.2.3.4 Nitrate

Assessing the impact of climate change on nitrate appears to be the most dominant water quality issue addressed by researchers. Stuart et al (2011) reported on the impact of climate change on future nitrate concentrations for UK groundwaters. Using a source-pathway-receptor framework, they identify that changes in temperature, precipitation and atmospheric carbon dioxide will influence the agricultural nitrate source term because of changes in soil processes and agricultural activity. Non-agricultural source terms such as sewer leakage in urban areas are also expected to be affected. Although the authors admit there is limited data, they suggest that without adaptation measures likely changes in nitrate leaching may show a small increase to a possible doubling of aquifer concentrations by 2100.

In the Midwestern USA, Wu et al (2012) used a Soil and Water Assessment Tool (SWAT) to evaluate impacts of increased atmospheric CO_2 and potential climate change on the water cycle and nitrogen loads in the James River Basin. They assessed the responses of soil water content, recharge and nitrate loading under a number of climate sensitivity scenarios in terms of CO_2, precipitation, and air temperature with predictions into the mid-21st century. Under the scenarios tested they saw a significant reduction in groundwater recharge and decreased nitrate load to streams, but a concomitant increase in nitrate concentration due to a decrease in streamflow.

In a case study from the Macha Oriental system in Spain, Pulido-Velazquez et al (2015) analyse the potential impacts of climate and land-use change by using an integrated modelling framework to look at groundwater quantity and quality. The authors used the agriculturally based hydrological model SWAT (Soil and Water Assessment Tool), the groundwater flow model MODFLOW, and for nitrate mass transport MT3DMS. The results show that with decreasing groundwater recharge they observe an increase in groundwater nitrate concentrations.

For the Canadian Prince Edward Island, Paradis et al (2016) assess how groundwater nitrate concentrations could evolve due to the forecasted climate change and it related potential changes in agricultural practices through using a 3-dimensional numerical groundwater flow and transport model (FEFLOW). Based on the simulations up to 2050, nitrate concentrations would increase and this was due to 2 main causes. Firstly, the progressive attainment of steady-state conditions related to present day nitrogen loadings and, secondly the increase in nitrogen loadings due to changes in agricultural practices provoked by future climatic conditions. The authors estimate that this combined effect would lead to a 25-32% increase in groundwater nitrate concentrations, although the change in groundwater recharge regime induced by climate change (with current agricultural practices) would only contribute 0-6% of that increase for various climate scenarios.

McGill et al (2019) identified the interactions between climate change, sanitation and groundwater quality can be complex. Through analysis of long-term rainfall, a study from Ramotswa in Botswana, southern Africa indicated that droughts were increasingly likely in the area. Through key informant interviews it was established that due to drought people were increasingly using pit latrines rather than flush toilets. In turn it was suggested that human waste leaching from these latrines was a likely source of nitrate pollution in the Ramotswa aquifer. The results when taken together indicate critical indirect linkages between climate change, sanitation, groundwater quality and water security in the area.

Sidiropoulos et al (2019) undertook an integrated modelling approach for a number of climate and water resources scenarios up to 2100, using MODFLOW and MT3DMS, to understand nitrate fate and transport in an over-exploited aquifer (Lake Karla) in Greece. The results indicate that groundwater nitrate concentrations are likely to increase due to a falling groundwater table from a decrease in groundwater recharge in the future water balance.

To evaluate the direct effects of climate change on the transport and accumulation of nitrate, Akbariyeh et al. (2019) developed and applied an integrated modelling framework combining climatic change, nitrate infiltration in the unsaturated zone, and groundwater level fluctuations. The study was calibrated for a site growing corn under irrigation at the field scale in Nebraska, USA. For predictions run up to 2060, future groundwater recharge was predicted to decline in the Upper Platte basin study area whereas the mass of nitrate in the saturated and unsaturated zones combined will increase from 2057-2060. The rate of nitrate accumulation was sensitive to irrigation and the depth to groundwater, and an increase in irrigation could largely accelerate the mass of nitrate.

Based on climatic predictions of temperature and precipitation for the period 2021 to 2050, Mas-Pla and Menció (2019) calculated water balances for the hydrological basins of distinct aquifer systems in the Catalonia region of NE Spain. The authors state that for this area, climate change will represent a decrease in water availability and this is a major issue in terms of controlling surface water-groundwater interactions and subsurface recharge. In turn, this leads to a general modification of nitrate in groundwater as dilution will vary. Nitrate concentration evolutions based on a mass balance model show all 6 of the hydrological systems studied display a decline in the nitrate concentration towards an estimated final equilibrium value. Given their high initial concentration, recharge under future hydrological scenarios will decrease present groundwater nitrate concentration despite a reduction of the total rainfall recharge. The authors go on to say that these counter-intuitive outcomes indicate that the hydrological dynamics of the system will naturally decrease pollution levels despite a loss of the dilution capacity, as long as input loads are kept at EU Directive levels which are lower than historic inputs.

Saleem et al (2020) assessed the impacts of future climate and land use changes on groundwater nitrate concentrations in an agricultural catchment in southern Ontario, Canada. Using a combination of an integrated hydrological model (HydroGeoSphere) and a root zone water quality model (RZWQM2) the authors developed a water flow and nitrate transport model. The selected climate change scenarios had less water availability in the future (2040-2059) and simulated future nitrate concentrations were lower than present. Using a mono-culture of corn land use produced higher nitrate concentrations compared with a corn-soybean rotation. However, the best management practice was found to be a corn-soybean-winter wheat-red clover rotation which produced significantly lower groundwater nitrate concentrations. It was suggested that such management practices should be implemented in future to reduce potential negative impacts of future climate change on groundwater quality.

Using a catchment to coast based approach, Rozemeijer et al (2021) study the eutrophic response on climatic variability in The Netherlands. They show that climate change may amplify eutrophic effects on water resources and that the complexity of climate-nutrient relations and interactions increases as you move from the from catchment to the coast. The authors also note that the effects of extreme climate conditions propagate from catchment to coast.

2.2.3.5 Organic carbon

A review by Lipczynska-Kochany (2018) looked into the effect of climate change on humic substances (HS) and the associated impacts HS can have on surface and groundwater quality. Although HS play an important role in greenhouse gas generation, climate change itself also enhances the biodegradation of HS leading to a feed-back loop. The author suggests that increased temperature and enhanced biodegradation of soil organic matter will lead to an increase in dissolved organic matter. This may impact on water treatment as the quality of freshwater sources deteriorates making drinking water production more expensive.

A global study looking at the influence of land use and climate change on variations in groundwater dissolved organic carbon (DOC) was undertaken by McDonough et al (2020). They found that the dissolved inorganic chemistry, local climate and land use explained 31% of the observed variability in groundwater DOC, whilst aquifer age (not groundwater age) explained an additional 16%. The authors identify a 19% increase in groundwater DOC associated with urban land cover. Major increases in groundwater DOC following changes in precipitation and temperature are predicted, although the relationship is quite complex. For example, the model indicated an overall increase in groundwater DOC of ~3.5% for every 1°C rise in average air temperature in the wettest quarter of the year, but an ~9% decrease in groundwater DOC for every 1°C increase in temperatures in the warmest quarter of the year. They go on to conclude that climate change and conversion of natural or agricultural areas to urban will adversely impact groundwater quality and similarly to the conclusion of Lipczynska-Kochany (2018) also increase water treatment costs.

Bank filtration (BF) is a well-established natural water quality treatment approach where surface water is infiltrated to an aquifer through river or lake banks. Sprenger et al (2011) reviewed the vulnerability of bank filtration systems to climate change based on hypothetical 'drought' and 'flood' climate scenarios. The study suggested that only BF systems comprising an oxic to anoxic redox sequence ensure maximum removal efficiency. Droughts are found to promote aerobic conditions during movement through the bank, while flood events can drastically shorten travel time and cause breakthrough of a number of potential contaminants as well as DOC. The authors conclude BF is vulnerable to climate change although anthropogenic impacts are at least as important.

2.2.3.6 Organic contaminants

The first review of climate change and organic contaminants in groundwater was undertaken by Bloomfield et al (2006) who looked at the fate and behaviour of pesticides in surface and groundwaters from the UK. The authors adopted a source-pathway-receptor approach, with climate sensitivities impacting on the pesticide source term. The main climate drivers for changing pesticide fate and behaviour were considered to be changes in rainfall seasonality and intensity along with increased temperatures. The authors concluded that the long-term, indirect impacts, such as shifts in land-use driven by climate change, may have a more significant effect on pesticides in surface and groundwaters than the direct impacts of climate change on fate and transport of pesticides.

In a study of extreme events analogous to climate change, Ascott et al. (2016) examined how a range of (mostly) organic compounds impacted on groundwater quality in a riverbank filtration scheme following extreme flooding during the winter. During the inundation event, riverbank filtrate water quality became dominated by >140% increase in DOC baseline values and a ten-fold increase in micro-organic contaminants. A rapid recovery in water quality was observed, with most floodwater impacts lasting 2-3 weeks after the flooding event and a return to normal groundwater conditions within 6 weeks. It was noted that in this case study, increased abstraction rates and a high transmissivity aquifer facilitate rapid water quality recoveries, with longer term trends controlled by river and groundwater quality. Temporary reductions in abstraction rates appear to slow water quality recoveries. It was therefore recommended that flexible operating regimes are needed for shallow aquifer riverbank filtration systems to be resilient to future inundation events.

A recent review paper by Cavelan et al. (2022) summarises and discusses the effect of climate change on Light Non-Aqueous Phase Liquids (LNAPLs). An increase in temperature, changes in precipitation and groundwater level fluctuations are known to influence both the mobility and release of LNAPLs into air and groundwater. Based on available literature, the authors conclude that a higher amplitude of groundwater table variations and higher temperatures will likely increase LNAPL biodegradation, mobility and spreading, favouring release of more LNAPL compounds to groundwater but decreasing LNAPL mass and longevity. Outcomes will however vary across arid, cold or humid coastal environments where different effects of climate change are expected.

2.2.3.7 Microbiology

Climate change induces a shift in the dynamics in groundwaters and aquifers which are typically nutrient poor but home to a diverse and specialised microbiome and fauna. Retter et al (2021) review the potential threats to groundwater ecosystems and specifically the impacts on microorganisms. The authors identify that groundwater microbes (e.g. bacteria, archaea, fungi and protozoa) are particularly susceptible to increases in temperature, changes in organic matter. Higher temperatures will speed up microbial mediated processes including the turnover of organic matter and redox processes. Increases of 1-2°C are unlikely to have much impact in the short term, however, on a long term basis even slightly elevated rates of carbon turnover may exhaust carbon and nutrient pools if not replenished in time. A change in the availability of key nutrients can have dramatic effects on microbial growth and system diversity, in some cases leading to predator-prey interactions within the microbial food web. Changes in dissolved organic matter content arising from climate change can cause a reorganisation of microbial communities in a compositional and functional manner.

In a study by Dwyer et al (2021) the microbial groundwater quality in private domestic wells following the 2018 European drought was assessed. Despite an absence of recharge or infiltration for microbial transport, the researchers found surprisingly high concentrations of *E. coli* during the drought conditions. This was attributed to a switching of contaminant pathways from a combined mainly regional and small local source under normal conditions, to just local sources (e.g. septic systems and local soil) during drought, possibly due to less dilution and legacy microbial sources.

A strong climatic influence on microbial populations was also identified by Sorensen et al (2021). Whilst undertaking sampling during the summer of 2020 when groundwater levels were failing, a period of extreme rainfall resulted in a significant, sporadic faecal contamination from *E. coli* of public water sources.

2.2.3.8 Metals

The mobility of metals which have been retained in the shallow sub-surface/soil zone or are an integral part of the aquifer matrix material can be significantly influenced by climate change. Visser et al (2012) examined the effects of future projected climate change on the hydrology and leaching of heavy metal contamination in a lowland catchment in the Netherlands. They used a quasi-2D unsaturated zone Soil Water Atmosphere Plant model with 100-year simulated daily time series of precipitation and potential evaporation. The future climate scenarios project higher evapotranspiration and lower groundwater levels resulting in lower concentrations of Cd and Zn in receiving waters.

In a study to understand how groundwater quality was impacted following a drought, Darling et al. (2012) monitored hydrochemical, stable isotope and age indicators in springs, boreholes and surface waters from the Pang and Lambourn Chalk catchments in southern England during a major recovery between 2006-2008. All the water bodies showed little change in their water quality over the monitoring period. The authors concluded that despite potential extremes in rainfall and temperature and the resulting water level changes of greater amplitude, the buffering effect of the Chalk aquifer should protect the quality of Chalk springs and streams.

A review paper by Anawar (2013) summarises the impacts of climate change on geochemical reactions, acid mine drainage (AMD) generation, and water quality in semi-arid/arid mining environments. Water scarcity is frequently found in semi-arid/arid regions and most of the mining activity occurs in remote areas situated far away from urban areas leading to very poor water supply systems. In these mining areas, potential sources of toxic elements are the sulfidic minerals that are oxidised in contact with oxygen and water. In strongly-evaporative environments of semi-arid/arid regions, increased rainfall intensity as a result of climate change has a significant influence on pyrite/sulfidic minerals oxidation leading to release in acidity, sulfate, toxic elements and other ions into groundwater. By contrast, if climate change leads to reduced rainfall in these areas then oxidation of sulphide materials and AMD generation will decrease.

In another study Chen et al (2016) look at the influence of a changing climate on the mobility of heavy metals in groundwater and how this relates to their anthropogenic source or ‘human activity mode’ e.g. mining, waste disposal, agriculture. The authors found that human activity mode significantly influences Cu and Mn but not Zn, Fe, Pb and Cd concentrations. By contrast annual mean temperature only significantly influences Cu and Pb concentrations, whereas annual precipitation only significantly affects Fe, Cu and Mn concentrations.

In a study from the Indo-Gangetic Basin, MacDonald et al (2016) used high-resolution in situ records of groundwater levels, abstraction and groundwater quality to understand the long-term trends in groundwater as a result of changing climate and human influence. Groundwater quality issues are dominated by salinity and high arsenic concentrations which restrict 60% of access to potable water from the aquifer. The authors conclude that sustainable groundwater supplies are constrained more by this contamination than by depletion.

From a study investigating the relationship between climate variables and groundwater quality along the KT Boundary, Thivya et al (2018) monitored a groundwater level and geochemical parameters over six years. The authors linked higher intensity rainfall to an increase in electrical conductivity.

The impact of climate variation and human activities on groundwater quality has been studied in the semi-arid region of the Mahabad aquifer in northwest Iran by Khalaj et al. (2019). The authors report that crop pattern change, through groundwater exploitation and irrigation return flow have adversely impacted groundwater quality. This is demonstrated by an increase in groundwater electrical conductivity over the past 40 years.

Through examining pre- and post-monsoon hydrochemical data, Rani et al (2021) have looked at climate driven changes in water chemistry from springs situated in the Kumaun Himalaya in India. They found that the water chemistry pre-monsoon to be a mixture of Ca-Mg-HCO_3 and Ca-Mg-Cl type waters. In post monsoon this shifted to just a Ca-Mg-HCO_3 type water indicating freshening and improving the quality for drinking. With higher intensity rainfall from climate change, this could mean a greater development of the aquifer system as a potable source.

2.2.4 Summary

The papers reviewed here all fall into one of three major categories i) a local or regional scale dataset linked to climate-based inputs through one or a number of models to produce projections of future water quality ii) using seasonal changes or extremes to examine what these mean for groundwater quality and by inference suggest this is what will also happen under a changing climate and iii) A more heuristic approach based on likely climate scenarios to infer in a more qualitative way how future water quality parameters will evolve. All of these approaches have varying degrees of uncertainty which varies with spatial and temporal aspects. Based on this literature the general trend is a decrease in groundwater quality over the next 50-80 years although some studies suggest this decline is not inevitable and indeed some parameters may even improve after initially getting worse. What is abundantly clear is the considerable uncertainty associated with these trends. Based on expert judgement, Table 5 identifies a confidence level for the parameters assessed against a number of climate change variables. Although in some cases we may have a high degree of confidence in a causal relationship, the magnitude or even long-term direction of change of that water quality parameter is far from clear. For a number of the other climate drivers and quality variables our understanding of impacts is near non-existent and highly speculative at best. To better understand impacts of climate change on groundwater quality, it is clear that the need for monitoring networks is essential if appropriate management practices are to be developed and implemented.

	Sea level rise	Increased Temperature	Changing Rainfall	Climate Change Induced Land Use Change
Shallow groundwater temperature	-	High	Low	Low
Salinity	High (coastal aquifers only)	Moderate	High	High
Nitrate	-	Moderate	High	High
Dissolved Organic Carbon	-	Moderate	High	High
Organic Contaminants	-	Moderate	High	Moderate
Microbiology	-	Moderate	Low	Low
Metals	-	Moderate	Moderate	Low

Table 5 Level of confidence in understanding of the influence of climate change variables on groundwater quality parameters

3 Stakeholder perspectives on groundwater quality into the future

As a part of this project on Wednesday 9th February 2022, the BGS and EA held a workshop on "Groundwater Quality into the Future". The purpose of the workshop was to gather input from both Environment Agency and external stakeholders regarding what the key issues are related to future groundwater quality, and what are the priorities for adaptation, management and research. The workshop findings are detailed in full in Ascott (2022).

In this report, we provide a brief overview of the perspectives raised in the workshop. These are organised into a number of cross-cutting themes and key focus areas for future work as prioritised by workshop delegates, detailed herein. Together with the finding of the international literature review (section 2.2), these perspectives from key stakeholders in England inform the recommendations in section 6.

3.1 CROSS-CUTTING THEMES

3.1.1 Uncertainty

Uncertainty and lack of knowledge is a significant limitation to adaptation, management and mitigation of the impacts of climate change on groundwater quality, particularly over longer timescales (> 50 years). There is uncertainty in both changes in drivers (e.g. climate change, population growth and socioeconomic change) and pressures (e.g. changes in recharge processes, land use, groundwater abstraction) controlling groundwater quality. This is compounded by uncertainty in the hydrogeological system response to a given change in pressure and because multiple competing drivers and pressures will control groundwater quality in the future. Consequently, detection and attribution of changes in future groundwater quality associated with individual drivers and pressures is likely to be highly challenging. There is a need for further research to build the evidence base to support decision making regarding appropriate mitigation and adaptation measures.

3.1.2 Holistic, systems approaches

At present siloed approaches to both the science and management of the terrestrial water cycle are common. This includes divisions between both different components of the hydrological cycle (e.g. climate, soil water, groundwater, surface water) and how they are managed to address different issues (e.g. flooding, resources, quality). The interconnectivity between groundwater resources and quality, and surface water and groundwater necessitates the development of integrated, systems based approaches to both the science and management of groundwater in the terrestrial water cycle. Whilst there is now a growing body of scientific literature in the UK addressing this need (e.g. Hutchins et al. (2018); Mortazavi-Naeini et al. (2019); Dallison et al. (2022), in general such approaches are scant, particularly outside of research publications. This gap was also identified from a drought perspective by Ascott et al. (2021).

Such joined up approaches should be founded on a vision of what, in the face of change, good groundwater management in the terrestrial water cycle looks like in 50-80 years. Approaches should build a culture of wider inclusion and interdisciplinarity with greater collaborative working and partnerships from the outset, including knowledge and data exchange.

3.1.3 Monitoring

Environment Agency groundwater quality monitoring has decreased over the years associated with reductions in funding and resource availability. There is an ongoing requirement for long term monitoring to characterise groundwater quality before (baseline), during and after changes in pressures (e.g. changing land use, recharge, abstraction). If future financial constraints limit the extent of monitoring undertaken this may limit our ability to (1) characterise the impact of environmental change on groundwater quality and (2) predict future groundwater quality and assess the effectiveness of regulation and interventions.

Further, networks should be re-evaluated to ensure that the monitoring networks of the future are fit-for-purpose in a changing, uncertain world. This includes evaluation of whether monitoring frequencies are adequate given the changing nature of extreme events, monitoring for a growing range of emerging substances and changing pollutant sources, and use of innovative monitoring techniques (e.g. application of citizen science and remote sensing datasets). Section 5 explores this further.

3.1.4 Education

Despite decades of previous efforts (e.g. the UK Groundwater Forum) to raise awareness around groundwater, the level of public understanding remains poor. A new generation of funded engagement activities with the public and other stakeholders (e.g. Local Planning Authorities, developers, senior decisionmakers, parliamentarians) that "makes the invisible visible" is required. These activities should cut across political and financial agendas, and consider adopting new approaches (e.g. social media) to communicate the value of groundwater, using language that is simple to understand and engaging. Improved education of children about groundwater in the terrestrial water cycle in schools is required.

Numbers of undergraduate earth science students at universities are reported to have decreased substantially. These courses feed in to masters level training opportunities in hydrogeology; the graduates of which form a significant proportion of groundwater professionals in the UK. Further engagement to highlight the value of earth sciences to prospective undergraduate students is required to safeguard future generations of qualified groundwater professionals in the UK.

3.2 KEY FOCUS AREAS

3.2.1 Nutrients

Nutrients (nitrogen (N), phosphorus (P) and carbon (C)) remain an issue for groundwater quality, with significant costs associated with treatment and blending for public water supply as well as surface water eutrophication. For all nutrient sources there is uncertainty as to how these may change in the future as a function of climate and socioeconomic changes. Potential direct impacts of climate change on nutrient fluxes include changing rainfall patterns resulting in mobilisation from nutrient-saturated soils and changes in recharge pathways. Indirect changes in nutrient sources and pathways are likely to be significant, such as changes in population, dietary habits and energy demands resulting in growing of different crops.

When considering nutrient concentrations at receptors, there is uncertainty regarding the relative contribution of legacy sources and current sources (e.g. fertilizer applications). Furthermore, legacy C and P are generally less well understood than N, and different management approaches are likely to be required for different nutrients.

3.2.2 Emerging substances

The emergence of new individual and mixtures of organic contaminants (e.g. Per- and polyfluoroalkyl substances (PFAS), pharmaceuticals) in groundwater is of concern, particularly where these are highly persistent and bioaccumulative. There is considerable uncertainty regarding the sources, fate and transport of these contaminants both at present and over the next century. Changing rainfall patterns may result in increases in winter runoff and mobilisation of contaminants. Hotter temperatures may result in increased wildfires and associated use of foams and fire suppressants as well as use of PFAS-containing ground source cooling schemes. Further, our understanding of what concentrations constitute a significant risk in terms of ecotoxicology and human health is poor.

3.2.3 Changing rainfall characteristics

Changing rainfall patterns are predicted under climate change, resulting in wetter winters, drier summers and more extreme rainfall events. These changes are likely to affect groundwater quality through: (1) increased leaching of contaminants and pathogens in winter, (2) changing runoff and recharge characteristics, (3) flushing of contaminants in the unsaturated zone. These possible changes are associated with increases in winter rainfall and increasing extreme rainfall events. Under drought scenarios, less water is available for dilution and so more extreme droughts may result in higher concentrations of contaminants in aquifers. These possible changes may result in increased contaminant concentrations at public water supplies and other receptors. Changes in land use to increase storage and slow runoff, and flexible abstraction regimes (e.g. increasing abstraction when groundwater levels are high) may help manage this.

3.2.4 Changing temperature

Groundwater temperatures are likely to change over the next century as a function of two processes. In shallow groundwater systems, groundwater temperatures are anticipated to rise by c. 1-2 degrees associated with increases in air temperature due to anthropogenic warming. The development of ground source heating and cooling schemes associated with decarbonisation is also likely to be affecting groundwater temperatures, with potentially much greater changes in temperature than those associated with anthropogenic warming.

Changes in temperature may affect contaminant mobility through greater dissolution, and contaminant degradation through greater microbial activity. Small changes in groundwater temperature of the order of 1 - 2 degrees may only have a limited impact on chemical and microbiological processes in groundwater. Larger changes in temperature associated with ground source heating schemes may have an impact but this is largely unknown at present. At present there is no national groundwater temperature monitoring network to assess changes in groundwater temperature.

3.2.5 Groundwater rebound

Groundwater level rebound is likely to occur in some areas in the future associated with the requirement to reduce overabstraction. Groundwater level recoveries have the potential to affect groundwater quality. There is uncertainty associated with the extent to which groundwater levels will rebound in response to a given reduction in abstraction, particularly given the vertical heterogeneity in hydraulic properties of different strata, and potential interaction with boundary conditions such as the land surface and the sewer network. The impact of these changes on groundwater quality is a further source of uncertainty. Increasing groundwater levels will result in increased saturation of the unsaturated zone and mobilisation of contaminants. There is limited understanding of the vertical profile of contaminants in the unsaturated zone associated with historic loadings from the land surface. This means for any given change in abstraction there is limited predictive power in what changes in groundwater quality will be.

3.2.6 Urban development and construction

The need for housing for our growing population is exerting, and will continue to exert, a significant pressure on groundwater. There is increased development on both brownfield and greenfield sites which is resulting in increased nutrient loading associated with both direct discharges to ground and loadings to sewage treatment works. Increased loads at sewage treatment works may result in increased need for sludge spreading, on a decreasing area of agricultural land due to urbanisation. Targets are set for house building that do not consider groundwater protection, and if poor practices are used in attempts to reduce costs then there is a risk of contaminants leaching to groundwater. Whilst regulations are clearly set out and guidance is in place, there is a pressure to meet growth targets,limited resource for regulatory enforcement, and limited knowledge regarding groundwater issues in developers and Local Planning Authorities. Large construction projects (e.g. HS2) are also anticipated to continue to exert pressure on groundwater through deep excavations.

3.2.7 Changing salinity

Changes in salinity in groundwater are likely to occur due to minewater processes and saline intrusion in coastal aquifers. Both of these are likely to change in the future. Mobilisation of minewater contaminants (including increases in salinity) may occur due to groundwater level rebound following stopping pumping and increases in groundwater levels due to climate-induced changes in recharge. Management of saline waters from legacy mine workings in an environmentally sustainable manner remains an ongoing challenge. Increasing sea levels will change the extent of saline intrusion in coastal aquifers. This is likely to be exacerbated during drought periods when driving groundwater heads are lower. This may result in a need to reduce pumping rates at some locations (perhaps seasonally) to ensure that salinity at public water supply abstractions does not increase.

3.2.8 Groundwater ecosystems

At present there is a very limited understanding of the distribution, species composition and functioning of groundwater ecosystems. We do not know the current level of biodiversity in groundwater ecosystems and the ecosystem services that microbial communities provide. To assess the impact of future changes in groundwater ecology we need a much better understanding of the current baseline functioning of these systems. A key risk is the potential loss of unique ecosystems and the ecosystem services they provide. These services may include breakdown of contaminants and dissolved organic matter.

4 Case Studies

4.1 METHODOLOGY

4.1.1 Case study selection and hydrogeological and water quality conceptualisation

Five case study areas (Figure 11) were chosen in collaboration with the Environment Agency project steering group to illustrate different water quality issues and contaminants. They represent a range of different geologies (Chalk, Permo-Triassic sandstone and Carboniferous Limestone), geographical position (north, south, inland, coastal) and land uses (rural, urban). Each case study includes, or is close to, one or more observation borehole used in the eFLaG project. These boreholes were used to evaluate potential impacts of climate change

For each case study area we detail the hydrogeological conceptualisation and groundwater quality issues of concern based on BGS geological and hydrogeological maps and data, and water quality data from the baseline (BGS/EA) and water quality (EA/Entec/ESI) reports supplemented by more recent data from the Environment Agency's water quality database (https://environment.data.gov.uk/water-quality/). The Water Framework Directive classification results and objectives are summarised at the end of the chapter in Table 12. The keys for all the accompanying geological maps only include the units within the case study areas.

4.1.2 Extraction and processing and presentation of climate data

For each of the five case study areas, we extracted climatological and hydrological data from the eFLaG dataset (Hannaford et al. 2022), BGS GeoCoast dataset and directly from the UKCP18 user interface (for temperature). The eFLaG data consists of transient time series to 2080 derived from the UKCP18 regional projections under the RCP8.5 scenario. eFLaG rainfall, and evapotranspiration are 1 km gridded datasets which cover Great Britain. They were spatially averaged for each case study area. The recharge data (generated from the BGS national groundwater potential recharge model) cover 558 groundwater bodies over Great Britain, whilst groundwater levels are for individual observation boreholes at 54 sites. GeoCoast is a package of datasets to support coastal management and adaptation which includes sea level rise and coastal inundation. With the exception of sea level rise, we extracted change metrics that capture changes in average conditions, seasonality and high/low percentiles for each variable as detailed in Table 6. It should be noted that the GeoCoast and eFLaG datasets use different emissions scenarios (GeoCoast RCP4.5, eFLaG RCP8.5) which should be considered when evaluating the results from these datasets. It should also be noted that whilst we report changes in high and low percentiles based on transient eFLaG data, these do not represent changes in extremes due to the small sample size of data in each future timeslice. For example, in eFLaG for a given 10 year timeslice there are only 10 years of weather data, so calculation of a 1% annual exceedance probability event is of limited meaning based on this data. These results should therefore be considered to be indicative illustrations of possible changes in high/low percentile events. Change metrics were calculated based on the eFLaG baseline (BL, 1989-2018), near future (NF, 2020-2049) and far future (FF, 2050-2079) periods. We also presented transient time series to show the evolution of change over the 21st century.

4.1.3 Assessment of the possible relationships between climate change and groundwater quality

Having detailed the groundwater quality issues and presented impacts of climate change on the meteorology and hydrogeology, for each case study we then made a qualitative evaluation of the potential impacts of climate change on groundwater quality.

Table 6 Details of variables, data sources and change metrics calculated

Variable	Data source	Change metrics
Temperature	UKCP18 UI	Long term average, seasonality, 5th and 95th percentile
Rainfall	eFLaG	
Evapotranspiration		
Recharge		
Groundwater levels		
Sea level rise	GeoCoast	Long term average

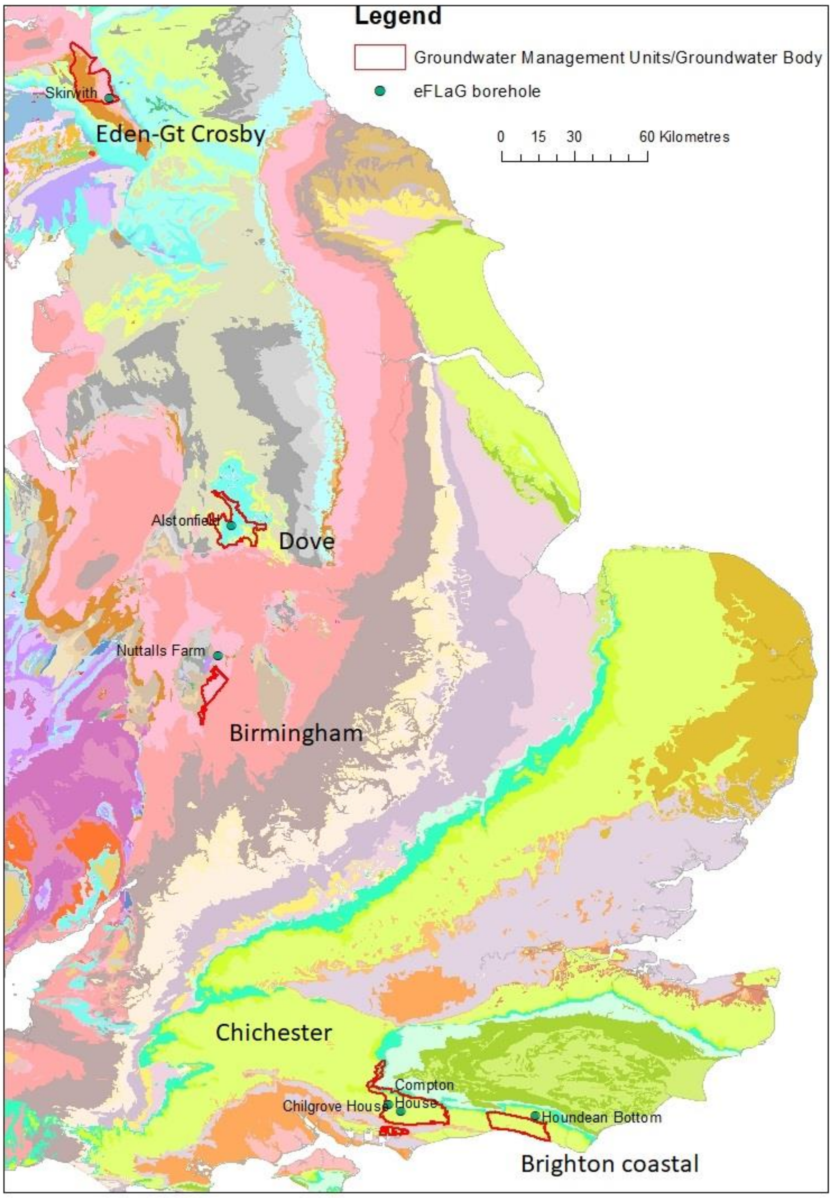

Figure 11 Locations of case study areas. Geology is BGS 1:625k bedrock data. Contains public sector information licensed under the Open Government Licence v3.0.

4.2 CASE STUDY 1 - COASTAL BRIGHTON CHALK BLOCK

4.2.1 Setting

This case study area concerns the part of the Brighton Chalk Block, Water Framework Directive Groundwater Body that is connected to the sea and lies within 6 km of the coast. It is located along the south coast of England between the southward draining Rivers Adur (west) and Ouse (east), both of which are tidal to beyond the northern extent of the study area. The area comprises part of the South Downs, rising from sea level in the south to over 200 m at Truleigh Hill in the north-west. The coastal strip is predominantly urban (Old Shoreham, Southwick, Portslade, Brighton and Hove and Newhaven), and Brighton extends north for over 5 km to cover nearly the full north-south extent of the study area. Inland the land use is a mixture of arable farming and grassland typical of Chalk downland, with pasture on the steep scarp slope. Soils tend to be thin and stony. The mean annual rainfall varies from 700 mm along the coast to over 900 mm over the higher ground, and recharge over the whole Brighton Chalk Block (not just the coastal 6 km) is quoted as 477 mm (Jones and Robins, 1999).

4.2.2 Geology

The area is underlain by Chalk bedrock, locally overlain by small outcrops of Lambeth Group (with a single small area of London Clay Formation at Rushy Hill, Peacehaven) towards the coast plain; except the north-west and north-east corners that are underlain by the Gault and Upper Greensand formations, and Gault Formation, respectively (Table 7, Figure 12).

Table 7 Stratigraphy of Coastal Brighton Chalk Block Groundwater Body

Group	Formation	Description
Thames Group	London Clay Formation	Up to 15 m, silty, grey clay with sand in places, basal well-rounded pebble bed
Lambeth Group	Woolwich and Reading Formations	22-35 m silty clay with thin lignite beds and shell beds in places. Basal glauconitic pebbly sand.
White Chalk Subgroup	Culver Chalk Formation (Tarrant Chalk Member)	30-40 m white chalk with seams of large nodular and tabular flints
	Newhaven Chalk Formation	50-75 m white, soft chalk with many marl seams and some nodular flints
	Seaford Chalk Formation	60-80 m pure white, soft to firm chalk with regular seams of nodular and several semi-tabular flints
	Lewes Nodular Chalk Formation	45-60 m off-white, hard, nodular chalk with regular seams of large nodular flints
	New Pit Chalk Formation	40-50 m white, massively bedded, soft to firm chalk with some flints in upper part
	Holywell Nodular Chalk Formation	25-35 m white, very shelly chalk. Melbourn Rock Member (hard, nodular,

		shell-free chalk) and Plenus Marls Member (grey marls and soft chalk) at base
Grey Chalk Subgroup	Zig Zag Chalk Formation	45-75 m white and grey chalk with thin limestone/marl couplets at base
	West Melbury Marly Chalk Formation	30-35 m pale-grey, marly chalk with fossiliferous limestones. Glauconitic Marl Member olive green, glauconitic sandstone) at base
Selborne Group	Upper Greensand Formation	0-36 m siltstone to fine-grained bioturbated sandstone
	Gault Formation	65-105 m stiff, shelly, glauconitic, micaceous mudstone with seams of phosphatic nodules in part

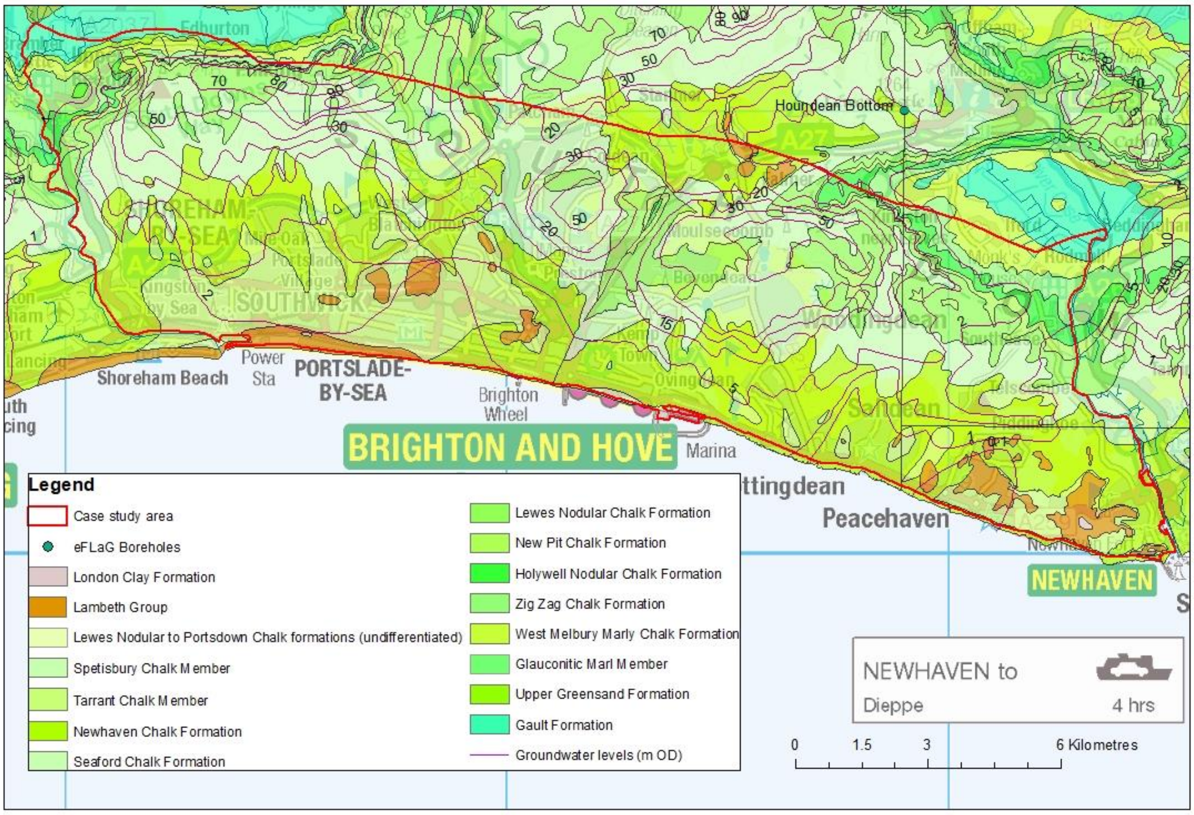

Figure 12 Bedrock geology and water levels (September 1993) in coastal Brighton Chalk Block Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

The bedrock is locally overlain by superficial deposits (Figure 13). These consist of alluvium (silty clay, peaty pebbly and shelly in part) along the Adur and Ouse valleys, with limited associated river terrace deposits (sandy gravel and pebbly sand). Head (sandy and silty clay, chalky and pebbly in part) is present along the dry valleys and clay-with-flints (silty clay with angular and well-rounded flints) on the interfluves. From Old Shoreham eastwards to Brighton, beach deposits (interbedded sand) are present along the coast with head (sandy and silty clay, pebbly in part) and brickearth (clayey, sandy silt) further inland.

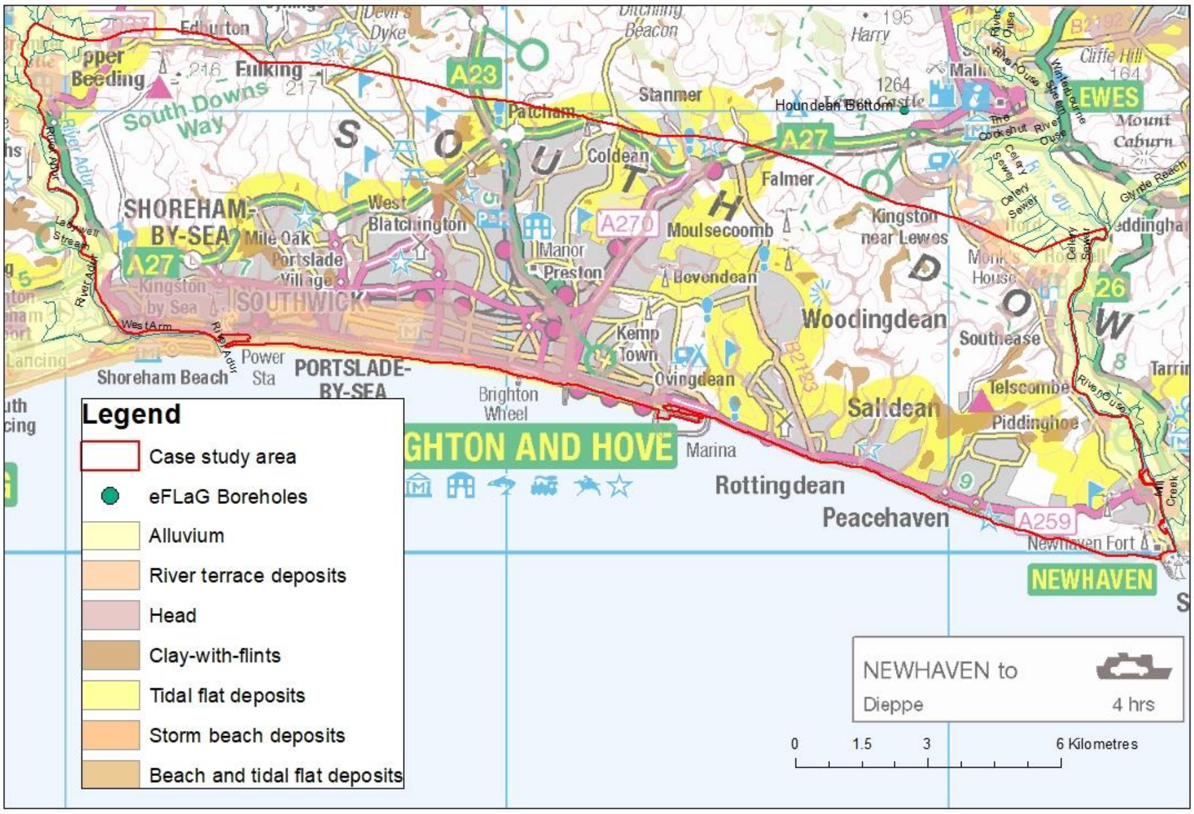

Figure 13 Superficial deposits in coastal Brighton Chalk Block Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

4.2.3 Hydrogeology

The Upper Greensand thins east of the River Adur and at outcrop rarely yields more than a few m^3/d. It is separated from the main Chalk aquifer by the low permeability ‘Chalk Marl’ (West Melbury Marly Chalk) that forms an aquitard (Young and Lake, 1988). The Chalk is a microporous limestone comprising coccolith debris and has dual porosity with a high intergranular porosity (with mean values of 38.8±5.8% for the upper part of the White Chalk, 28.4±4.2% for the lower part of White Chalk and 22.9±7.7% for the Grey Chalk (Allen et al, 1997)) but is not readily drained due to the small size of the pore throats. Hence it has a low matrix permeability and a high fracture permeability with water moving predominantly through a network of fractures that can be solution enhanced and are generally better developed in the zone of water table fluctuation (Allen et al, 1997). Solution features such as sinkholes and dolines develop particularly at contacts between Chalk and less permeable formations, such as the Palaeogene and clay-with flints, as acidic runoff dissolves Chalk. Groundwater flow is affected by the presence of flint bands, marl seams and hard bands that may act as inception horizons for dissolution. The Chalk transmits water less readily at interfluve locations than it does in valley localities due to the presence of fewer fractures. The Chalk also generally transmits water less readily with depth as fractures become smaller and less common, although occasional hard limestone bands such as the Melbourn Rock can have well-developed fracture systems. Groundwater flow is generally southwards in the west of the groundwater body, and spring discharges have been identified along the coastal margin at Hove, Brighton and Saltdean, especially near the wave-cut platform east of Brighton (Davies, 1973; Brereton and Downing, 1975). In the east it is more complex as flow is diverted east-north-east along the highly transmissive Falmer dry valley towards Lewes, defined by the topographic highs south of Falmer and north of Brighton. An anticline south of the dry valley exposes the less transmissive Grey Chalk. Therefore, transmissivity is related to topography (with the highest values in valleys and lowest under interfluves), lithology, structural features and proximity to Paleogene cover (with a concentration of chemically aggressive runoff). Transmissivity values from Allen at al. (1997) vary from 75 to 390 m^2/d.

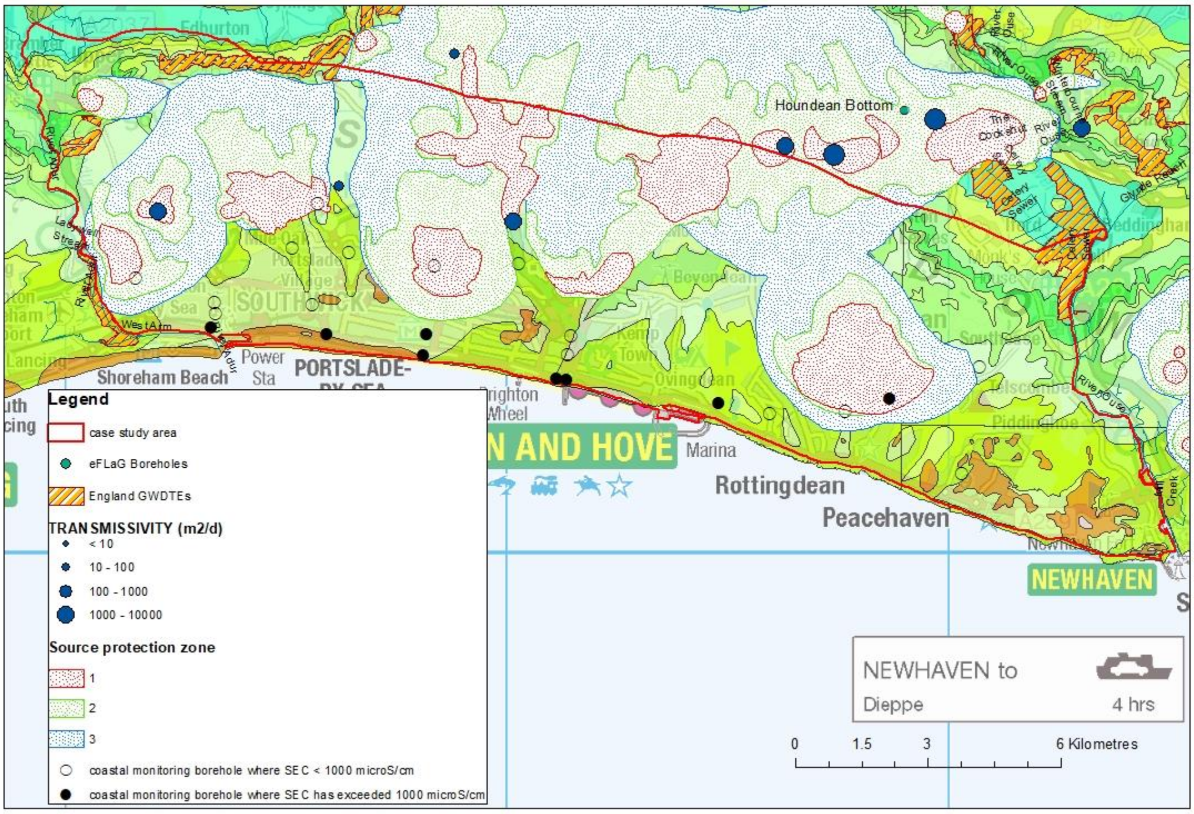

Figure 14 Various hydrogeological parameters in coastal Brighton Chalk Block Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0. Source protection zones © Environment Agency copyright and/or database right 2016.

Monkhouse and Fleet (1975) showed that along the coast little groundwater flow occurs at depths of more than 100-140 m below the water table, relating to base levels caused by changes in sea level during the Pleistocene creating active fractures at many different depths (Allen et al, 1997).

There are 10 public water supply sources within the area. The source protection zones for three other sources located outside the study area, extend into the area (Figure 14). Abstraction from the aquifer was managed (to limit saline intrusion, see 4.2.5) by pumping from the coastal pumping stations in winter to maximise interception of seaward outflow from the aquifer whilst reducing abstraction from the inland sources to allow recovery of water levels and aquifer storage. Whilst in summer once chloride levels start to rise in the coastal sources, the inland sites were preferentially pumped (Jones and Robins, 1999). However it is believed that the groundwater abstractions may not now be so diligently operated in this way (Frances Sinclair, *pers. comm.*)

4.2.4 Groundwater dependent terrestrial ecosystems

The area includes parts of two groundwater dependent terrestrial ecosystem SSSI: Beeding Hill to Newtimber Hill (Chalk downland) and Lewes Brooks (River Ouse floodplain with salinity varying from brackish to spring-fed).

4.2.5 Groundwater quality

The high intergranular porosity provides a very high surface area enhancing the potential for reaction of the chalk surfaces. In addition, the matrix, having a permeability several orders of magnitude lower than the fractures, acts as a reservoir for older water and may affect groundwater quality by slow exchange with the fracture water. The White Chalk Subgroup comprises very pure carbonates. The most important non-carbonate minerals are quartz, montmorillonite, white mica and apatite, with kaolinite only found in the Grey Chalk Subgroup. These have a disproportionate effect on water chemistry releasing small amounts of magnesium, manganese, strontium and iron during congruent (rapid) and incongruent (slower) reactions. Rainfall acidity is quickly neutralised by reaction with chalk sediment and the reactivity of the soil water is greatly enhanced by the solubility of the carbon dioxide produced biogenically in the soil zone. Congruent dissolution of the chalk occurs during infiltration through the unsaturated zone and saturation with calcite is typically attained within a few metres of the surface (Edmunds et al, 1992); below this depth chalk dissolution is greatly diminished. However, a small percentage of by-pass flow through open fractures may result in calcite-undersaturated water being transported to deeper parts of the unsaturated zone with capacity for fissure enlargement, as demonstrated by tritium studies (Foster and Smith-Carrington, 1980).

Apart from coastal saline intrusion, the groundwater is generally of good quality but with a constant threat of pollution mainly from agrochemicals. The water is of calcium-bicarbonate type, with a specific electrical conductance typically of 550-700 µS/cm. Calcium concentrations are generally in the range 85-110 mg/l, magnesium less than 5 mg/l and sodium 15-25 mg/l. Bicarbonate is generally in the range 200-270 mg/l, sulphate 10-30 mg/l and chloride 25-40 mg/l (Edmunds and Brewerton, 1997; Jones and Robins, 1999; Entec, 2008a). Fluoride concentrations are generally less than 0.1 mg/l (Edmunds and Brewerton, 1997).

Edmunds and Brewerton (1997) described the Brighton Block Chalk groundwaters as aerobic (maintaining low dissolved iron (<5 µg/l that precipitates as oxides on fracture surfaces), and persistence of NO_3) and of low mineralisation, less evolved than other Chalk groundwaters in southern England (such as Berkshire) reflecting atmospheric inputs as well as reactions at shallow depth between water and Chalk sediment. The waters can be described in terms of initial rapid water-rock interaction; slight modification of composition by incongruent reaction with increasing residence time; mixing with small amounts of saline and chemically-evolved water from matrix storage; and mixing with seawater near the coast. Low, uniform concentrations of strontium indicate short residence times up to decades (Edmunds and Brewerton, 1997).

4.2.6 Agricultural pollution

Nitrate concentrations are generally elevated above baseline and in the range 4-11 mg/l (as NO_3-N) due to leaching from agricultural land, and locally exceed the maximum admissible concentration (PCV (prescribed concentration or value)) of 11.3 mg/l for public and private supplies. Time series plots for nitrate show some seasonal variation (particularly in wet winters), with peaks in winter and spring, coinciding with periods of highest recharge, soil leaching and water levels (Jones and Robins, 1999; Entec, 2008a).

Entec (2008a) sampled for the pesticide atrazine at 29 out of their 37 sites and detected it at 16 sites, and above the DWI PCV of 0.1 µg/l at 1 site. Jones and Robins (1999) state that the higher concentrations of atrazine are clustered along the route of the Brighton-Lewes road and railway line. Fenuron was analysed for at 6 out of 37 sites and detected at 1 site where it was above the DWI PCV of 0.1 µg/l.

4.2.7 Chlorinated solvents

Entec (2008a) sampled 28 out of their 37 sites in the Brighton Chalk Block (a larger area than the coastal 6 km) for chlorinated solvents and detected tetrachloroethene and trichloroethene at 16 and 12 sites, respectively. The highest concentrations of total chlorinated solvents were found in the Falmer dry valley and within Brighton, near the A27 and A270 roads, but drinking water standards were not exceeded. Later data from the EA database for SO-5GWQ0496 has recorded maximum concentrations of tetrachloroethene of 0.81 µg/l (2013) and trichloroethene of 1.05 µg/l (2011); again below the DWI PCVs.

4.2.8 Saline intrusion

The Brighton Chalk block is vulnerable to saline intrusion both directly from the sea and from the tidal Rivers Adur and Ouse. The rivers that form the western and eastern boundaries of the area have chloride concentrations greater than 1000 mg/l under low flow conditions for the full length of these boundaries (Institute of Geological Sciences and Southern Water Authority, 1978). Chloride concentrations and conductivity particularly increase adjacent to the Adur and along the coast. Small seasonal variations in chloride concentrations of less than 5 mg/l have been observed inland, and time series plots of chloride, conductivity and sulphate show significant seasonal variation in the boreholes at Shoreham close to the River Adur where chloride concentrations can vary by more than 200 mg/l (Jones and Robins, 1999; Entec, 2008a). The highest values occur in autumn and the lowest in spring, corresponding respectively with times of low and high discharge from the aquifer.

Jones and Robins (1999) recorded that there are serious saline intrusion problems at three public supplies and intermediate chloride concentrations at three other sources. Another supply that had occasional elevated chloride concentrations has been abandoned. Changes in salinity in coastal boreholes have also been noted in response to pumping from boreholes located as much as 6 km inland (Monkhouse and Fleet, 1975).

The Ghygen-Herzberg relationship of relatively fresh water floating above a lens of saline water, that becomes more saline with increasing depth, does not apply in the dual porosity and permeability Chalk aquifer where the size, location and nature of the fracture systems through which the water predominantly moves is the major influence on the occurrence and extent of saline intrusion. Extensive geophysical logging of boreholes indicates that each hole has an unique conductivity profile and response, to variations in groundwater head, tide and abstraction rates (Jones and Robins, 1999). These logs also revealed that saline water moves inland along discrete horizons, whilst at the same time, freshwater is moving seawards.

The quality at one source (2.5 km from the sea) where most of the supply is obtained from a single large fracture at 21 m depth, responds rapidly to pumping, particularly at high tide, and there may be a direct fracture/conduit connection between the borehole and the sea (Warren, 1962). The source cannot be pumped for three hours either side of high tide and it also affects conductivities in the saline monitoring borehole at Roedean, 3 km to the south-west and 400m from the coast. Whilst at another source (500 m from the River Adur and 2 km from the coast) salinity rises and declines slowly, possibly indicating transport through less permeable horizons via many small fractures, with saline intrusion slow and on a broad front (Monkhouse and Fleet, 1975). Further from the coast, pressure differences propagate through the aquifer, even though the saline water cannot, and the dominant effect is on the groundwater head and/or the saline interface (Jones and Robins, 1999).

In the Western Lawns borehole (Hove), fluid conductivity logging indicated that the highest value of 27500 µS/cm occurred about an hour after high tide, and the lowest of 1600 µS/cm a similar time after low tide, accompanied by a fall in the depth of the saline interface from 24 m to 110 m (Jones and Robins, 1999). The saline water at high tide entering via an enlarged fracture at 130 m depth, and probably leaving via a fracture at 66 m. Reverse flow occurs during a falling tide with fresh water leaving the borehole via fractures at 114 and 130 m depth (Jones and Robins, 1999).

At Greenleas (2.5 km from the coast) conductivities are in the range 800 to 950 µS/cm, increasing slightly with time, however from a 1998 chemical analysis it was suggested that this related to agrochemical pollution, rather than saline intrusion (Jones and Robins, 1999).

Just to the west of the area, in the Worthing Chalk block, modern saline water was differentiated from older formation water being retained in the matrix, by simple dilution of strontium (Lancing interstitial waters) compared with strontium being significantly enriched compared with chloride (Sompting interstitial waters) (Jones and Robins, 1999).

4.2.9 Climate change data outputs

Figure 15 to Figure 19 summarise the impacts of climate change on temperature, rainfall, PET, groundwater recharge and groundwater levels based on the RCP8.5 emissions scenario as derived from UKCP18 and its application to eFLaG models in Brighton. There is high confidence that temperature will rise throughout 21st century (Figure 15), with warmer temperatures on average for all months of year. Warming is greatest in summer (JJA) and Autumn (SON), and the hottest days will get hotter.

Long term average rainfall (Figure 16) is projected to decrease over time on average across UKCP18 RCMs, but the direction of change is uncertain. On average across the RCMs wetter winters and drier summers are predicted, with the change enhanced over time. There is a high confidence of wetter winters/drier summers for the end of century. There is high confidence that rainfall extremes will become more extreme in winter by end of century and less extreme in summer. Note, however, that the latter results contradict the UKCP18 simulations driven by their convection permitting model (CPM) which indicate that summer rainfall extremes will become more extreme under the RCP8.5 emissions scenario. Simulations of summer rainfall extremes by the CPM are likely to be more reliable.

There is high confidence that PET will rise throughout 21st century (Figure 17), with mean month monthly PET increasing between April and October. PET increases are highest in summer and PET extremes (95th percentile) will get more extreme.

Long term average (LTA) recharge is projected to increase on average across RCMs, but the direction of change is uncertain (Figure 18). Winter recharge is projected to increase on average across RCMs, and recharge extremes (95th percentile) are projected to increase in winter.

LTA groundwater levels are projected to be stable over 21st century and the direction of change is uncertain (Figure 19). Spring groundwater level maxima are projected to increase and Autumn/Winter groundwater levels (during hydrograph rise) are projected to decrease on average across RCMs. The direction of changes in extremes is uncertain. The mean change across RCMs implies no significant change in groundwater level extremes during hydrograph peaks and troughs, suggesting no significant impact on flood/drought occurrence due to changes in recharge and rainfall.

Sea level rise is predicted to occur along coast and along the tidal extent of major rivers (Figure 20) which is likely to affect the extent of saline intrusion.

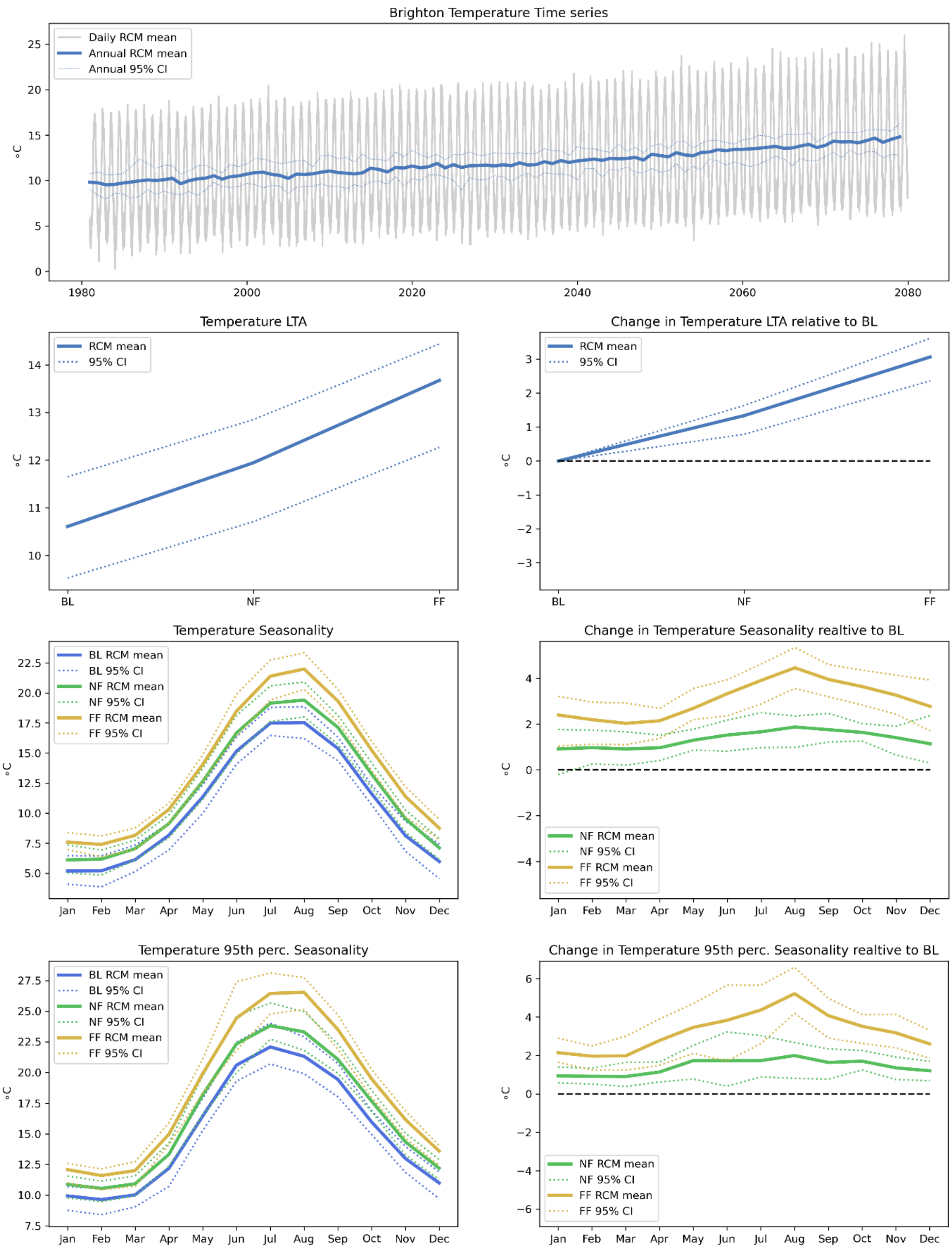

Figure 15 Summary of climate change impacts on air temperatures at Brighton from UKC18 regional projections. Top row: transient temperature time series, second row: changes in long term average temperature, third row: changes in temperature seasonality, fourth row, changes in 95th percentile by month.

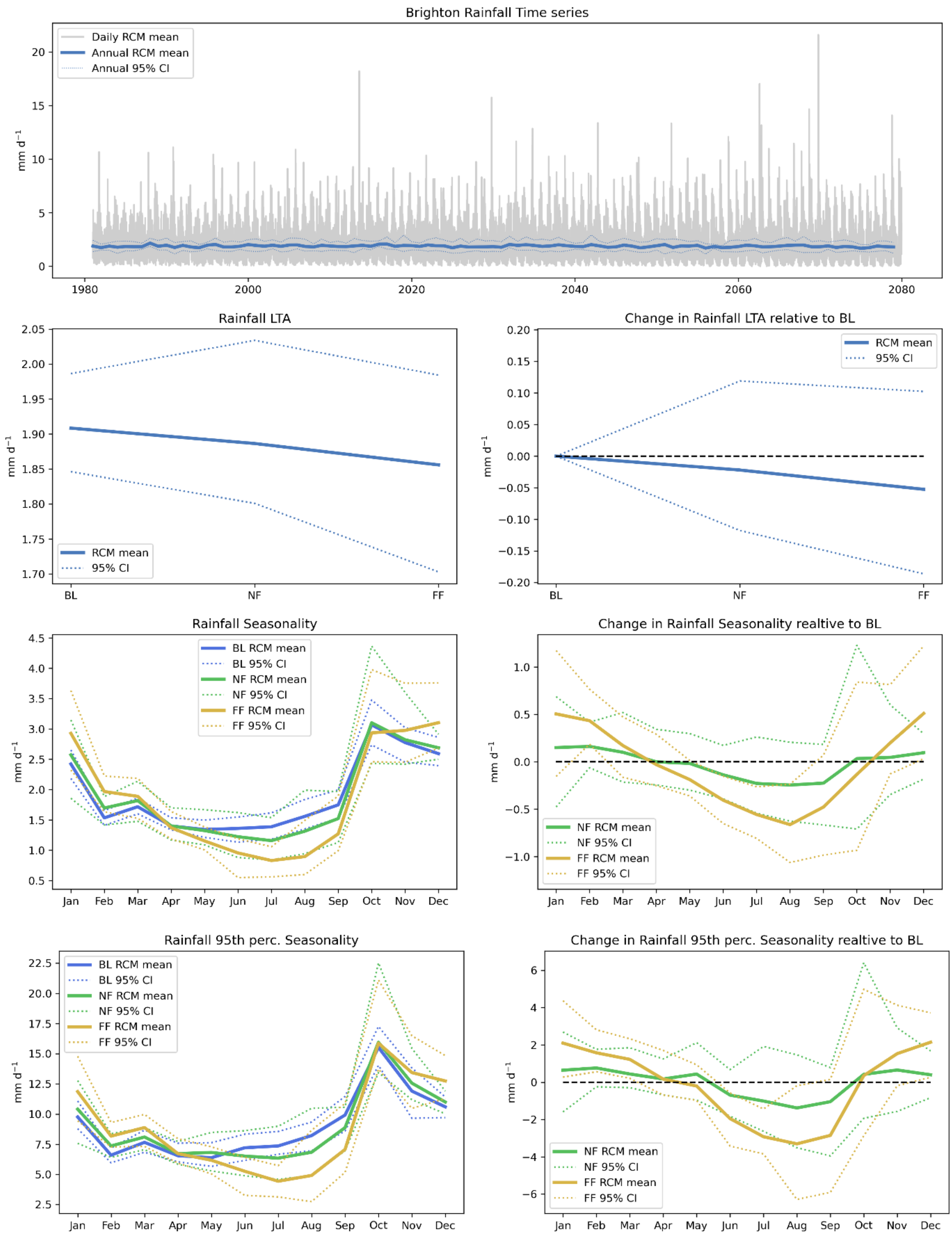

Figure 16 As Figure 15 but for rainfall at Brighton

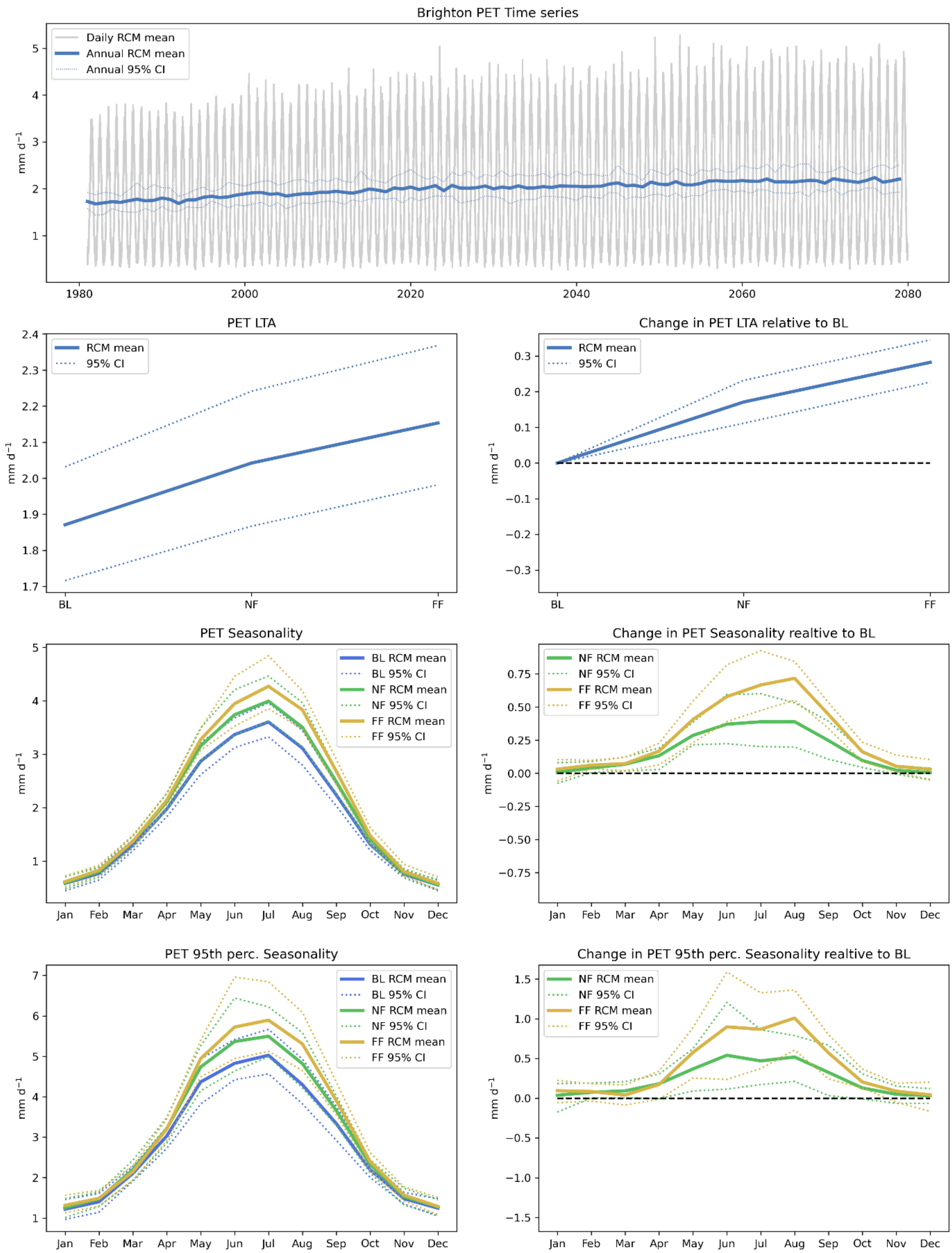

Figure 17 As Figure 15 but for PET at Brighton

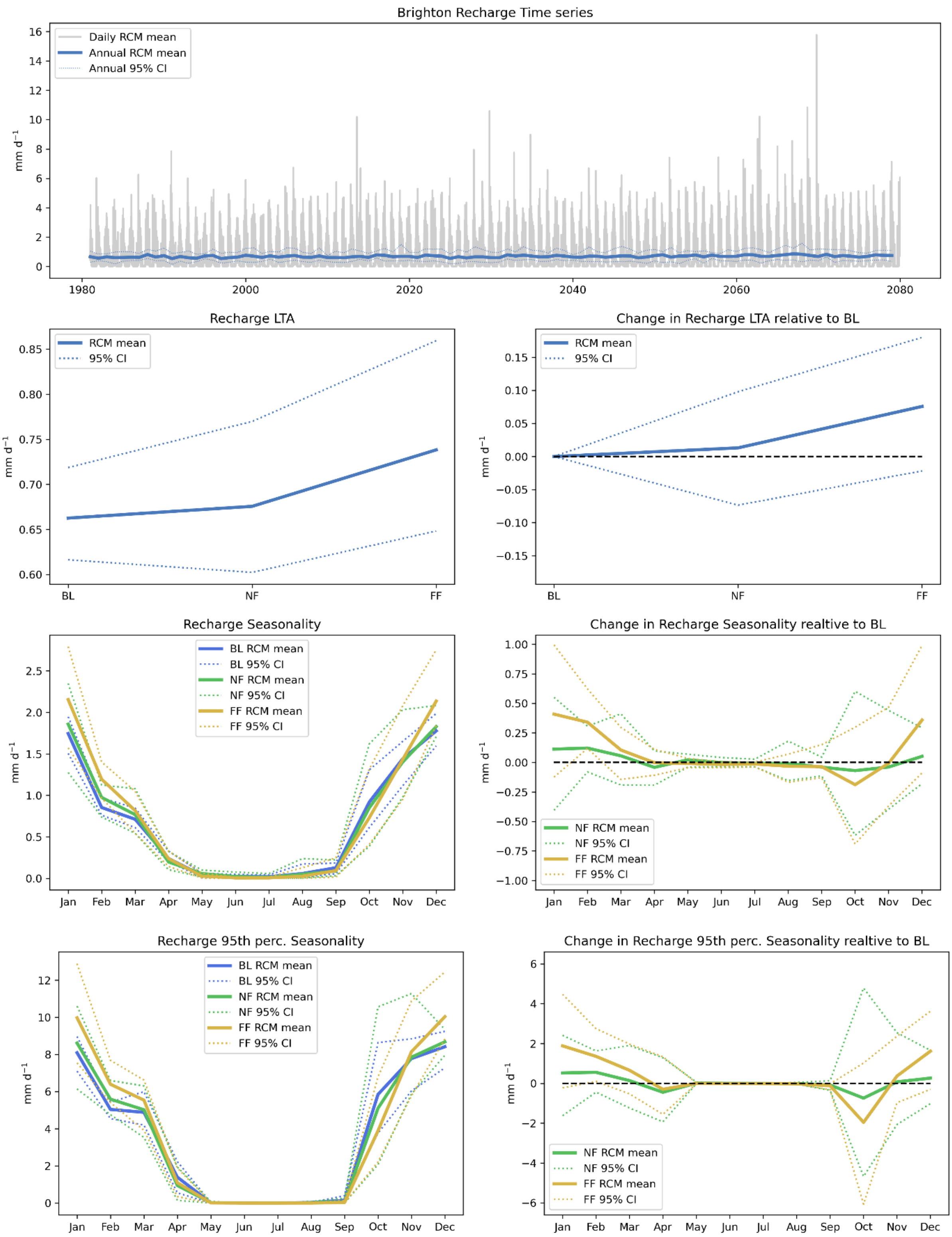

Figure 18 As Figure 15 but for groundwater recharge at Brighton

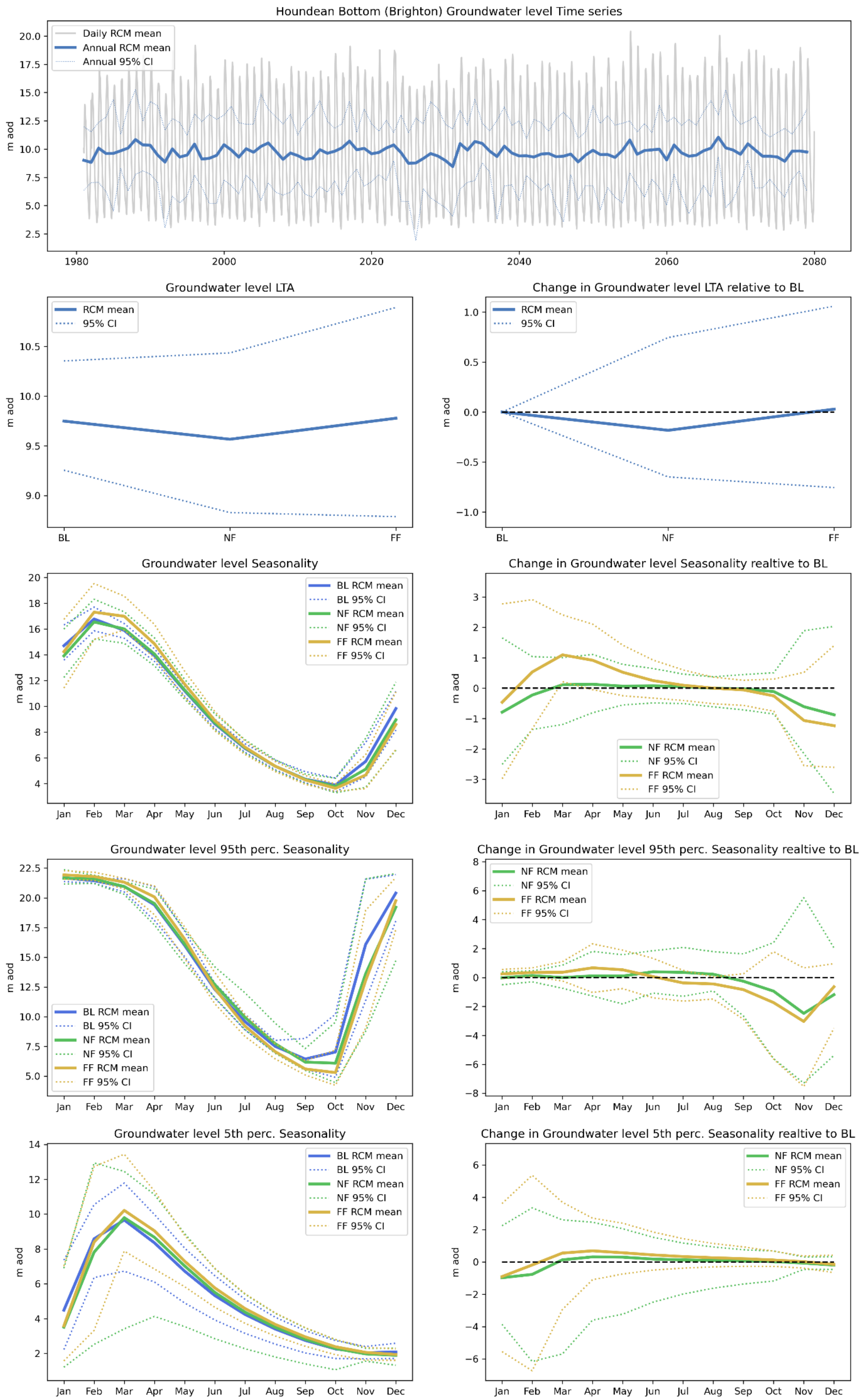

Figure 19 As Figure 15 but for groundwater levels at Brighton (Houndean Bottom)

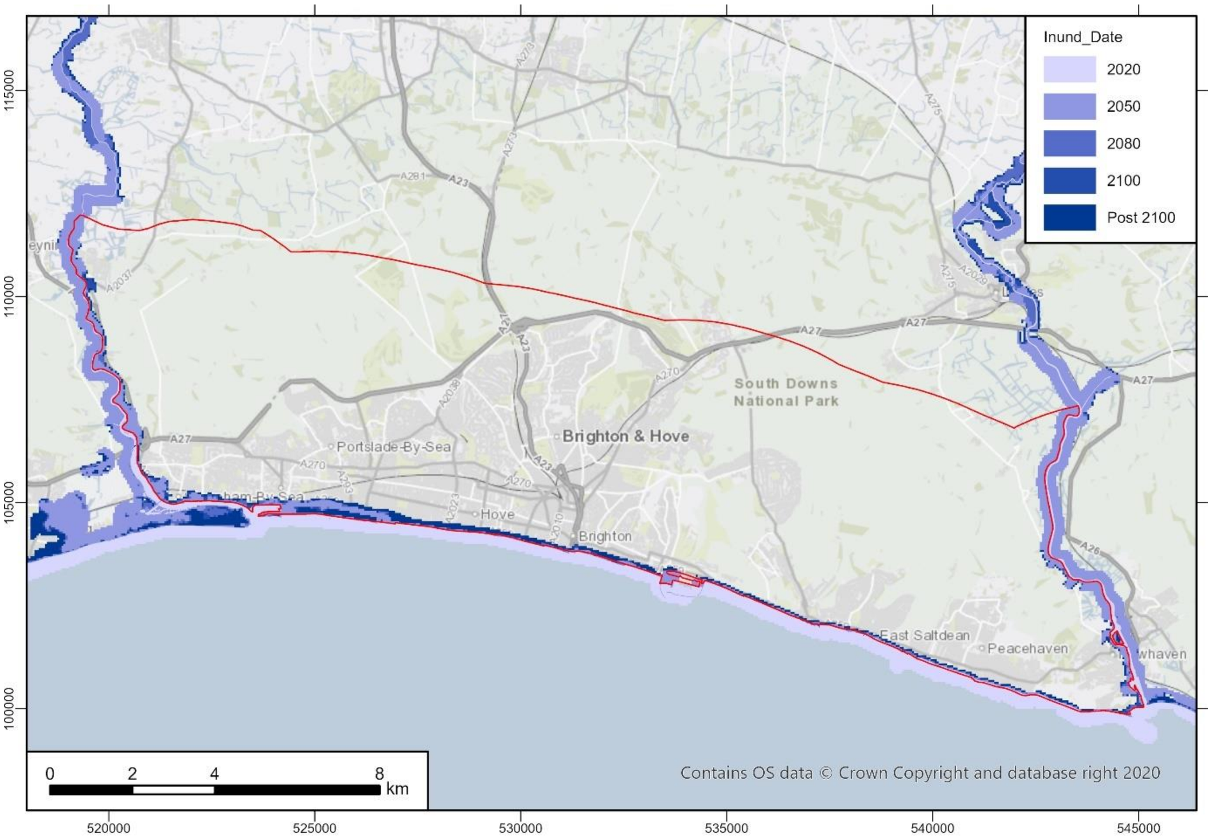

Figure 20 Brighton Study Area showing GeoCoast: Inundation UKCP18

4.2.10 Potential impacts of climate change on groundwater quality

Based on the groundwater quality issues for this case study described in sections 4.2.5 to 4.2.8 and the UKCP18 projections described in section 4.2.9, the following direct impacts of climate change on groundwater quality are anticipated:

1. Increased temperatures of up to 3°C could increase reaction rates for degradation of nitrate, pesticides and chlorinated solvents, but this may only be marginal.
2. Although the long term average rainfall is predicted to decrease on average, the direction of change is uncertain and winter long term average rainfall and recharge will increase. More extreme rainfall events in winter could increase the number of recharge events, and produce more nitrate and pesticide spikes with increased mobilisation of these contaminants. However, this may be offset by greater dilution.
3. Drier summers could lead to less dilution and more concentrated recharge of nitrate and pesticides.
4. The effect of wetter winters and drier summers increasing the size of the seasonal fluctuations in water levels, would also decrease the thickness of the unsaturated zone in spring, potentially decreasing the time lag for nitrate to reach the water table.
5. Greater winter groundwater levels could potentially increase mobilisation of agricultural pollutants.
6. The rise in sea level due to increasing temperatures will lower head gradients at the coast and hence increase the potential for saline intrusion from the sea and tidal Rivers Adur and Ouse. The effect of wetter winters and drier summers increases the size of the seasonal fluctuations in water level and hence could affect the timing of pumping from the coastal (winter) and inland (summer) public supply sources that prevent higher salinity waters being pumped and allow recovery of water levels and aquifer storage.

Some of these issues could potentially be investigated by modelling.

Potential changes of land use (whether directly or indirectly associated with climate change) may have more influence on agricultural pollution than other changes in climate.

4.3 CASE STUDY 2 - BIRMINGHAM

4.3.1 Setting

This case study area is the Birmingham Groundwater Management Unit. It forms part of the Tame-Anker-Mease Permo-Triassic sandstones Birmingham- Lichfield WFD groundwater body. The area is urban and comprises the part of the West Midlands conurbation predominantly within the western part of the area defined by the M6, M5 and M42. It has previously been studied as two different areas, the main northern 'Birmingham aquifer' and the southern 7.5 km (south of the Birmingham fault) from Bournville to Barnt Green as part of the 'South Staffordshire and North Worcestershire aquifer'. The majority of the area is drained by the River Tame and its tributaries (Perry Brook, Witton Brook and Hockley Brook) and the River Rea and its tributaries (Bourn Brook, Wood Brook, Merritt Brook) that drain east into the Trent. However, half the southern area (3-4 km) comprises managed grassland and is in the River Arrow catchment that drains south into the Severn. Several canals cross the area (Worcester and Birmingham, Grand Union, Birmingham, Birmingham and Fazeley and Tame Valley canals). The northern part of the area rises from an elevation of 91 m at Nechells in the north-east to 236 m in the west at Warley Park. The southern part rises from 145 m at Cofton where the River Arrow leaves the area to 216 m in Cofton Park. Mean annual rainfall is around 760 mm and Knipe et al (1993) used an effective rainfall of 251 mm/a for outcrop sandstone in their model, with additional input from urban return flows, reduced depending on the lithology of the superficial deposits and degree of urbanisation. Rivett et al (2012) used a range of recharge values of 108, 174 and 221 mm/a in their model. Daily and Buss (2013) quote recharge values of less than 1 mm/a for urban areas, 130-150 mm/a suburban areas, 80-90 mm/a for permeable superficial deposits (glaciofluvial sand and gravels and river terrace deposits), less than 50 mm/a for low permeability superficial deposits and for outcrop areas, 220-250 mm/a (arable) and 100-120 mm/a (woodland).

4.3.2 Industrial history

Birmingham has a long industrial history that has led to pollution of both the ground, and surface and groundwaters. Past industrial uses include gasworks, railway depots and carriage works, power stations, industrial sites (including chemical works, foundries, rolling mills, engineering workshops, scrap metal yards), sand pits used as landfills and sewage works. Former gasworks used for coal carbonisation, purification, tar storage, dumping of 'spent oxide' and coke storage have led to acidic soil pH, and high concentrations of sulphates, phenols, coal tars and other aromatic hydrocarbons, oils, free and complexed cyanides, elemental sulphur and sulphides. Railway depots and carriage works have used ashes, cinder, coke and other fill deposits to raise the land surface and spillage of liquids and waste including heavy metals, oils, solvents and paints have taken place. Other former industrial sites contain spilt metals, oils (including diesel and petrol), acids, degreasing agents, solvents, contaminated ash water and residues of electrical and household appliances (including polychlorinated biphenyls). Infilled sandpits have taken a variety of industrial waste including some with high metal contents. At sewage works the original soils are now buried beneath later fill material, retaining pollutants such as heavy metals, and contain a high proportion of organic material that can slowly decompose and under saturated conditions produce nitrogen-rich leachate and methane gas. There has also been leakage from sewers (Knipe et al, 1993).

4.3.3 Geology

The bedrock mainly comprises sandstones of the Permo-Triassic age Sherwood Sandstone Group (Table 8, Figure 22). Thicknesses are variable reflecting deposition on a surface of varied relief and geology (Allen et al, 1997). There is a structural low in the Edgbaston area adjacent to the Birmingham Fault. This fault forms the eastern boundary of the main northern area and downthrows the overlying Mercia Mudstone to the east against the Sherwood Sandstone, the throw decreasing from about 200 m around Erdington-Gravelly Hill to 60 m south-west of the junction with the north trending Northfield Fault (Powell et al, 2000). The southern part of the area is delineated to the west by the Birmingham, Rednal, Fiery Hill and Cherry Hill faults, and to the east by the Barnt Green, Longbridge, Hopwood, Tessell and Northfield faults.

The bedrock is overlain by alluvium (silty or sandy clay and clay with lenses and beds of sandy gravel and thin lenses of peat) with associated river terrace deposits (sand and gravel) along the River Tame valley (Figure 22). Glacial deposits comprising glaciofluvial deposits (sandy gravel), glaciolacustrine deposits (clay, laminated clay and silt) and till (clay and sandy clay with pebbles) are present over large parts of the area. These superficial deposits locally reach 20-40 m in thickness south of Smethick and in the proto-Tame valley, this lies to the south of the current course of the river where they are less than 10 m (Powell et al, 2000).

There are also extensive areas of made ground along the River Tame and Hockley Brook and worked and made ground at Queslett (Figure 22).

Table 8 Stratigraphy of Birmingham Groundwater Management Unit

Group	**Formation**	**Description**
Sherwood Sandstone Group	Helsby Sandstone Formation (formerly Bromsgrove Sandstone Formation)	29-124 m red-brown, medium-grained calcareous sandstone with thin beds of red mudstone; red mudstone clasts and caliche pellet breccias locally common; quartz and feldspar pebbles common at base
	Wilmslow Sandstone Formation, Wildmoor Sandstone Member	0-122 m orange-red, fine-grained sandstone, clayey in part; thin beds of red mudstone
	Chester Formation (formerly Kidderminster Formation)	26-112 m medium to coarse-grained, red and brown sandstone with beds and lenses of pebbly sandstone and conglomerate and scattered quartz pebbles
	Hopwas Breccia Formation	0-37 m breccia predominantly comprising quartzite and sandstone clasts with thin beds of red mudstone and sandstone
Warwickshire Group	Client Formation	c. 20 m red-brown and purple breccia with clasts of sandstone, shale and volcanic rock in sandy clay matrix
	Salop Formation (Enville Member)	50-160 m red mudstone, with bands of massive sandstone and locally calcareous conglomerate

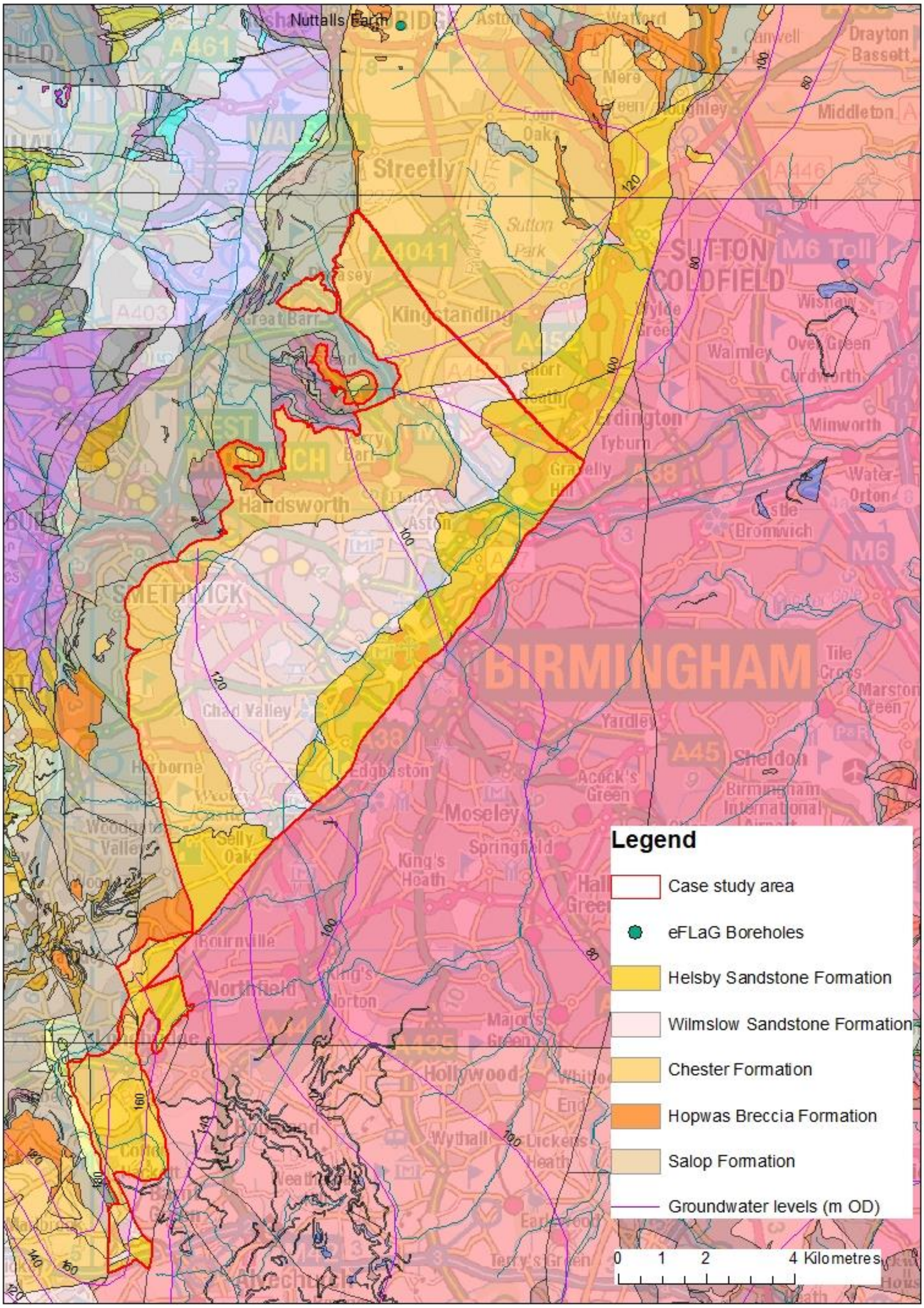

Figure 21 Bedrock and groundwater levels in Birmingham Groundwater Management Unit. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

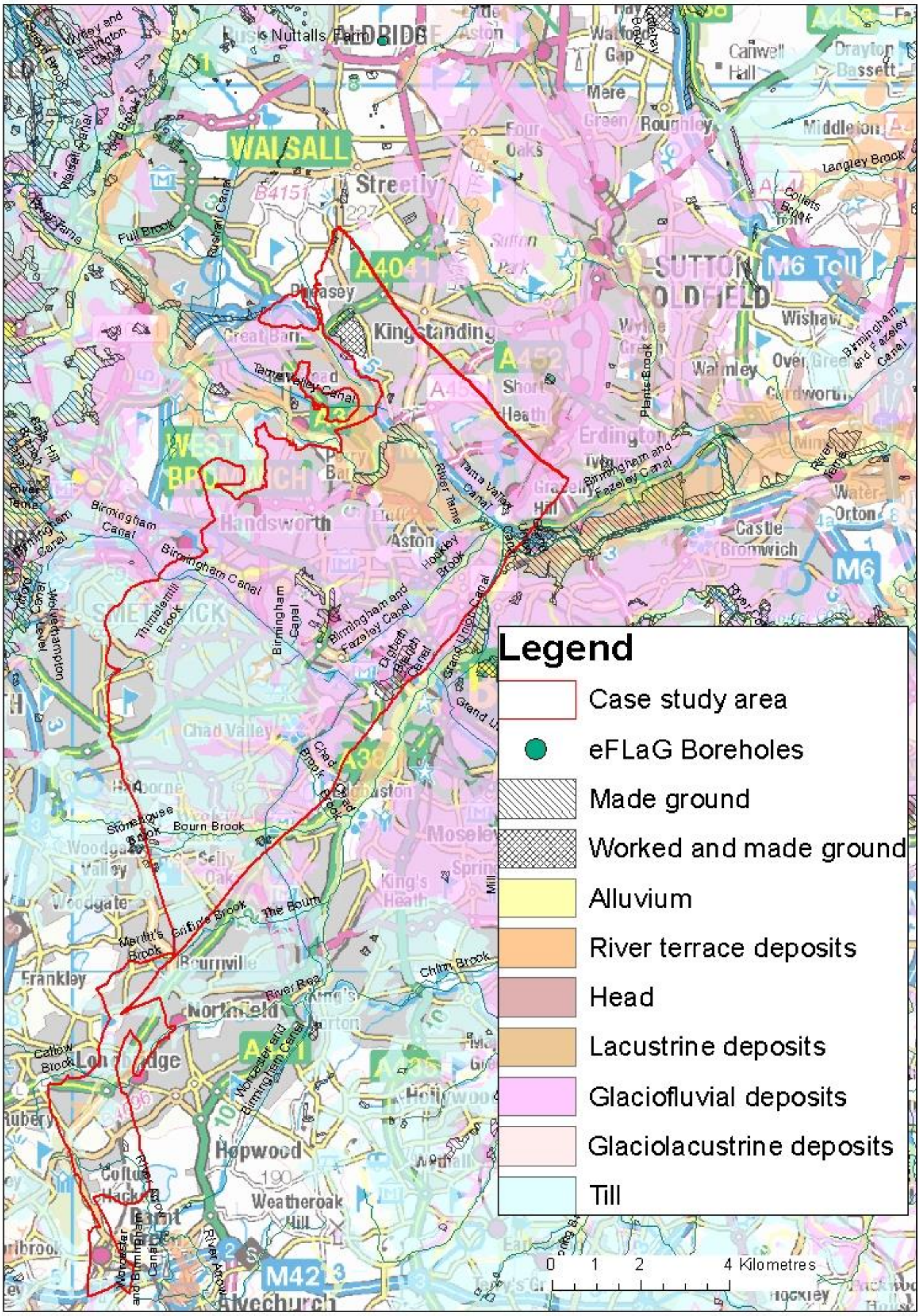

Figure 22 Superficial deposits in Birmingham Groundwater Management Unit. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

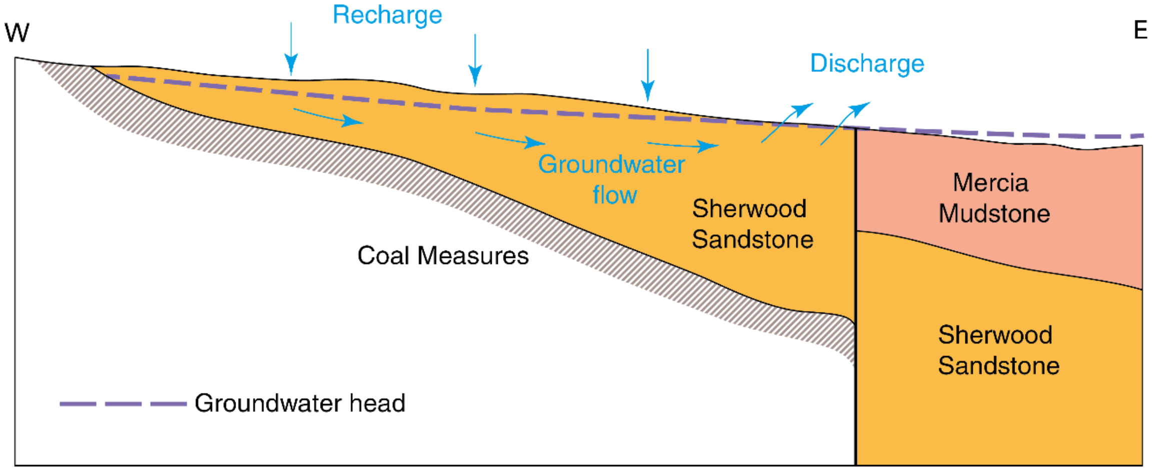

Figure 23 Groundwater flow across the Birmingham Fault

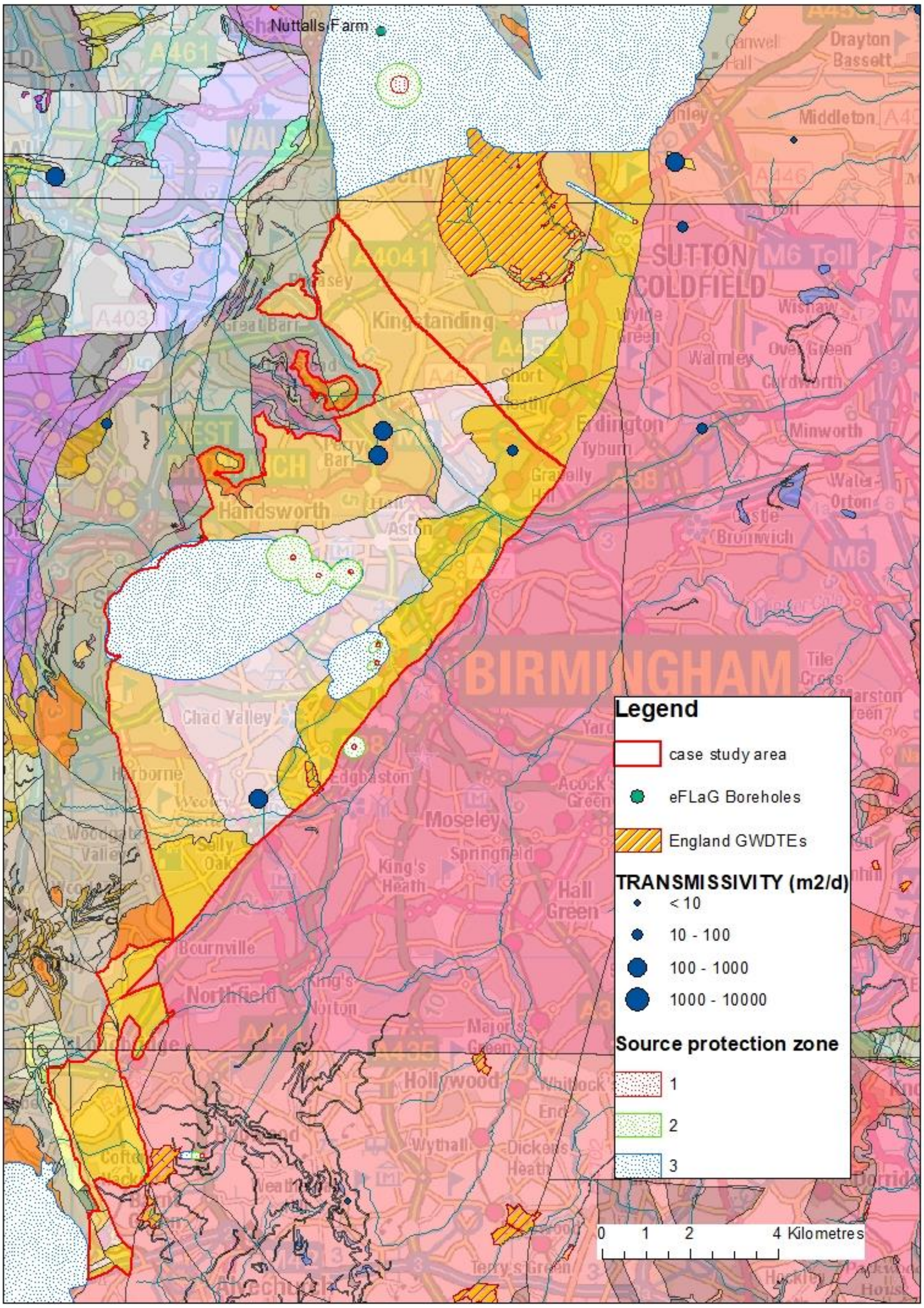

Figure 24 Various hydrogeological parameters in Birmingham Groundwater Management Unit. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0. Source protection zones © Environment Agency copyright and/or database right 2016.

4.3.4 Hydrogeology

The Permo-Triassic Sherwood Sandstone forms a regionally important aquifer exhibiting both intergranular and fracture flow. The Environment Agency have a numerical groundwater model of the Birmingham area (Daily and Buss, 2013). Prior to the 1850s, it is likely that recharge dissipated through the aquifer towards the main discharge areas along the River Tame and River Rea. Clays in the superficial deposits confined the aquifer along the Tame and Rea valleys, and it was potentially overflowing between Witton and Nechells. There is anecdotal evidence that the Nechells area in the lower Rea valley was a wetland in the late eighteenth century (Knipe at al, 1993). Land (1966) reported that springs discharge at Tamhorn Park by the River Tame, adjacent to the Birmingham fault. Out of the ten wells recorded as overflowing at the time of construction, none did so in 1966, with five located immediately north-west of the Birmingham fault (Land, 1966).

Groundwater levels in the area were historically depressed due to abstraction from the Permo-Triassic Sherwood Sandstone aquifer exceeding recharge. Falling water levels induced leakage from the canal network into the unconfined aquifer (Daily and Buss, 2013). By 1885 pumping was also taking place from the confined aquifer beneath Mercia Mudstone Group, east of the Birmingham fault; this would have resulted in groundwater being drawn eastwards from the unconfined to the confined area (Jackson and Lloyd, 1983). However, abstraction from the Birmingham Groundwater Unit decreased by more than 80% between the peak in the 1940s and 1993 from 75 Ml/d to less than 15 Ml/d (Knipe et al, 1993), and 8-15 Ml/d for 2005-2010 (Rivett et al, 2012). This has led to a rapid rise in groundwater levels and without intervention eventually they will return to historic elevations. New boreholes were commissioned in 2008 to discharge to the River Tame and augment summer flows in the Trent (Rivett et al, 2012); some of these have since been developed for public water supply (Daily and Buss, 2013). Groundwater levels in the aquifer have been published for several different dates: pre-abstraction (Knipe et al, 1993), 1966 (Land, 1966), 1976 (Jackson and Lloyd, 1983), 1988 (Knipe et al, 1993) and 2008 (Rivett et al, 2012). During the period of peak abstraction, the River Tame and groundwater levels were completely disconnected (Daily and Buss, 2013). Since recovery and stabilisation, levels are similar to those postulated to have occurred pre-abstraction with flow converging on the Tame valley. Levels have returned to lie within the superficial deposits, which are in good hydraulic connection with the river and facilitate groundwater discharge to it (Daily and Buss, 2013). The northern boundary of the Birmingham groundwater unit is slightly further south-east than the Streetly – Roughley area considered to form the groundwater divide.

The River Tame is in a semi-natural channel as it crosses the Sherwood Sandstone outcrop and its beds and banks allows unrestricted inflow of shallow groundwater. The River Tame rises on Coal Measures to the west, where chloride concentrations are higher, possibly contributing over 25% of groundwater flow, and is diluted progressively by less mineralised Sherwood Sandstone groundwaters as it crosses its outcrop. However, the Hockley Brook is piped and culverted and the River Rea (primarily on Mercia Mudstone outcrop) is in a brick and concrete channel; meaning they receive little direct inflow, but general down-valley flow, parallel to the water course, with inflow some considerable distance (potentially kilometres) downstream. Knipe et al (1993) estimated that with a low rate of abstraction, water levels could rise by 10-12 m between 1990 and 2020 south of the River Tame, but relatively little change north of it. The rising levels cause water to overspill into the superficial deposits, particularly along the Birmingham Fault where levels had already risen, with water moving into the superficial deposits overlying the Mercia Mudstone Group in places (Figure 23). Knipe et al (1993) described where rising water levels had already affected foundations and predicted that water level rises of between 0 m (along Tame) and 15 m (south-west of Smethick where highest elevation and furthest from rivers) could occur beyond 2020. The 2008 contours (Rivett et al, 2012) indicate that apart from in the immediate area around former abstractions recovery has been slower than Knipe et al (1993) predicted. Daily and Buss (2013) indicate that most groundwater recovery had occurred prior to the end of the first decade of the twenty-first century with a flattening off reflecting the establishment of a new recharge and abstraction regime.The Birmingham fault to the east acts as a barrier to shallow groundwater flow, with water level differences (drops) across it of up to 45 m in 1966 and 25-30 m in 1993. However, as the potentiometric surface has risen on both sides of the fault, some flow must occur across it, but this may be limited to areas where sandstones are juxtaposed on both sides of it, with the connectivity related to the thickness of sandstone in contact, which is small around the structural low (Jackson and Lloyd, 1983). Daily and Buss (2013) stated that heads are predominantly 10 m higher on the west side of it, with recovery occurring faster in the confined than the unconfined aquifer. Daily and Buss (2013) state that Severn Trent Water thought that the Birmingham Fault was the only fault to disrupt hydraulic continuity. However, Jackson and Lloyd (1983) suggested that downdip in the confined aquifer, some of the other faults restrict groundwater flow.

The Sherwood Sandstone is normally considered as a single aquifer unit. This is true of the Wildmoor Sandstone and Chester Formation, but within the Helsby Sandstone fine-grained horizons can cause aquifer stratification, and mudstone horizons can affect vertical hydraulic continuity when it can act as a multi-layered aquifer, potentially with perched water levels. Land (1966) reported that the head in the Helsby Sandstone was up to 15 m higher than that in the Wildmoor Sandstone at Birmingham, presumed to be due to marl horizons restricting vertical movement.

Allen et al (1993) quote interquartile core porosity ranges for the West Midlands area for the different formations of the Sherwood Sandstone as: Helsby Sandstone 25.7-29.7%, Wildmoor Sandstone 24.2-28.2% and Chester Formation 19.6-28.4%. The lower values in the Chester Formation reflecting the greater amount of cementation. Horizontal intrinsic permeability values measured on core samples vary from similar to about ten times less than bulk hydraulic conductivity values (interquartile range 1.1-33 m/d) derived from pumping tests; indicating the variable effect of fracturing (Allen et al, 1997). Horizontal core values are about twice those of vertical ones for the Wildmoor and Chester sandstones, but are similar in the Helsby Sandstone.

Allen et al (1997) state that hydraulic conductivity values used in groundwater models are at the higher end of the core data range and lower end of those derived from pumping tests, and are most successful at reproducing the piezometry if they incorporate vertical layering.

Mean transmissivity values (Figure 24) vary between 96 and 230 m^2/d (Allen et al, 1997). Daily and Buss (2013) quote some additional more recent values that range between 38 and 913 m^2/day. Knipe et al (1993) used a hydraulic conductivity of 1-1.5 m/d in their models, with transmissivity values of 20-330 m^2/d and a storage coefficient 0.15. Halcrows (1994) considered the Chester Formation the most permeable and also used a specific yield of 0.15 west of the Birmingham Fault.

There are three public water supply sources within the area, and two private water supplies. There is also a recent public water supply, for which no source protection zone has yet been defined. The source protection zone for another private water supply extends into the area, although the source itself is not.

4.3.5 Groundwater dependent terrestrial ecosystems

The Edgbaston Pool SSSI (artificial, created in 1790) is located on glacial sands and gravels over the Helsby Sandstone Formation.

4.3.6 Groundwater quality

Only the narrow, southern, less urban, 7 km of the area (south of Bournville) are included in any regional water quality review reports (Tyler-Whittle et al, 2002) and this only contains a single sample site from this eastern part of the South Staffordshire and North Worcestershire area, for which no analyses are quoted, just incorporated in the statistical summaries.

Groundwaters in the unconfined Sherwood Sandstone are generally of calcium-bicarbonate type and oxidising. Historically the water quality was good and used extensively for drinking and food industries. When bacterial infections related to poor sanitation became recognised, the source of drinking water was transferred in 1906 to the central Wales reservoirs with groundwater used primarily for industry (Halcrow, 1994).

Land (1966) quotes the following most frequent ranges from calcium-bicarbonate groundwaters sites predominantly located on Sherwood Sandstone outcrop 40-90 mg/l calcium, 3-12 mg/l magnesium, 8-15 mg/l sodium and potassium, 70-90 mg/l bicarbonate (as CO_3), 40-60 mg/l sulphate, 20-30 mg/l chloride, 20-40 mg/l nitrate (NO_3), and total dissolved solids of 200-330 mg/l. Iron varies from a trace to 2 mg/l. However higher values of some determinands were found west of the Birmingham Fault, due to local pollution, with up to 189 mg/l calcium, 79 mg/l magnesium, 49 mg/l sodium and potassium, 135 mg/l bicarbonate (as CO_3), 235 mg/l sulphate, 247 mg/l chloride, 89 mg/l nitrate (NO_3), and total dissolved solids of 1032 mg/l (Land, 1966). Nitrate and natural hardness can be constraints in the unconfined aquifer (Rivett et al, 2012). Knipe et al (1993) quote a representative water quality analysis from the unconfined aquifer (Grand Hotel) of 22 mg/l calcium, 24.7 mg/l magnesium, 6 mg/l sodium, 135.4 mg/l bicarbonate, 0.5 mg/l carbonate, 37 mg/l sulphate, 12 mg/l chloride, 6 mg/l nitrate (NO_3), with a pH of 8.1, Eh 350 mV and total dissolved solids of 176.7 mg/l.

The Wildmoor Sandstone Member and Chester Formation are usually undersaturated with respect to calcite and dolomite; whereas in the Helsby Formation, groundwaters are usually saturated or oversaturated (Rivett, 1988). The pH is controlled by calcite in the rock matrix, this is absent in the upper part of the sequence, allowing a low pH to develop and keeping metals in solution. Below the main water table, the presence of calcite increases the pH and metals (except chromium) are lower. Acid attenuation reactions are dependent on the sandstone mineralogy with near surface weathered rock attenuating by ion-exchange and silicate and iron oxyhydroxide dissolution, while deeper sandstones have additional buffering capacity involving calcite dissolution (Buss et al, 1997).

Sources at Edgbaston and Ward End had considerably lower non-carbonate hardness in the Wildmoor Sandstone than in the Helsby Sandstone. In sources at Aston and Smethwick both total dissolved solids and chloride increased over time.

Jackson and Lloyd (1983) state that chloride is generally in excess of 20 mg/l (with the highest values closest to the River Tame and Hockley Brook) and nitrate over 30 mg/l (as NO_3) representing waters with modern ages; however sites in the area of the structural low had chloride less than 15 mg/l, and nitrate less than 10 mg/l (as NO_3) with ages over 3000 years, probably representing a mixture of modern and older water more than 4000 years old. The highest chloride concentrations in the unconfined aquifer coincide with areas of prolonged abstraction leading to induced recharge of (high chloride) surface water. Older groundwaters with high chloride, are likely to occur where dissolution of chloride minerals in the aquifer has occurred.

The shallow active groundwater zone has for many decades been below the quality required for public supply due to pollution from urban and industrial areas (Knipe et al, 1993). Good quality water is present at depth, particularly in the Helsby Sandstone, due in part to the confining nature of the marl bands with central Birmingham groundwater still used by hotels, dairies, breweries and food processing (Halcrow, 1994) and more recently public supply. Jackson (1981) postulated that the decrease in total dissolved solids down hydraulic gradient with higher sulphate, chloride and nitrate and lower bicarbonate in the shallow aquifer compared with at depth, was caused by abstraction causing the mixing of older waters with recharge. Water quality has been improved by drilling and casing to greater depths. However, boreholes can act as conduits for poor quality water.

There were a series of PhD theses from the University of Birmingham from the 1980s onwards studying the pollution of the Birmingham aquifer. Jackson (1981) and Ford (1989) looked at the inorganic quality and Rivett (1988) organic. Organic and inorganic pollution show very different distributions, reflecting differences in physical transport processes, chemical interactions and histories of chemical usage. There are records of sulphuric acid being used as early as 1740, whilst solvent use only dates back to the 1950s (Ford and Tellam, 1994). Ellis (2002) looked at the impact of urban groundwater (including copper, nickel, sulphate, nitrate, chlorinated solvents (eg TCE) and biodegradation products) on surface water quality and concluded that groundwater concentrations were generally lower than anticipated due to dilution and natural attenuation in the sandstone aquifer and river beds.

Despite groundwater levels having been shallow for the previous 5-10 years, Knipe et al (1993) observed no significant inflows of contaminated groundwater into rivers and sewers in the made ground and superficial deposits adjacent to the principal water courses. This could reflect wide spacing of sampling points, or the fact that the Tame is already poor quality when it reaches the area. The greatest concentration of known and suspected contaminated sites are in the lower Rea valley from Saltley to its confluence with the Tame, in the lower Hockley Brook valley down to the Tame and in the Tame valley east of Wilton; coinciding with where the manufacture of town gas and its by-products once took place. Previously abstraction kept regional water levels below the levels of the rivers and brook. Now groundwater levels have risen above river beds and the direction of groundwater flow is towards surface water courses, creating potential for migration of mobile pollutants from made ground and shallow soil. Potential pollutants are predominantly water-soluble but include any lighter fraction oils and tars that tend to float on groundwater. The rate and direction of flow will depend on the hydraulic gradient and watertightness of the river channels. The lack of connection of the Hockley Brook and River Rea with the aquifer and hence inflows occurring some considerable distance downstream, allows some dilution and dispersion of pollutants.

In recharge areas, water movement in the saturated zone may have an appreciable vertical component, with shallow water travelling slowly into deeper parts of the aquifer. Elsewhere flow is predominantly horizontal. Point sources and contaminated sites will tend to produce plumes of pollution along the direction of groundwater flow. Pollutants in groundwater tend to be removed or reduced in concentration with time and distance travelled with rates dependent on depth to water, sorption, permeability, hydraulic gradient and length of flowpath. The mechanisms involved include breakdown by oxidation, adsorption, precipitation (usually under changed pH or Eh), dilution and filtration of particulates. Groundwater below superficial deposits is likely to show further attenuation and dilution of pollutants. In particular the higher pH and passage through carbonate-rich water in the sandstones will tend to reduce the solubility of metals and cause them to precipitate out, although organic contaminants are likely to be more persistent. Local plumes of pollutants may not spread widely if intercepted by pumping boreholes (Knipe et al, 1993).

4.3.7 Inorganic pollution

Ford and Tellam (1994) found that only nitrate and barium consistently failed drinking water standards and Halcrows (1994) stated that inorganic contamination is a limited problem.

Ionic concentrations at shallow depths are often much higher than from pumped waters (apart from nitrate which varies less with depth). A fluctuating water table increases trace metal mobility (Ford, 1989) and intermittent pumping draws in a disproportionate percentage of shallower, high concentration groundwaters. In the Tame valley waters from the near surface zone have low Eh, complete oxygen removal, nitrate reduction occurring and also decreasing mobility of heavy metals.

However, the Wildmoor Sandstone and upper part of the Chester Formation are depleted in calcite, so the neutralising of acids is not guaranteed, with a decline in pH occurring, this is accentuated by rising water levels, resaturating the poorly buffered near surface part of the aquifer and saturating polluted lithologies (Ford, 1989).

Ford et al (1992) compared groundwater quality data from dates 10 years apart, and observed that it was becoming more acidic (0.6 pH units) with increased solubility of toxic metals. They discuss several possible causes including the rise of the water table into carbonate-poor upper levels of the aquifer, oxidation of the Quaternary deposits above the sandstone aquifer and other acidic recharge, but concluded that inorganic acid spills, oxidation of inorganic pollutants and degradation of industrial organic pollutants were likely to be dominant in different parts of the city.

4.3.7.1 Nitrate

There are a range of nitrogen sources in Birmingham, including sewer leakage, industrial sources (nitric acid, metallic nitrates), fertilisers in parks and gardens and urea used as a road de-icer. Nitrate is almost ubiquitous in the unconfined aquifer, and ammonia and ammonium from sewer leakage are oxidised. Data from the EA database gives values in excess of 20 mg/l (as N) from boreholes at sites MD-64524882 in 2000 and MD-64547530 in 2008.

4.3.7.2 Barium

Barite is present at trace levels in the Sherwood Sandstone, in micas, feldspars and carbonates and sorbed onto clays or organic material. In otherwise uncontaminated groundwaters with low sulphate concentrations, levels of barium have exceeded 500 µg/l and the source is assumed to be lithological (Ford and Tellam, 1994).

4.3.7.3 Phosphate

Ford and Tellam (1994) found that only groundwaters from the Wildmoor Sandstone had high phosphate, where 75% of samples contained significantly elevated concentrations with a maximum of around 1 mg/l (as PO_4). The control appears to be pH- related, with Wildmoor Sandstone groundwaters having characteristically low pH (5.5-6.8) compared with the other formations; and those Wildmoor Sandstone groundwaters with low or zero phosphate, having pH in range 6.9-7.4. Data from the EA database gave an orthophosphate concentration of 2.43 mg/l (as P) in November 2003 at site MD-64496880.

4.3.7.4 Metals

A few abstraction boreholes have highly contaminated groundwaters with levels of heavy metals of mg/l. There is diffuse pollution below the city centre, but generally the pollution is mainly from point sources. There has been an infiltration lag in west Birmingham due to the presence of extensive superficial deposits and the unsaturated zone thickness, and this former buffer zone is now acting as a sustained source of poor-quality recharge water (Ford, 1989). Boreholes close together can have very different water quality; e.g. due to the presence of oil (causing hydrocarbon reduction, rather than being oxidising) (Ford, 1989). Land use and chemistry are associated, with the highest salinity, sulphate, chloride, sodium, boron and total heavy metal concentrations associated with metal working. High concentrations of heavy metals and ammonium are unexpected in the near neutral pH groundwaters due to the high sorption capacity of the sandstones, and may relate to loading or colloidal transport and/or complexation.

4.3.8 Organic pollution

Rivett (1988) looked at organic pollution and concluded that chlorinated solvents, particularly TCE (trichloroethylene, related to metal cleaning), were widespread. Rivett et al (1990) sampled at 44 unconfined sandstone sites, of which 86% detected TCE and in 50% it exceeded the WHO guideline for drinking water of 30 µg/l, with a maximum of 5500 µg/l. The distribution was not just related to land use and the proximity of sources, but also the presence of thick clay-rich superficial deposits, with contamination greatest where the superficial deposits are sandy and thinnest (Tame valley) and lowest around Smethwick where the deposits comprise thicker till. Other factors such as a thin unsaturated zone, rise in water levels mobilising previously adsorbed contaminants and shallow borehole casing all coincided with increased solvent contamination in boreholes. Abstraction history is also a factor (Rivett et al, 1990). Inorganic contamination (nitrate and chloride) showed some correlation with solvents, but did not always occur with higher solvent concentrations. TCE has a low solubility and high density, allowing penetration deep into the aquifer as an immiscible phase. Aerobic conditions in the unconfined aquifer would be expected to prevent degradation. Over the period of Rivett's study TCA (1-1-1 trichloroethane) which replaced TCE in the metal industry in 1965, rose and TCE stabilised. The maximum concentrations of TCA found by Rivett (1988) was 780 µg/l and PCE (perchloroethylene) which replaced TCE in the dry cleaning industry in 1965, had a maximum of 460 µg/l. VOCs remain within the aquifer for a long time due to their retention on superficial deposits and sandstone and their low solubilities. Data from the EA database gave TCE values of 58 µg/l in February 2000 from site MD-64524882 (NGR 4063330, 288180) and 91.7 µg/l in March 2007 from MD-59012880.

Apart from chlorinated solvents, general organic contamination was low, with 86% of samples 0.01-0.05 µg/l, and few > 1 µg/l; probably from degraded lubricating oils (Rivett, 1988). Knipe at al. (1993) found that spilled oils and tars from gasworks have been washed down through the soil profile and become concentrated at, or just above, the water table. Hydrocarbons degrade producing large quantities of carbon dioxide. A fluctuating water table aids hydrocarbon degradation, as it can enhance carbon dioxide generation because of an increased supply of oxygen to the microbiota (Ford, 1989).

Shallow mudstone horizons have locally prevented the majority of DNAPLs penetrating to depth but successful remediation often depends on similar favourable conditions (Rivett et al., 2012). The closure of boreholes above the drinking water limit for TCE is related more to the recession in the metal working sector than to contamination. Chlorinated hydrocarbon contamination is likely to last for decades, with PCE more persistent (less soluble, more sorptive and introduced later) than TCE. Dioxane (additive in industrial TCA) may be present, but was not been analysed for (Rivett et al, 2012).

4.3.9 Climate change data outputs

Figure 25 to Figure 29 summarises the impacts of climate change on temperature, rainfall, PET, groundwater recharge and groundwater levels as derived from UKCP18 and it’s application to the eFLaG models for Birmingham. There is high confidence that temperature will rise throughout 21st century (Figure 25), with warmer temperatures on average for all months of year. Warming is greatest in summer (JJA) and Autumn (SON), and the hottest days will get hotter.

Long term average rainfall (Figure 26) is projected to decrease over time on average across UKCP18 RCMs. On average across the RCMs wetter winters and drier summers are predicted, with the change enhanced over time. There is a high confidence of wetter winters/drier summers for the end of century. There is high confidence that rainfall extremes will become more extreme in winter by end of century and less extreme in summer. Note, however, that the latter results contradict the UKCP18 simulations driven by their convection permitting model (CPM) which indicate that summer rainfall extremes will become more extreme under the RCP8.5 emissions scenario. Simulations of summer rainfall extremes by the CPM are likely to be more reliable.

There is high confidence that PET will rise throughout 21st century (Figure 27), with mean month monthly PET increasing between April and October. PET increases are highest in summer and PET extremes (95th percentile) will get more extreme.

LTA recharge is projected to decrease on average across RCMs, but the direction of change is uncertain (Figure 28). Winter recharge is projected to increase on average across RCMs decrease for other months. Recharge extremes (95th percentile) are projected to increase in winter.

Groundwater levels are projected to fall on average (across RCMs) throughout the 21st century for all months of year, but the direction of change is uncertain (Figure 29). Groundwater level extremes (5th/95th percentiles) are projected to fall over 21st century, but the direction of change is uncertain.

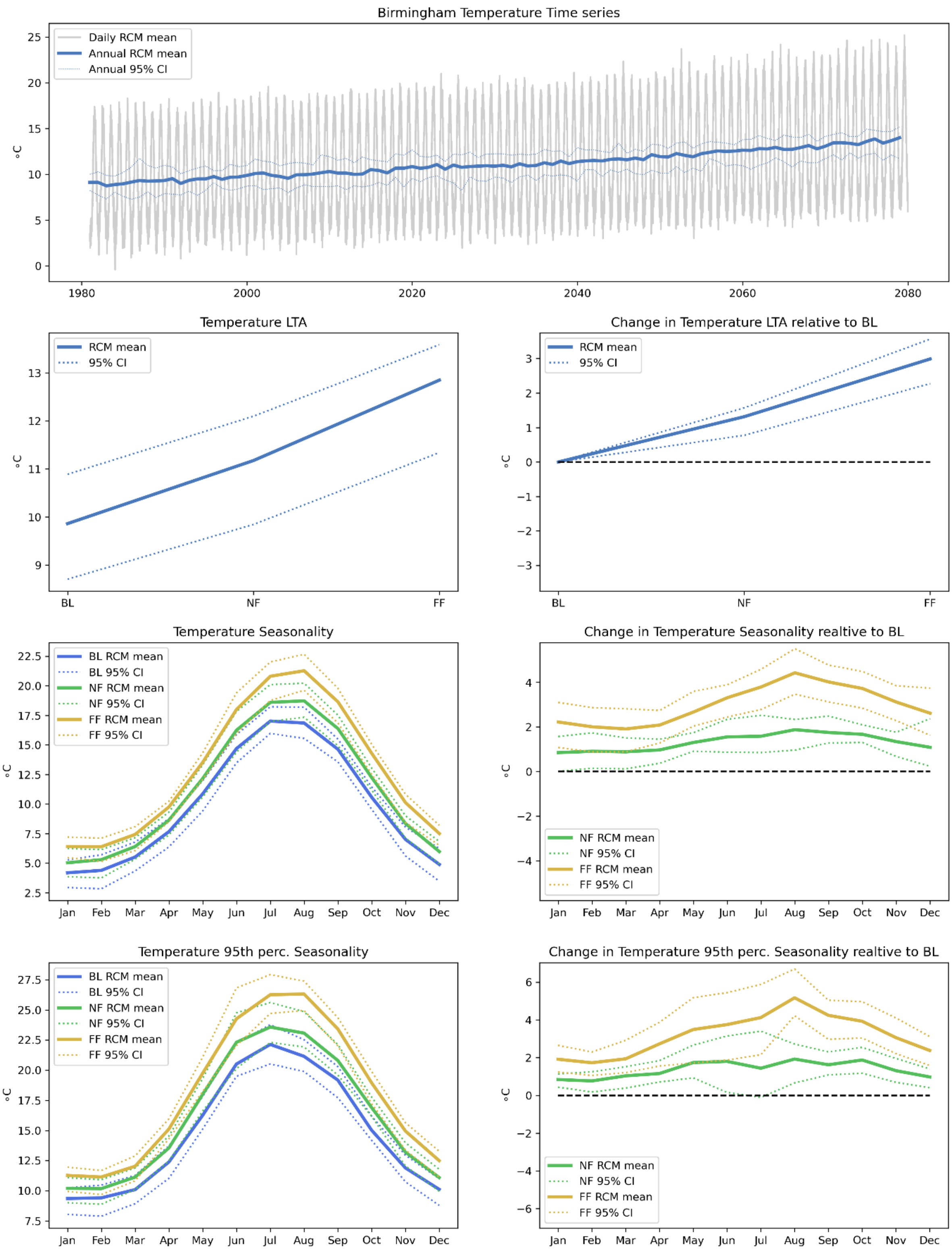

Figure 25 As Figure 15 but for temperature at Birmingham

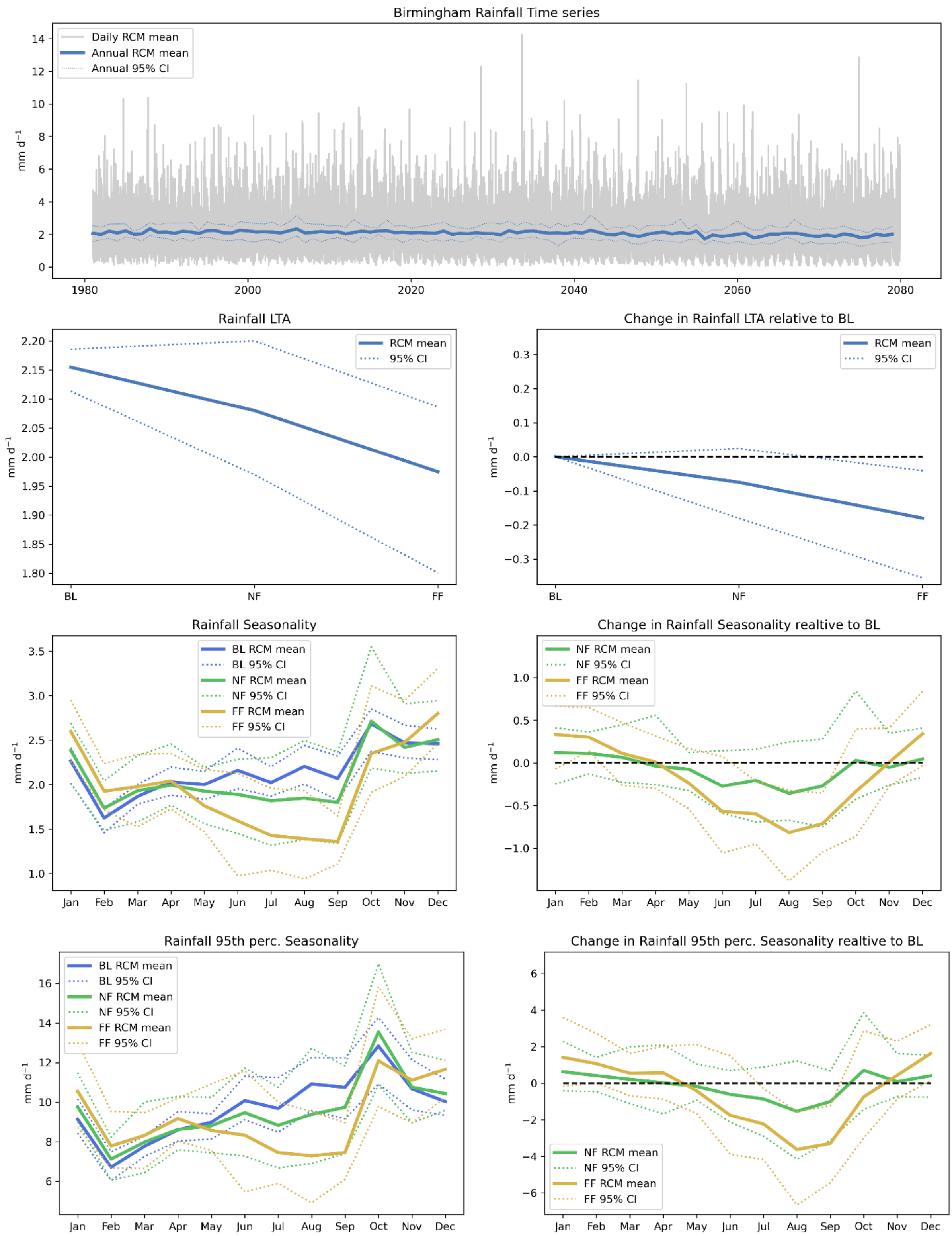

Figure 26 As Figure 15 but for rainfall at Birmingham

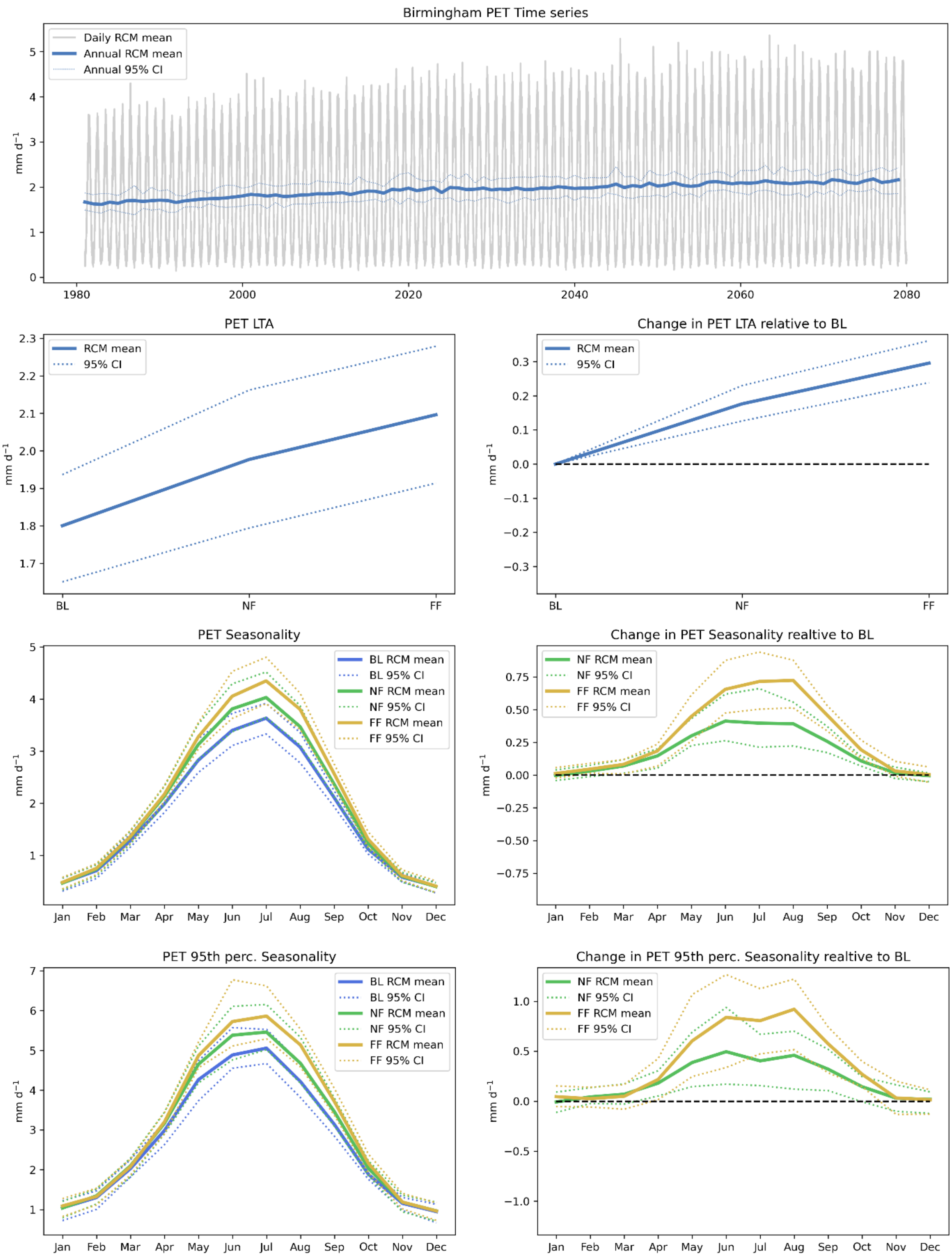

Figure 27 As Figure 15 but for PET at Birmingham

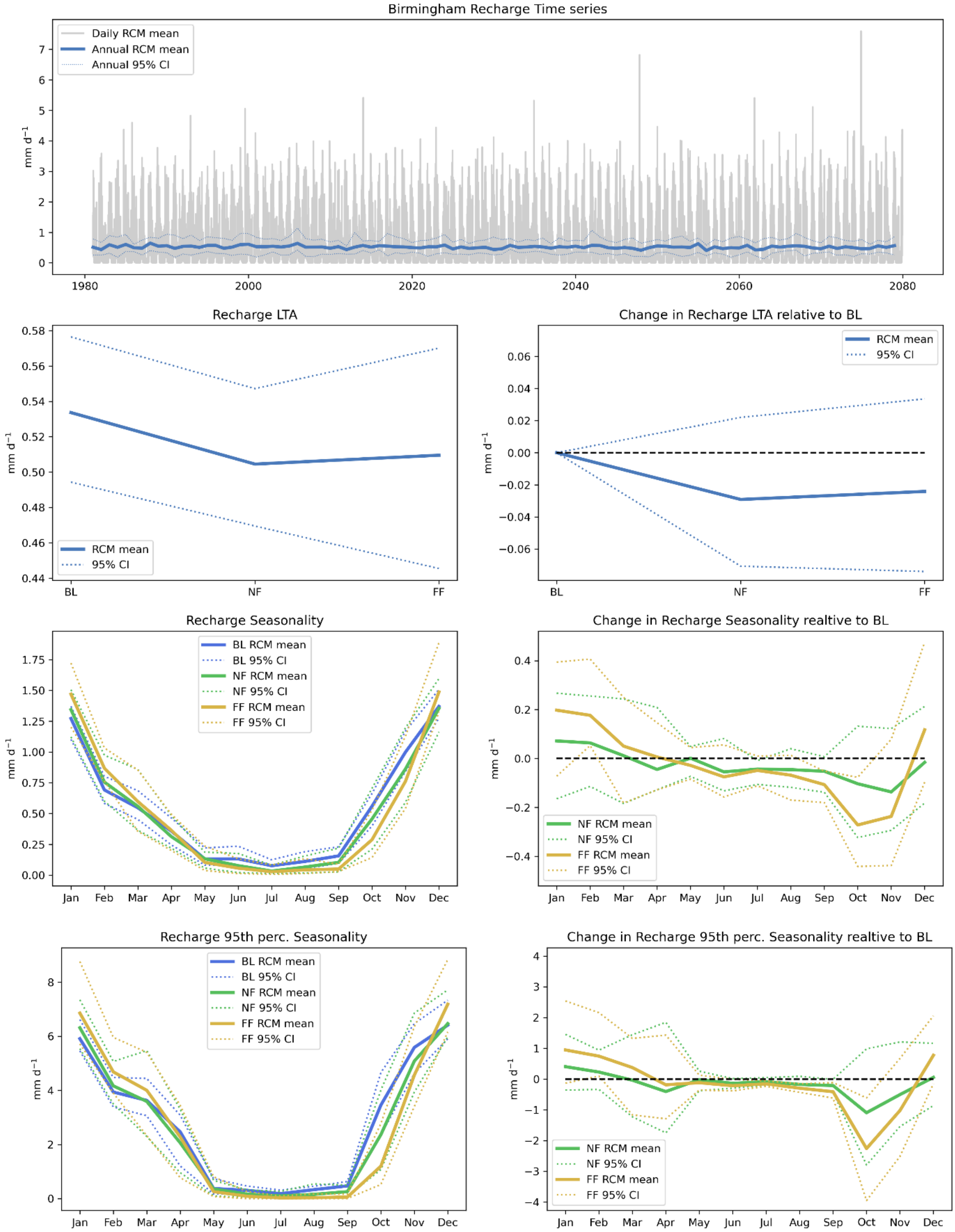

Figure 28 As Figure 15 but for groundwater recharge at Birmingham

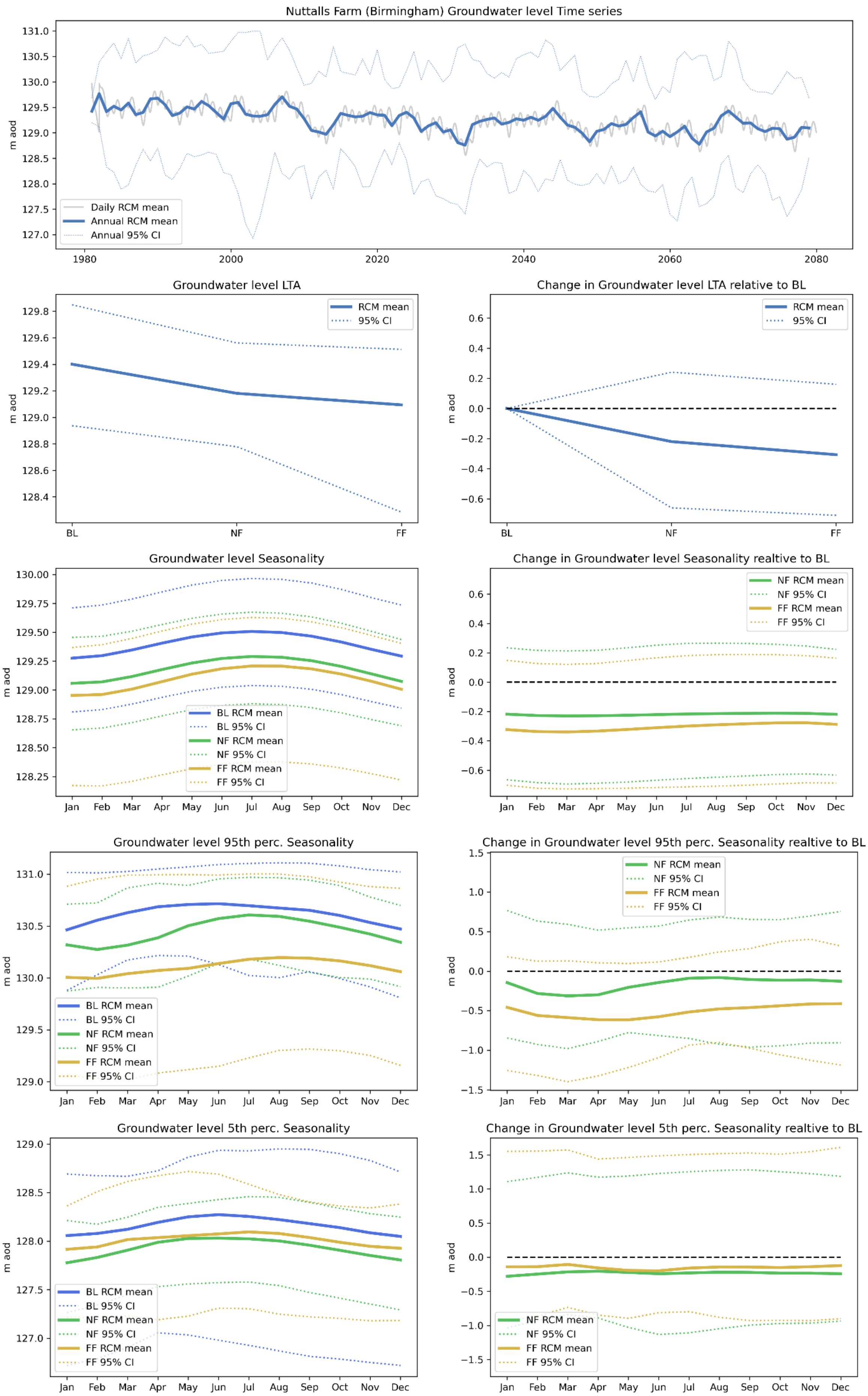

Figure 29 As Figure 15 but for groundwater levels at Birmingham (Nuttalls Farm)

4.3.10 Potential impacts of climate change on groundwater quality

Based on the groundwater quality issues for this case study described in sections 4.3.6 to 4.3.8 and the UKCP18 projections described in section 4.3.9, the following direct impacts of climate change on groundwater quality are anticipated:

1. Increased temperatures of up to 3°C could increase reaction rates for degradation of nitrate, pesticides and chlorinated solvents, but this may only be marginal.
2. Although the long term average rainfall and recharge are predicted to decrease, the direction of change is uncertain and extreme winter rainfall and winter recharge will increase. More extreme rainfall events in winter could increase the number of recharge events, and allow contamination from surface soils to enter shallow aquifers by downward percolation through the zone of aeration, and by induced recharge from surface water bodies increasing mobilisation of contaminants and producing more nitrate, metal and chlorinated solvent spikes. However, this may be offset by greater dilution.
3. Drier summers could lead to less dilution and more concentrated recharge of pollutants.
4. Although changes in climate are overall likely to decrease water levels, the rise in water levels due to recovery from long term over-abstraction will continue. Knipe et al (1993) describe the potential for water pollution due to groundwater level rise increasing nitrate, chloride, sulphate and organic compounds such as chlorinated hydrocarbons. Even if the recharge of pollutants could be eliminated, it would take decades for them to be flushed out from the sandstones. When water levels rise, pollutants can reach the aquifer more quickly with less attenuation or opportunity for adsorption than previously.

The Environment Agency's Birmingham numerical groundwater model could be used to investigate some of these issues.

Potential changes of land use (whether directly or indirectly associated with climate change) may have more influence on pollution than other changes in climate.

4.4 CASE STUDY 3 - DOVE

4.4.1 Setting

This case study area is the Dove Carboniferous Limestone Water Framework Directive Groundwater Body. The area extends from just south of Buxton in the north, to north of Ashbourne in the south, and from Mixon in the west to near Ashbourne in the east. It is drained by the River Dove and its tributaries, the Rivers Hamps and Manifold. The Bentley Brook and Henmore Brook that drain parts of the south-east of the area, join the Dove outside the study area. The River Dove is flashy due to rapid runoff from the northern highlands, with 50-60% of the total flow from groundwater (Chisholm et al, 1988). During periods of low flow, the lower River Hamps and stretches of the River Manifold (in the south-west of the area) disappear underground, for a total length of 14 km.

The highest points are High Edge in the north at 462 m, and Blackshaw Moor in the west at 467 m; the lowest elevation is about 122 m north of Mapleton where the Dove flows off the limestone onto Sherwood Sandstone. Rainfall varies from 890 mm in the south to 1270 mm in the north, with between 30 and 100% of this available as effective rainfall to infiltrate into the limestone (Edmunds, 1971).

The terrain is well-drained and dissected by both dry and river valleys. The soils are calcareous and generally poor, predominantly creating grazing land for sheep and cattle but with some arable farming. There is extensive historical and operational quarrying of limestone.

4.4.2 Geology

The area mainly comprises karstic Carboniferous Limestone at outcrop (Table 9, Figure 30), but this is overlain by low permeability Widmerpool Formation in the west and south-east. Two contrasting lithofacies can be distinguished in the Carboniferous Limestone, the carbonate-platform (shelf) facies of mainly shallow water deposits comprising thick, uniform, extensive beds of bioclastic and peloidal grainstones and packstones, that may contain lava interbeds; and carbonate-ramp (off-shelf) facies of deep water deposits predominantly of thinly bedded bioclastic packstones interbedded with shales. A third reef facies exists between the shelf margins and deeper water deposits, comprising poorly bedded micritic limestone with corals and bioclasts.

The Carboniferous Limestone is locally overlain by the Brassington Formation (deposits trapped and preserved by collapse into solution cavities in the limestone and dolomite), particularly in the east of the area. These are possibly Triassic to Palaeogene in age and comprise siliceous pebbly sands (up to 35 m) overlain by coloured silts and clays (up to 40 m) and grey clays (up to 6 m). They reach a maximum thickness of 67 m at Kenslow (outside the area), and over 55 m have been proved at Longcliffe (SK 2288 5573). The bedrock is locally covered by alluvium (silt or silty clay, commonly overlying sand) along the river valleys, with some head (silty loam with variable amounts of chert) and till (pebbly clay) (Figure 31).

Table 9 Stratigraphy of Dove Carboniferous Limestone Groundwater Body

Group	**Formation (offshelf-south)**	**Formation (high shelf-north)**
Millstone Grit Group	Up to 100 m mudstones and siltstones with some quartzitic sandstones	
Craven Group	Widmerpool Formation, up to 250 m dark grey thinly bedded turbiditic mudstones and limestones with some sandstones (Onecote Sandstones) and volcanics (Tissington Volcanic Member, up to 44 m hyaloclastite)	
Peak Limestone Group	Ecton Limestone Formation, 200-250 m grey well-bedded limestone	Eyam Limestone Formation, up to 54 m massive reef limestone over thinly bedded, dark grey, cherty, bioclastic limestone
		Monsal Dale Limestone Formation, 100 m grey, thickly bedded calcarenites, partly dolomitised
	Hopedale Limestone Formation, 60-300 m grey, thin-bedded limestones with knoll reefs	Bee Low Limestone Formation up to 300 m pale limestones with apron-reefs, partly dolomitised
	Milldale Limestone Formation, 300-470 m mainly dark grey thinly bedded limestones with large knoll-reefs locally	Woo Dale Limestone Formation, up to 100 m dark and pale limestones and dolomites

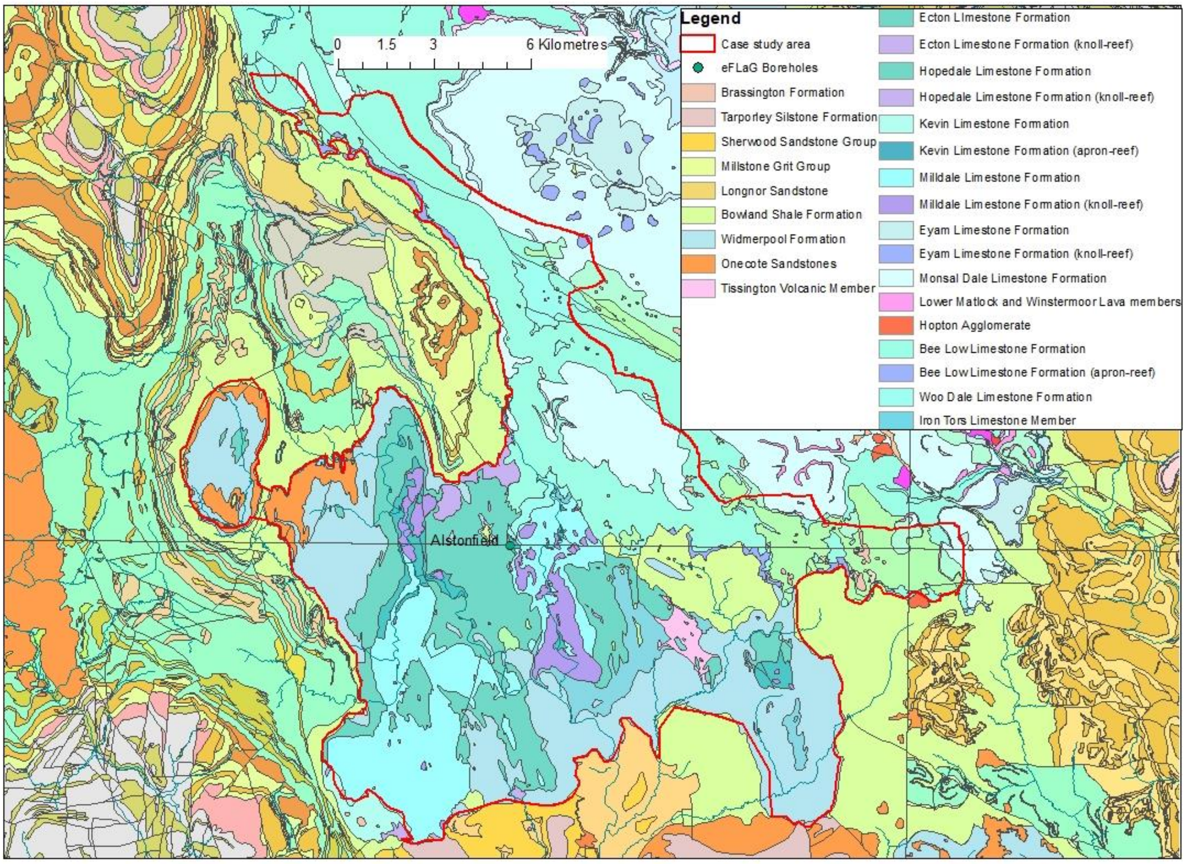

Figure 30 Bedrock in the Dove Carboniferous Limestone Water Framework Directive Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

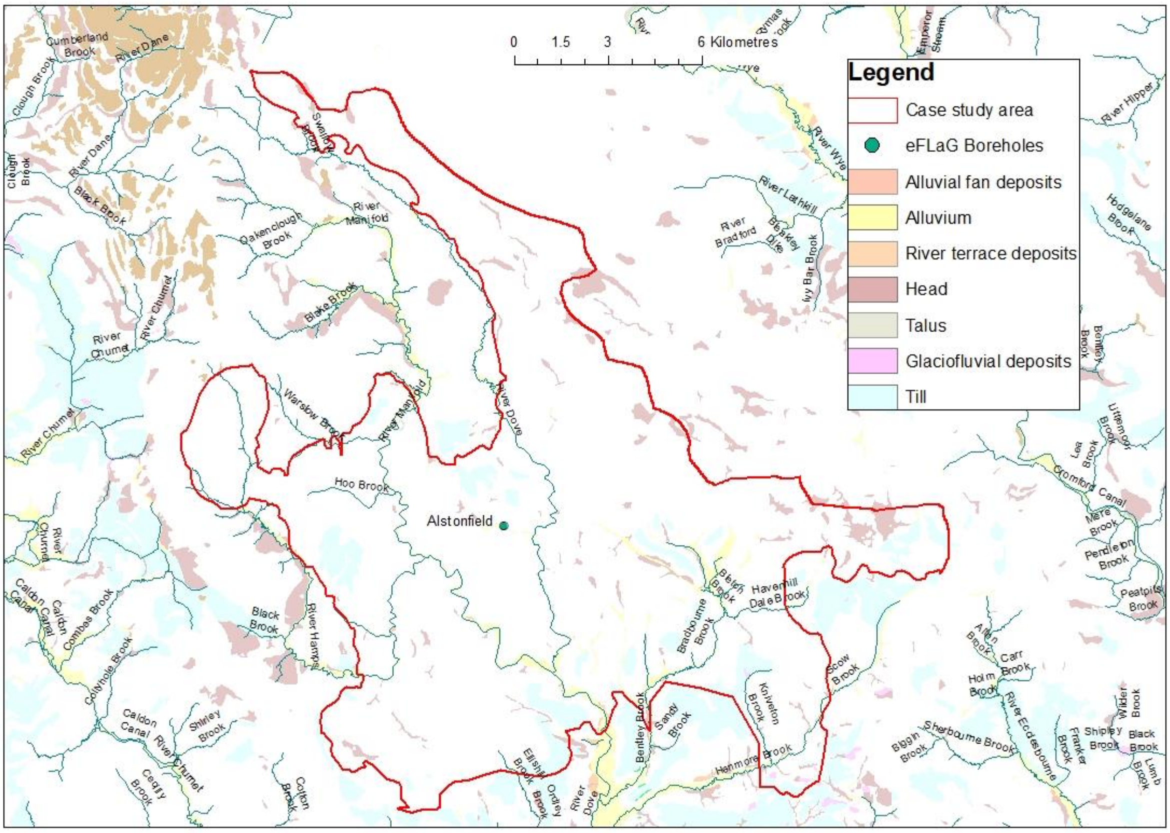

Figure 31 Superficial deposits in the Dove Carboniferous Limestone Water Framework Directive Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

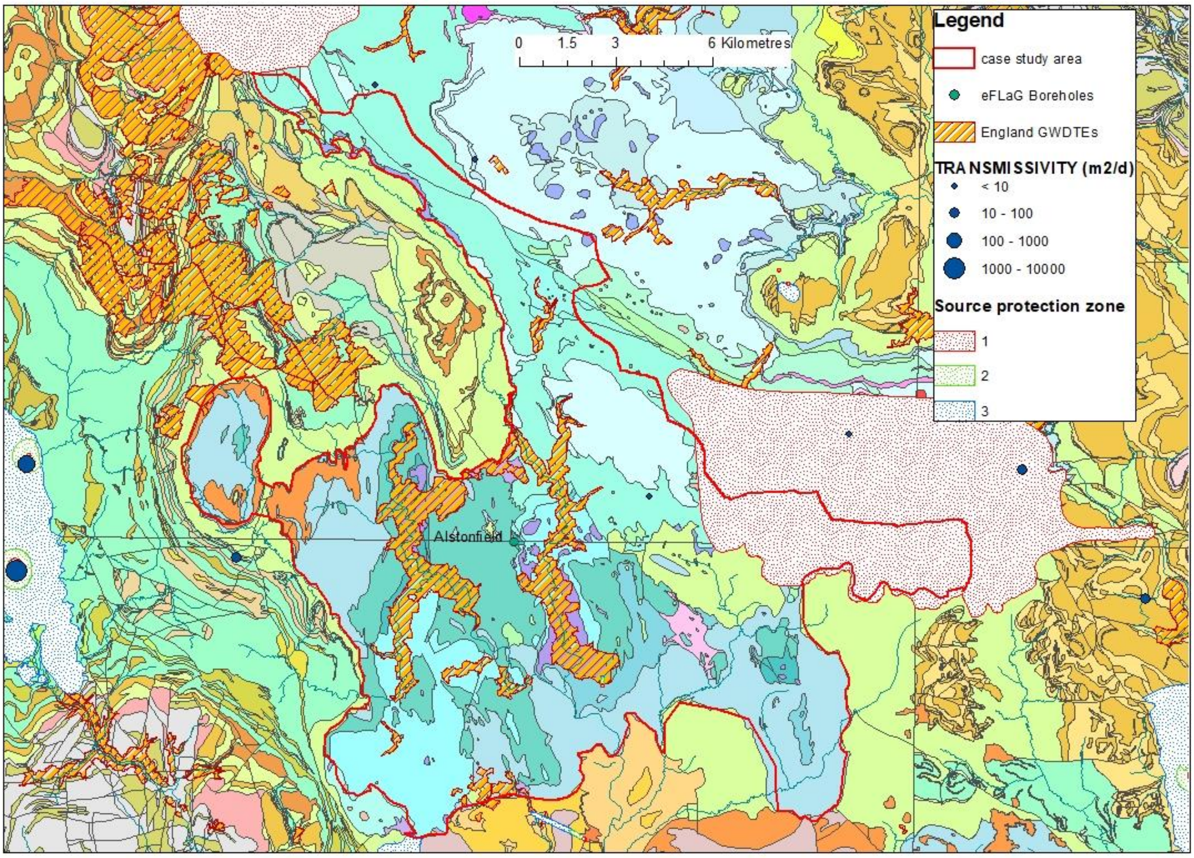

Figure 32 Various hydrogeological parameters in Dove Carboniferous Limestone Water Framework Directive Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0. Source protection zones © Environment Agency copyright and/or database right 2016.

The Carboniferous Limestone is mineralised, but less so in the Dove catchment than in the adjacent Derwent one to the east. The most important ores are galena (PbS) and sphalerite (ZnS), with fluorite (CaF_2), barite ($BaSO_4$) and calcite ($CaCO_3$) as gangue minerals. There is a progression from calcite, to barite to fluorite from west to east, related to an increase in the hydrothermal gradient. Copper ores are associated with the lead and zinc and found around Ecton. The main ore deposits occur as 'rakes', fissure fillings often 4.5 m or more wide, following the main east-west structural trend along the line of pre-existing faults and joints. The Carboniferous Limestone of the Derbyshire Dome is an anticlinal structure.

The Carboniferous Limestone can have a $CaCO_3$ content of over 98.5%. Dolomitisation is widespread and believed to be due to magnesium-rich downward percolating groundwaters in the Permo-Triassic (Aitkenhead et al, 2002) or possibly an early phase of mineralisation (Ford, 1999). Silica is common in some limestone formations, (occurring as chert, authigenic quartz, silicified fossils or quartz rock) but absent in others (eg reef facies).

4.4.3 Hydrogeology

Groundwater flow in the Carboniferous Limestone is controlled by topography, the physical properties of the rocks, geological structure and lithology as well as anthropogenic mining. Primary porosity (0.001-1%) and intergranular hydraulic conductivity (0.001-0.01 m/d) are very low (Gunn, 1992) and groundwater movement is almost entirely via solution-enhanced joints, fractures and bedding planes. It forms a regionally significant aquifer with the hydrogeology dominated by both natural karstic features and anthropogenic karst developed as a result of extensive historic mining creating a network of mine passages and drainage adits ('soughs'). However the former mines in the Dove catchment are more localised than the extensive interconnected mines in the Derwent catchment, although the normal limestone flow pathways are likely to be enhanced by mine workings around Brassington. The karst development has been aided by runoff from the area surrounding the limestone, comprising less permeable Namurian rocks. Rapid flow velocities of 100-500 m/hr through conduits have been measured (Gunn, 1992), but these vary depending on hydrological conditions. Allen et al (1997) contains a single transmissivity value of 4 m^2/d (Figure 32). Edmunds (1971) published groundwater levels for the area, which indicate a general southwards direction of flow following the River Dove, which is more south-easterly in the catchments of the Bentley and Henmore Brooks.

Thermal waters tend to issue around the margins of the limestone and younger Millstone Grit. Banks (1997) suggested the springs represent discharge points for long, deep flow paths through the limestone, controlled by fault structures. Brassington (2007) suggested they were caused by the formation of convection cells induced by a combination of a large thermal gradient between the top and bottom of the limestone and significant variations in bedrock thermal properties. Gunn et al (2006) concluded that the driving of the adits lowered water levels and caused upconing of deeper, thermal waters in the Matlock area (in the adjacent Derwent catchment) resulting in the discharge of thermal waters from mine adits such as the Meerbrook Sough. The main thermal springs are outside this area, but a spring source at Beresford (stated in Edmunds (1971) as SK 128 586, but shown on the site map as SK 1306 5842) had a temperature of 13.8°C, a few degrees above the mean annual air temperature.

There are no groundwater abstractions for public supply in the area. However, part of the source protection zone for two sources lie within this area; although the abstractions are located in the Derwent catchment (Figure 32).

4.4.4 Groundwater dependent terrestrial ecosystems

The limestone strata contribute to three SSSIs (Hamps and Manifold valleys, Dove Valley and Biggin Dale and the dry valley of Long Dale, Hartington) being classed as groundwater dependent terrestrial ecosystems.

4.4.5 Groundwater quality

Groundwater from the Carboniferous Limestone is usually of good quality, with a total hardness 250-350 mg/l (as $CaCO_3$), mainly carbonate hardness. Where the limestone is heavily fissured, water infiltrates rapidly giving little contact time with the rock and total hardness can fall to less than 100 mg/l. Chloride concentrations are below 20 mg/l, and fluoride can approach 0.5 mg/l. (Chisholm et al, 1988). The karstic nature of the aquifer leads to rapid flow of water and the aquifer is therefore vulnerable to contamination from the surface. Therefore after heavy rainfall, the water can become turbid and polluted, with high concentrations of suspended solids, organic matter, bacteria and nitrate.

Bedrock lithology and mineralisation are the dominant controls on groundwater composition. Groundwaters are buffered by near neutral pH (median 7.34), although pH is lower if they contain a significant proportion of Millstone Grit water, the rocks at surface below the headwaters of many of the streams outside the case study area.

The groundwater chemistry is dominated by calcium, magnesium and bicarbonate, reflecting the prevalence of limestone, with mineral solubility a strong control on the distribution of barium, fluoride, calcium, magnesium, silicon and bicarbonate as groundwaters are generally saturated with respect to the associated solid phases (barite, fluorite, calcite, quartz and dolomite) (Abesser and Smedley, 2008). Magnesium is generally low, but higher in the south-east where dolomitization is widespread. Sulphate increases eastwards with mineralisation and is often related to the presence of thermal waters. In shallow groundwaters, high sulphate, relates to the dissolution of barite and oxidation of sulphide minerals (galena, sphalerite or pyrite).

There is little systematic pattern in the distribution of trace elements, although nickel, lead and zinc appear to be higher in the eastern more lead-zinc mineralised areas of the Derwent catchment. Lithium is highest in thermal waters and those from the Millstone Grit. Iron is low (median 5 μg/l) and strongly controlled by redox conditions in the aquifer (with reducing waters mainly present in Millstone Grit or where the Carboniferous Limestone is confined by the Millstone Grit). Iron oxides are observed along fractures in Millstone Grit shales and gritstones and in the more oxidising limestone aquifer (Abesser and Smedley, 2008). Manganese is also controlled by redox conditions and bedrock lithology.

Non thermal waters from this less mineralised area have lower strontium (strontium is a constituent of barite which is a gangue mineral) and relatively constant Sr/Ca ratio suggesting strontium is mostly derived from the congruent dissolution of calcite, most waters are undersaturated with respect to strontianite but saturated with respect to calcite. In this area fluoride is less than 0.4 mg/l. Barium is higher in the central area probably related to barite gangue mineralisation, Barium is strongly controlled by sulphate concentrations and limited by barite solubility, with most groundwaters saturated with respect to barite. Where sulphate is low (<50 mg/l), barium can exceed the DWI drinking water limit of 100 µg/l (Abesser and Smedley, 2008).

At the periphery of the limestone and where confined by Namurian shales, there are increased concentrations of dissolved Si, Fe, Mn and As as a result of silicate mineral dissolution, pyrite oxidation, reduction of Fe/Mn oxides and denitrification. Variations in sulphate and chloride are also associated with the less mineralised Millstone Grit waters (Abesser and Smedley, 2008). Thermal waters occur around the periphery of the limestone outcrop and are enriched in most constituents (apart from NO_3), particularly Sr, Cl and SO4, compared with non-thermal limestone waters (Edmunds, 1971). Most noticeable are high Sr and Sr/Ca ratios which with enriched $\delta^{13}C$ signatures indicate enhanced water-rock interactions and prolonged residence times.

Mining has lowered the base drainage level and promoted upconing of thermal/deeper waters in parts of the aquifer, particularly in the Derwent catchment to the east, and provided artificially higher surface areas for enhanced bedrock weathering. There is an absence of toxic metals in the mine adits, Abesser and Smedley (2008) suggest possible causes being:

- the solubility of metals being limited by high alkalinity and pH of waters as well as mineral solubility controls (eg fluorite, barite);
- groundwater flow through these conduits being fast compared to the slow rate of sulphide oxidation;
- hydraulic equilibrium in the aquifer has been long established with limited water level fluctuations, it is possible most metals have been removed from the aquifer by prolonged weathering under oxidising conditions in particular in the zone of active groundwater flow. However in stagnant groundwater zones, limited oxygen flux may limit sulphide oxidation processes.

4.4.6 Agricultural pollutants

Nitrate concentrations are typically 4-6 mg/l (NO_3-N) across the Peak District, with slight bias at farm locations, where abstractions exist, influenced by local farm infrastructure. However, Abesser and Smedley (2008) state that anthropogenic nitrate appears to have doubled in the 40 years since 1967/8 and locally exceeds the drinking water limit of 11.3 mg/l (as NO_3-N). Dissolved phosphate is low and possibly from mineral sources rather than fertilisers. Later data from the EA database recorded nitrate of 28.9 mg/l (as N) in March 2014 in one borehole.

4.4.7 Climate change data outputs

Figure 33 to Figure 37 summarise the impacts of climate change on temperature, rainfall, PET, groundwater recharge and groundwater levels based on the RCP8.5 emissions scenario as derived from UKCP18 and it's application to the eFLaG models in the Dove. There is high confidence that temperature will rise throughout 21st century (Figure 33), with warmer temperatures on average for all months of year. Warming is greatest in summer (JJA) and Autumn (SON), and the hottest days will get hotter.

Long term average rainfall (Figure 34) is projected to decrease over time on average across UKCP18 RCMs, but the direction of change is uncertain. On average across the RCMs wetter winters and drier summers are predicted, with the change enhanced over time. There is a high confidence of wetter winters/drier summers for the end of century. There is high confidence that rainfall extremes will become more extreme in winter by end of century and less extreme in summer. Note, however, that the latter results contradict the UKCP18 simulations driven by their convection permitting model (CPM) which indicate that summer rainfall extremes will become more extreme under the RCP8.5 emissions scenario. Simulations of summer rainfall extremes by the CPM are likely to be more reliable.

There is high confidence that PET will rise throughout 21st century (Figure 35), with mean month monthly PET increasing between April and October. PET increases are highest in summer and PET extremes (95th percentile) will get more extreme.

LTA recharge is projected to decrease on average across RCMs, but the direction of change is uncertain and the magnitude of change is small (Figure 36). Winter recharge is projected to increase on average across RCMs decrease in summer months with high confidence. Recharge extremes (95th percentile) are projected to increase in winter and decrease in summer on average across RCMs.

There is high confidence that LTA groundwater levels will fall (across RCMs) throughout the 21st century (Figure 37). Late-winter groundwater level maxima are projected to increase on average across RCMs. There is high confidence that Summer/Autumn groundwater level minima will decrease. There is high confidence that late-winter groundwater levels extremes will increase, with potential increasing groundwater flood risk.

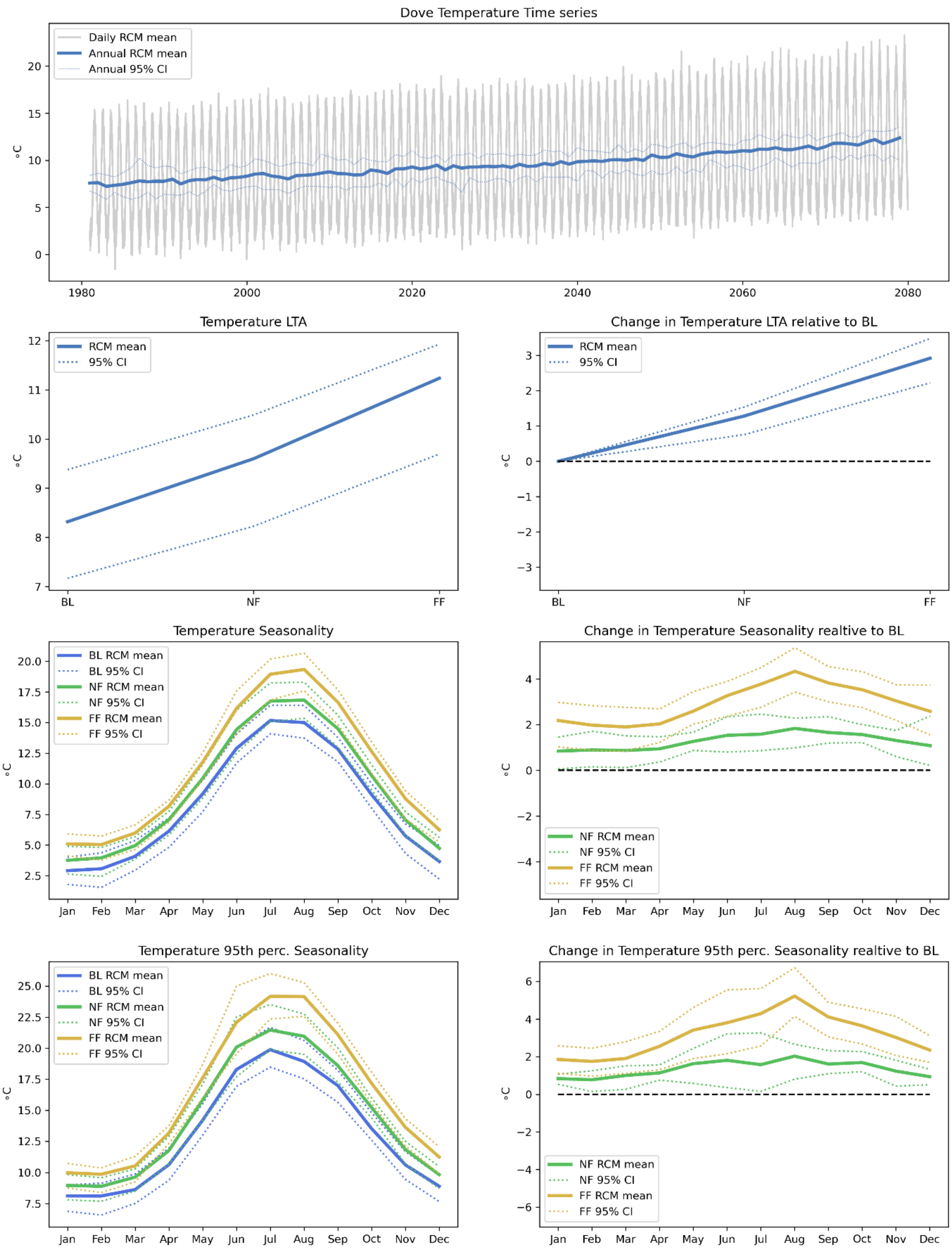

Figure 33 As Figure 15 but for temperature in the Dove

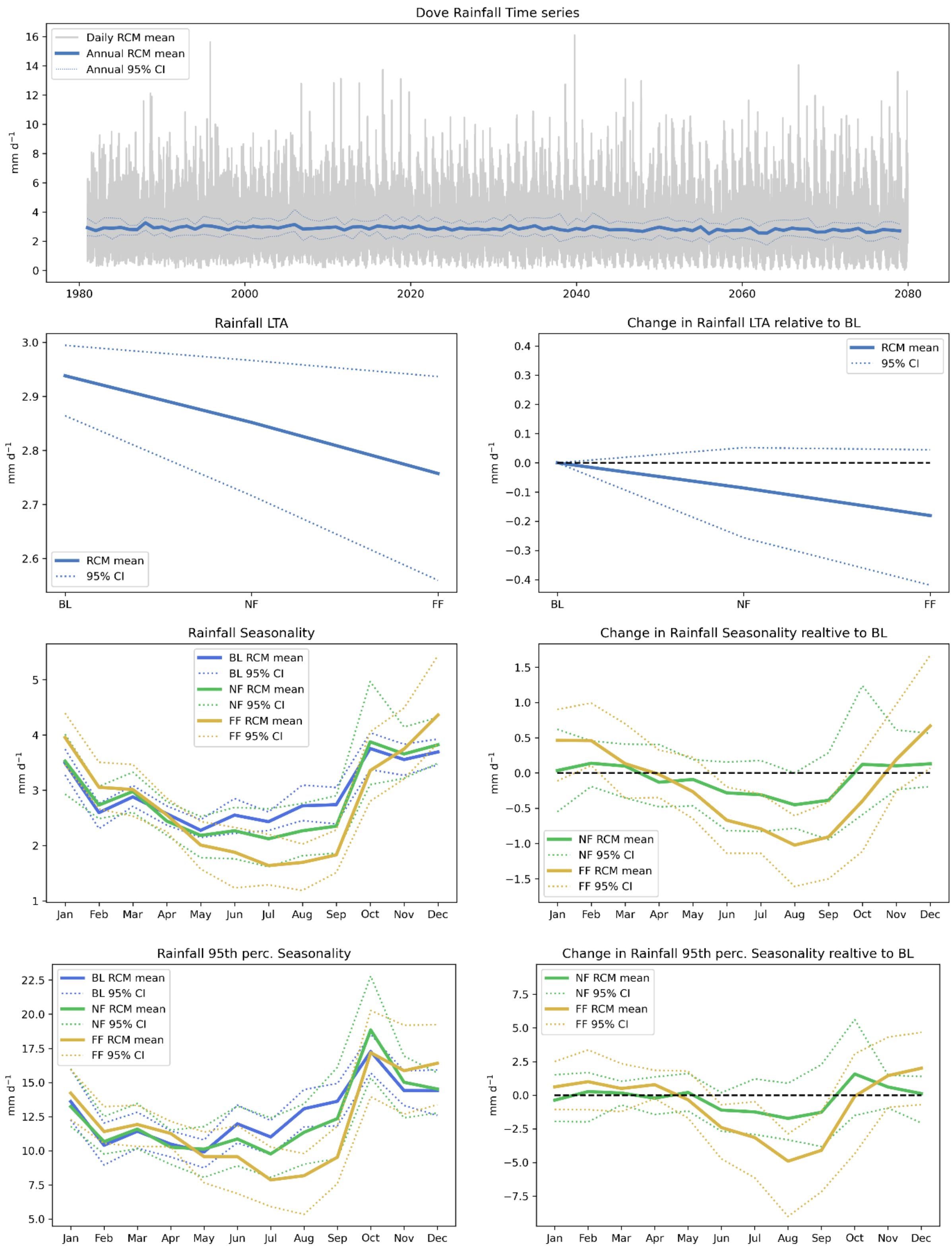

Figure 34 As Figure 15 but for rainfall in the Dove

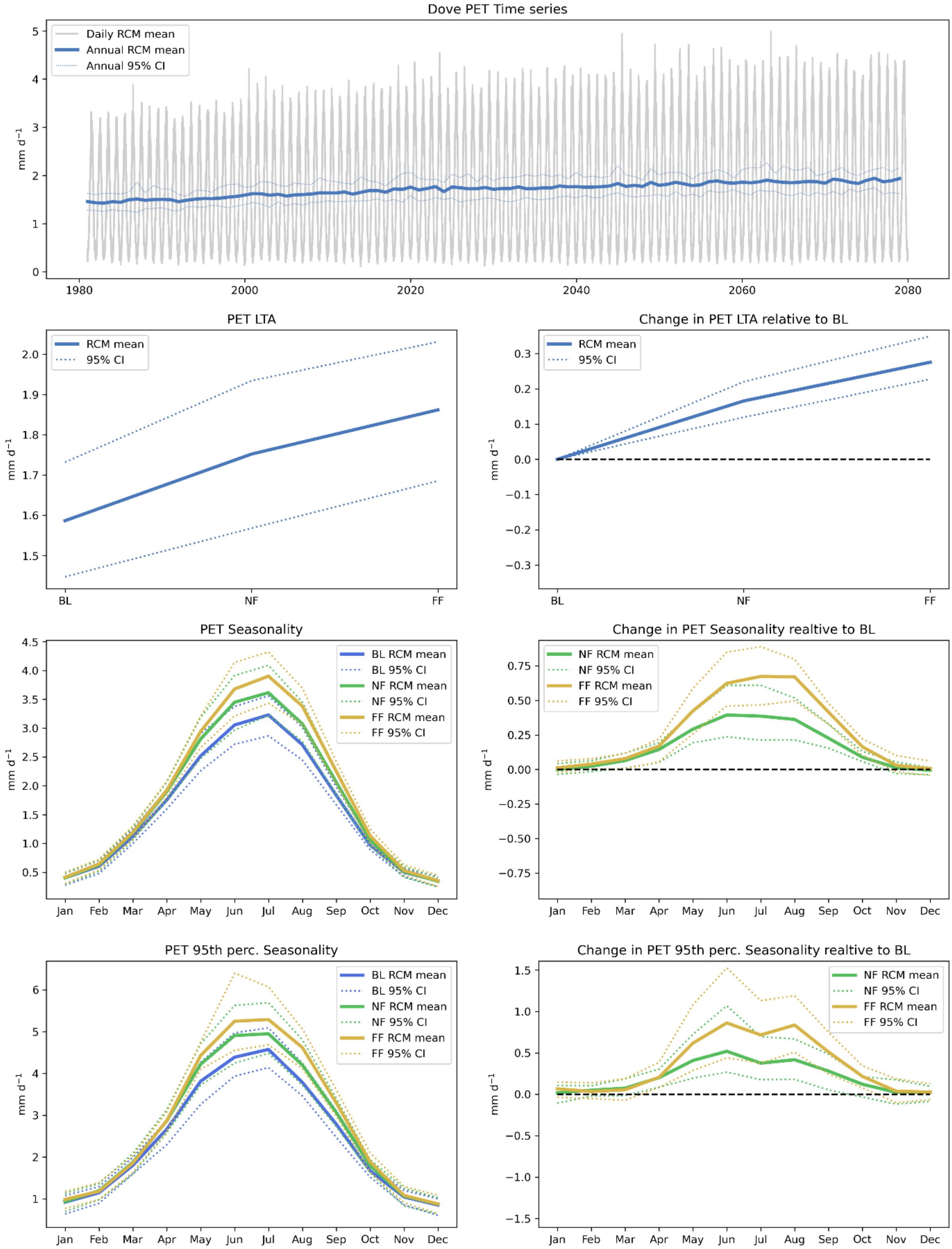

Figure 35 As Figure 15 but for PET in the Dove

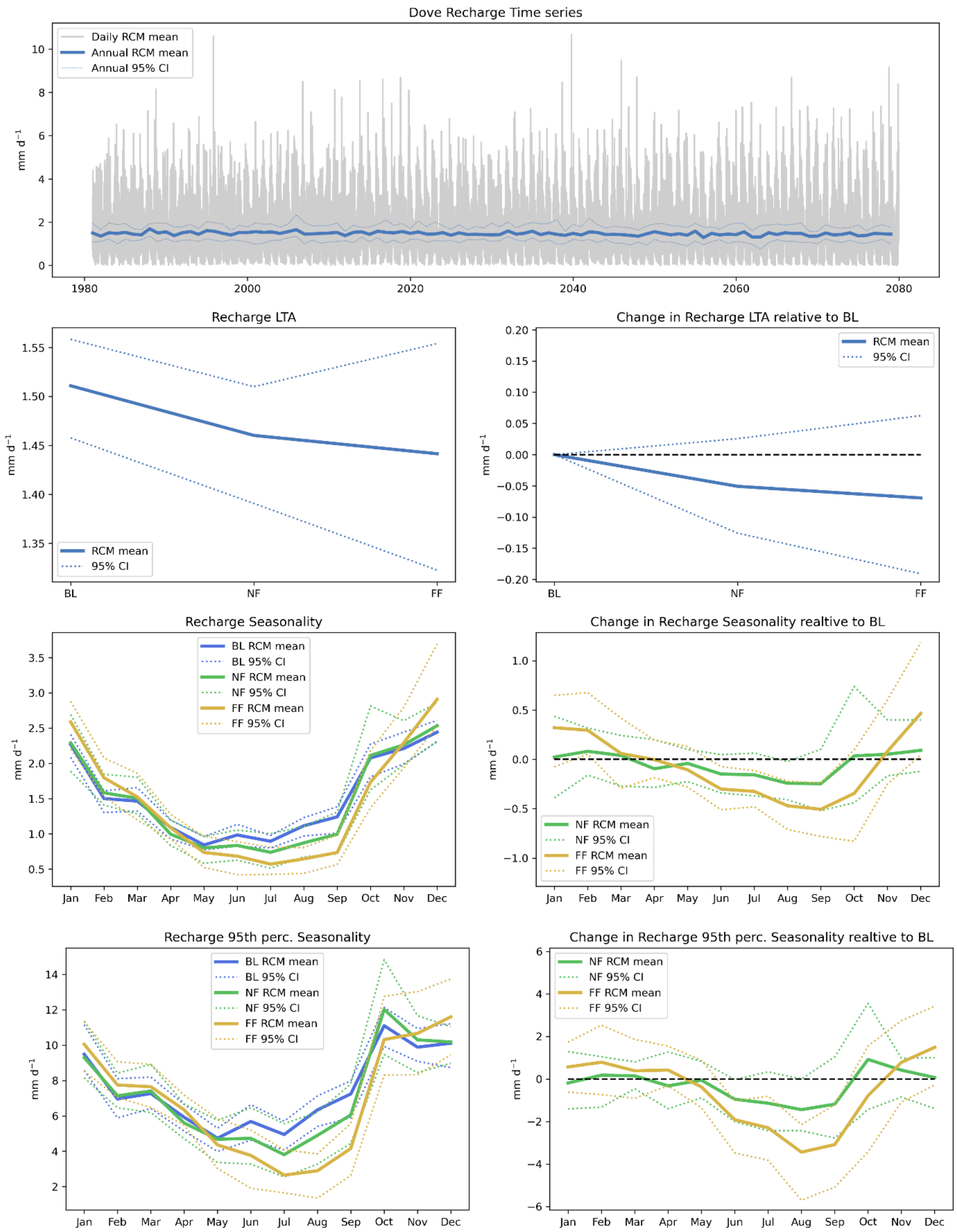

Figure 36 As Figure 15 but for groundwater recharge in the Dove

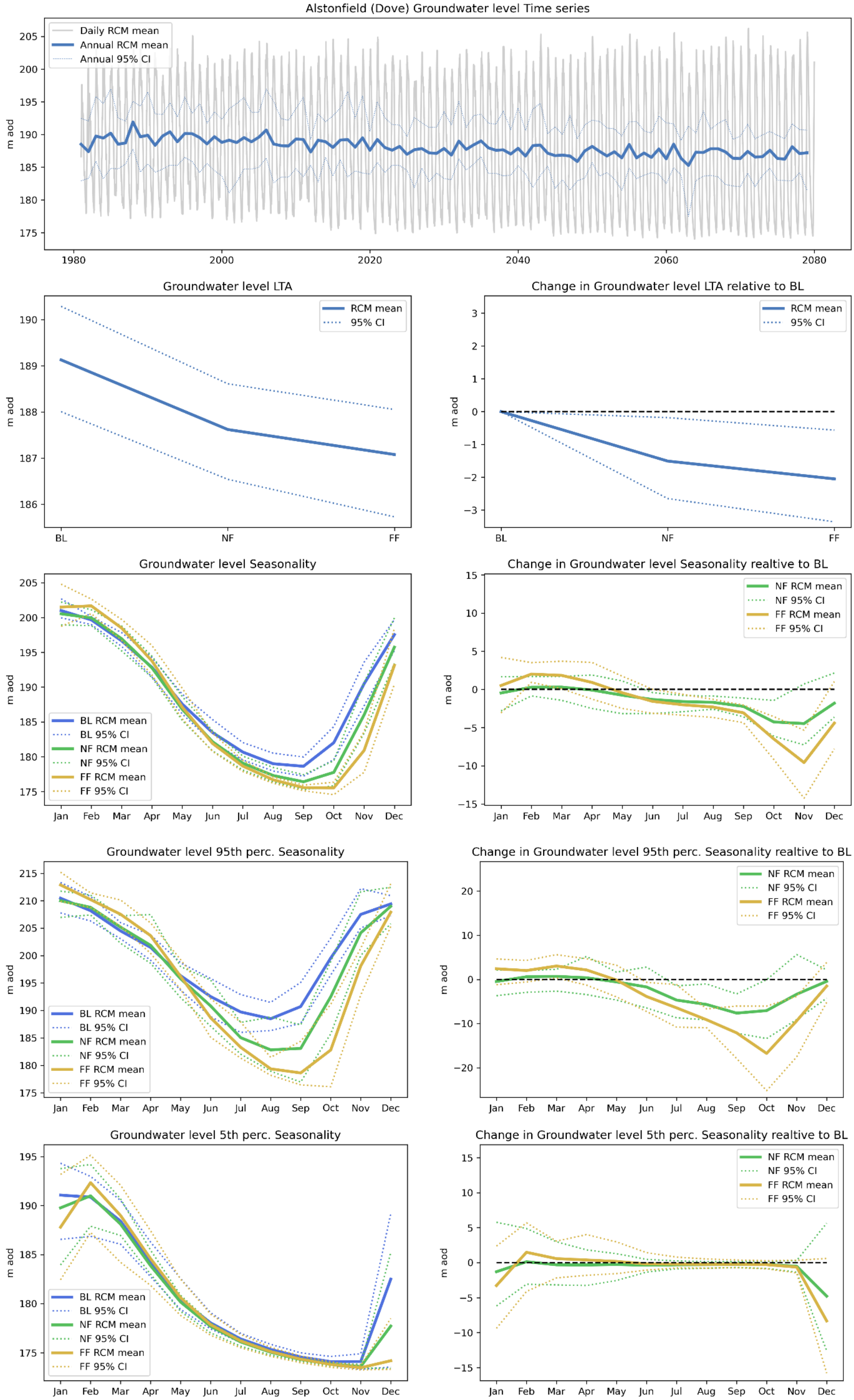

Figure 37 As Figure 15 but for groundwater levels in the Dove (Alstonfield)

4.4.8 Potential impacts of climate change on groundwater quality

Based on the groundwater quality issues for this case study described in sections 4.4.5 to 4.4.6 and the UKCP18 projections described in section 4.4.7, the following direct impacts of climate change on groundwater quality are anticipated:

1. Increased temperatures of up to 3°C could increase reaction rates for degradation of nitrate but this may only be marginal.
2. Increased winter rainfall and more extreme winter rainfall events are predicted to increase winter recharge into this highly karstic aquifer with poor soils and rapid fracture flow allowing little dilution or filtration. Hence increased numbers of turbidity, organic matter, bacteria and nitrate spikes may occur at this time of year.
3. Drier summers could lead to less dilution and more concentrated recharge of nitrate.
4. The decrease in long term average rainfall and fall in groundwater levels are likely to have a negative impact on the groundwater dependent terrestrial ecosystems. A decrease in the base drainage level could cause further upconing of thermal waters.

Potential changes of land use (whether directly or indirectly associated with climate change) may have more influence on agricultural pollution than other changes in climate.

4.5 CASE STUDY 4 - EDEN VALLEY

4.5.1 Setting

This case study area concerns the Eden to Great Crosby Groundwater Management Unit, forming the northern part of the Eden valley. It forms part of the Eden Valley and Carlisle Basin Permo-Trias WFD groundwater body. The area extends from Great Corby (south-east of Carlisle) in the north, southwards to the northern outskirts of Penrith. It varies in elevation from 25 m to 330 m and is drained by the northward flowing River Eden. The western boundary in the south is close to the M6. Average annual rainfall over most the Vale of Eden is 900-1000 mm/a with potential evapotranspiration of 425 mm/a (Arthurton and Wadge, 1981), but rainfall rises to 1500 mm/a over higher ground (Daily et al, 2006).

The main land use is managed grassland, with some arable, semi-natural vegetation and forestry/woodland (Daily et al, 2006).

4.5.2 Geology

The Eden valley is a mainly fault-bounded north-west trending trough between the older rocks of the Lake District (to the west) and the Pennines (to the east). The study area is defined by the outcrop of the Permo-Triassic sandstones (Table 10, Figure 38), and hence crosses river catchment areas, eg it includes the upstream part of the River Petteril catchment. The Permain age Penrith Sandstone and Triassic St Bees Sandstone are separated by the mudstones of the Eden Shales. The River Eden predominantly flows over the Penrith Sandstone outcrop or subcrop; except south of Langwathby where it crosses onto the Eden Shales; and in the north it flows over the St Bees Sandstone.

The Penrith Sandstone is aeolian and generally composed of well-rounded quartz grains often cemented with calcite and iron hydroxide minerals. The St Bees Sandstone is similar lithology but contains both feldspar and mica (Shand et al, 1997), and was deposited in a shallow fluvial environment.

The north of the area from Aiketgate to Renwick is crossed by the Paleocene age Armathwaite-Cleveland dyke (basaltic andesite). Superficial deposits cover the majority of the area, reaching thicknesses of up to 20 m (Figure 39). They comprise alluvium (sandy and silty loam with lenses of gravel) along the river valleys of the Eden and Petteril, with some associated river terrace deposits (mainly gravel), and elsewhere Devensian glaciofluvial sand and gravel and till (clayey sand with pebbles) deposits. There are also some areas of peat.

Table 10 Stratigraphy of Eden to Great Crosby Groundwater Management Unit

Group	**Formation**	**Description**
Sherwood Sandstone Group	St Bees Sandstone Member (Chester Formation)	up to 600 m unfossiliferous red sandstones with thin mudstone beds. Strongly cemented with combination iron oxide, calcite and silica
Cumbrian Coast Group	Eden Shales Formation with A, B, C and D beds	45-180 m red and grey mudstones and siltstones with 1-6 m thick layers of gypsum/anhydrite and at least one thin dolomite
Appleby Group	Penrith Sandstone Formation	0-300 m red-brown, mainly dune-bedded sandstones but locally contains water-laid breccia (brockram), flat-bedded sandstone, siltstone and mudstone
Pennine Coal Measures Group	Pennine Lower Coal Measures Formation	360 m mudstone and siltstones with layers of sandstone

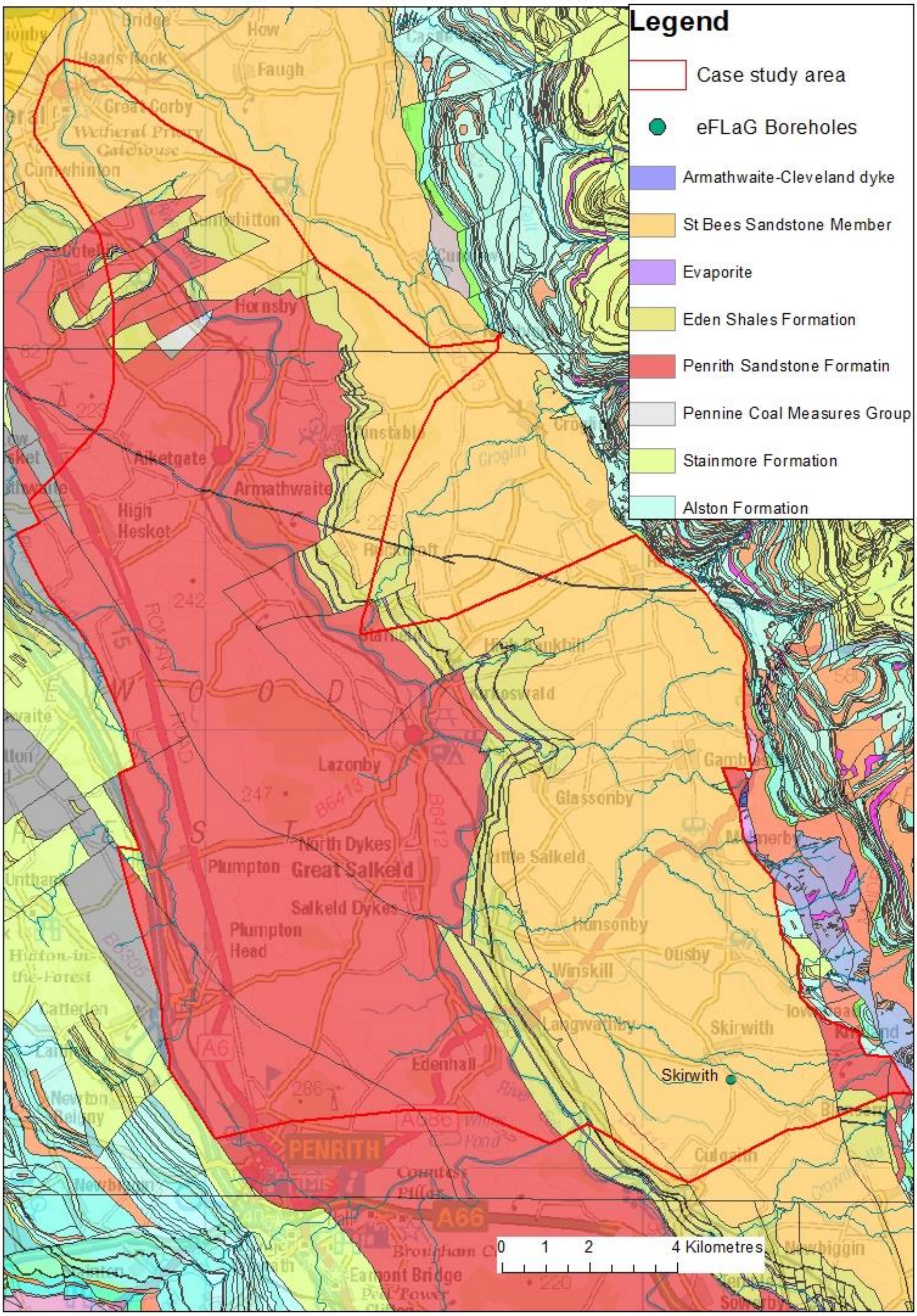

Figure 38 Bedrock in Eden to Great Crosby Groundwater Management Unit. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

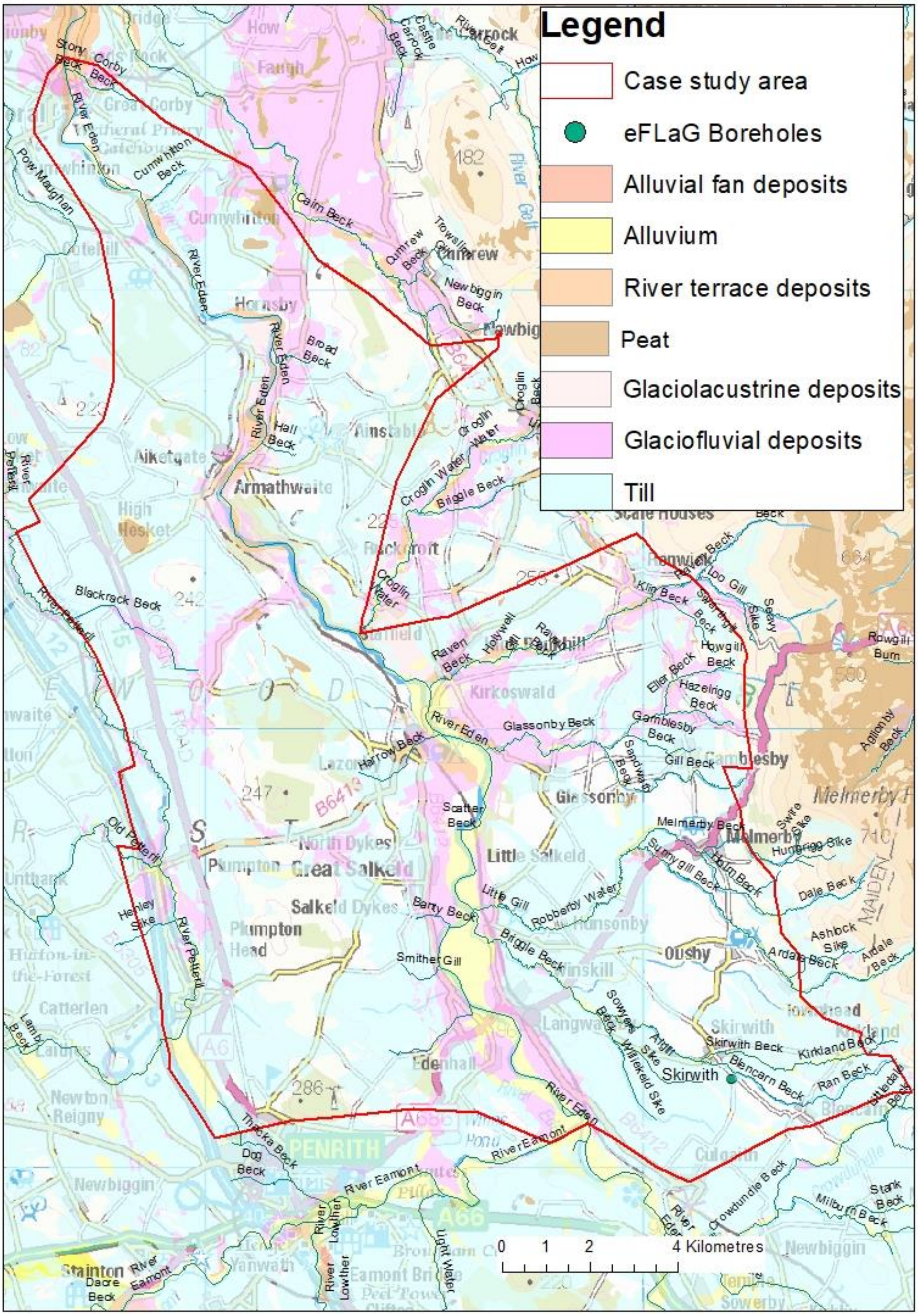

Figure 39 Superficial deposits in Eden to Great Crosby Groundwater Management Unit. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

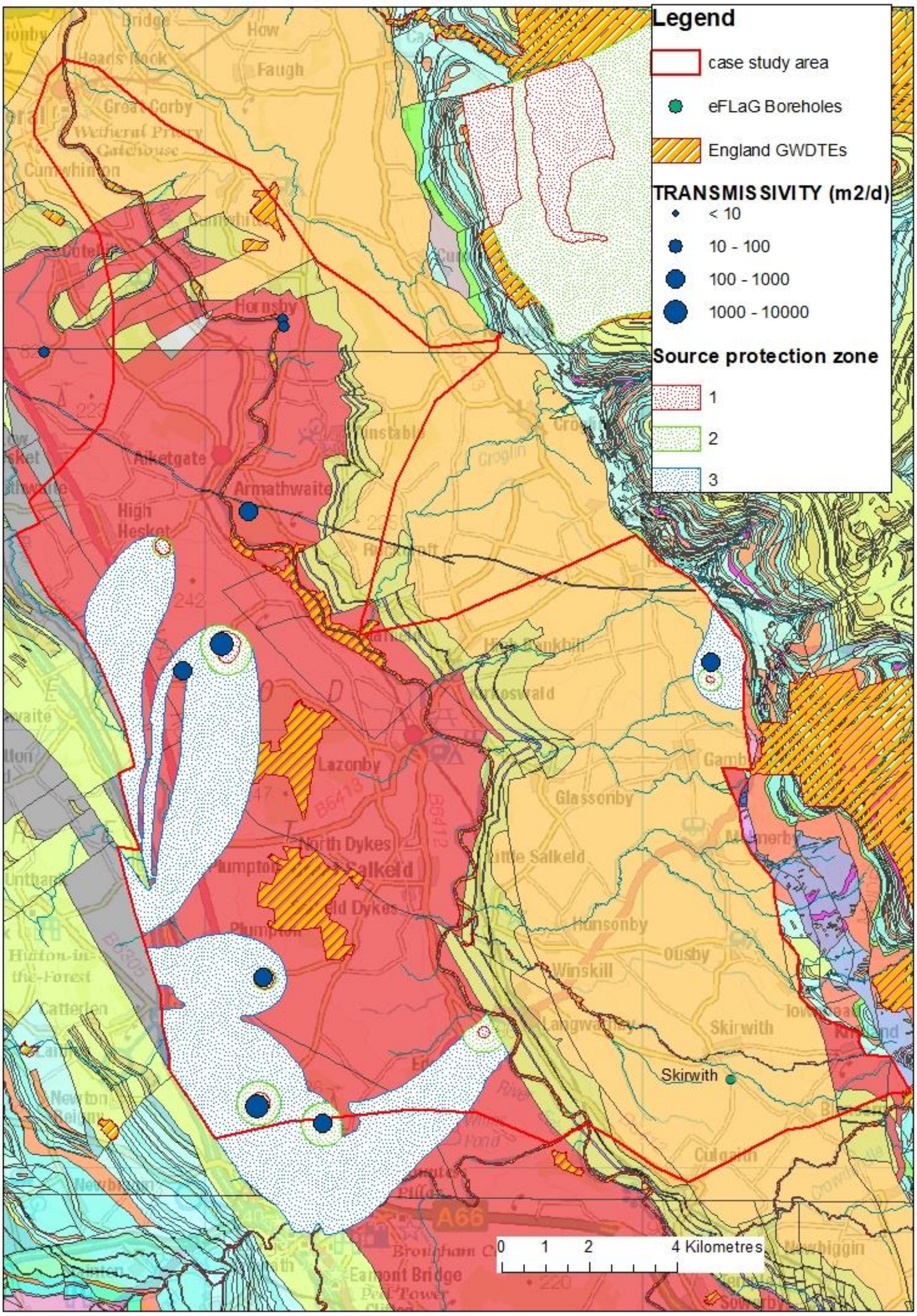

Figure 40 Various hydrogeological parameters in the Eden to Great Crosby Groundwater Management Unit. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0. Source protection zones © Environment Agency copyright and/or database right 2016.

4.5.3 Hydrogeology

The study area contains two aquifers (Penrith Sandstone and St Bees Sandstone) separated by the Eden Shales Formation aquiclude. The Penrith Sandstone primarily occurs in the west of the basin, but small outcrops occur adjacent to the Pennine Fault in the east around Gamblesby and Kirkland. It is variably cemented by silica (filling up to 70% of the pore space) forming areas of stronger relief (where the sandstone has been quarried for building stone) with low permeability and some perching of water levels, and less well-cemented areas of lower relief often covered by superficial deposits.

Recharge is primarily dependent on the nature and thickness of the superficial deposits, and hence is greatest where the sandstones are at outcrop (Daily et al, 2006). The Penrith Sandstone is covered by superficial deposits, with outcrops north of Penrith and on the watershed between the Petteril and Eden valleys. The St Bees Sandstone is also primarily covered by till but outcrops from Ousby to Melmerby.

Groundwater flow in the Eden valley sandstones is dominantly towards the River Eden, the river gaining over most of its length, although it is underlain by the Eden Shales south of Langwathby.

Seasonal water level fluctuations in the Penrith Sandstone are typically less than 1 m, reflecting the high storage. Hydraulic gradients are eastwards and generally low, but steepen to the north, reflecting the lower permeability of the upper 100 m of sandstone due to silicification. Lateral movement of groundwater across the Carboniferous/Permo-Triassic boundary may be significant (Allen et al, 1997; Daily et al, 2006), but this is likely to be greater further south where the Penrith Sandstone is directly underlain by Carboniferous Limestone, rather than in this area where the Stainmore Group or Coal Measures are present to the west.

Core data indicates significant anisotropy of the intergranular permeability created by cross-stratification and graded bedding, together with extremely irregular secondary silicification. Uncemented sandstones have intrinsic permeabilities of 10.9-14.8 m/d; whilst in more cemented strata they are one or two orders of magnitude lower. Intergranular horizontal to vertical permeability ratios are around 3, producing a bulk aquifer permeability exaggerated to 20-30 by the presence of sub-horizontal fractures (Arthurton and Wadge, 1981).

The St Bees Sandstone is of fluvial origin and well indurated, with the lowest 100 m the most cemented; the upper beds form the best aquifer where there are also fewer argillaceous partings. However, most groundwater flow is through secondary fractures, although the matrix controls the release of groundwater from storage to the fissure system, but silicified, fine-pore size zones inhibit free drainage to fissures and are capable of exerting capillary suction. The aquifer is strongly anisotropic (Arthurton and Wadge, 1981).

Hydraulic gradients are steeper in the St Bees Sandstone, reflecting its lower permeability. In the east of the valley, along the Pennine Fault, there may be some lateral movement of groundwater from the Carboniferous, however the numerous springs along this contact indicate that much of the groundwater becomes surface flow.

Transmissivity values in the Penrith Sandstone vary from 60 to 1900 m^2/d and for the St Bees Sandstone there is a single value of 230 m^2/d (Allen et al, 1997).

There are five abstractions for public supply and one for bottling, all from the Penrith Sandstone and located on the western side of the River Eden. There is also one public water supply from the St Bees Sandstone, east of the Eden.

4.5.4 Groundwater dependent terrestrial ecosystems

The headwaters of the River Eden drain the Yorkshire Dales, North Pennines and eastern fells of the Lake District. The river has therefore crossed Carboniferous limestones, sandstones and mudstones before reaching the Permo-Triassic sandstones, mudstones covered by glacial deposits and the sandstone gorge below Lazonby (River Eden and Tributaries and Eden Gorge SSSIs).

Wan Fell and Lazonby Fell SSSIs are examples of remaining lowland heathland on the Eden valley sandstones (that have not been converted to grassland and plantation woodland) with shallow well-drained podzolic soils.

Cumwhitton Moss SSSI is a peatland.

4.5.5 Groundwater quality

Shand et al (1997) state that the Eden valley groundwaters are mainly of calcium-bicarbonate or calcium-magnesium-bicarbonate type, but more evolved waters are present. Conductivities are generally low, although there are some high chloride and sulphate concentrations. Some sites show temporal variations in major ions. The dominant water-rock interactions controlling water chemistry include dissolution of minerals such as calcite, dolomite, gypsum/anhydrite or halite. It is likely the upper parts of the sandstones have been decalcified, leading to waters being undersaturated with respect to calcite. The presence of carbonate (particularly where brockram is present), helps buffer waters to neutral to alkaline pH. However, the fact that many waters are undersaturated with respect to calcite implies silicate weathering and dissolution of salts also exert an important control on major element chemistry. Calcite dissolution is likely to take place in deeper parts of the aquifer that have not undergone decalcification. Higher median silicon, lithium and Na/Cl ratios in some St Bees Sandstone waters indicate that silicate weathering has been more important (Shand et al, 1999). Of the major ions calcium and bicarbonate show the highest levels of enrichment compared with chloride relative to seawater (Shand et al, 1999). There is a trend of increasing pH and Mg/Ca ratios in deeper more reducing waters (Shand et al, 1997).

Waters from both aquifers are similar, but conductivities are higher in the St Bees Sandstone (presumed to be due to greater depths and longer residence times) with median sodium, chloride and bicarbonate concentrations slightly higher than in the Penrith Sandstone; pH is also slightly higher in the former, but sulphate is similar (Shand et al, 1999).

Groundwaters are generally oxidising with nitrate present and low iron and manganese, however in reducing, confined parts of the aquifer, nitrate is below the detection limit and iron is relatively high and can be extremely high from unfiltered samples, this is likely to be present as particulate matter (and can be very variable temporally). The redox boundary lies close to the confined/unconfined boundary (Shand et al, 1999). Trace metal concentrations are generally low reflecting their low concentrations in the rocks as well as the neutral to alkaline conditions in the aquifer, but there is often a correlation between high iron and manganese and relatively high trace metal concentrations such as lead and zinc (and copper). Fluoride concentrations are low reflecting a lack of fluorite in the aquifer (Shand et al, 1999). Some waters have relatively high barium (>400 µg/l) probably from feldspar or barite, but where sulphate is high, concentrations are kept low due to saturation with respect to barite. Strontium is often low implying little reaction with the rock. Bromide correlates poorly with chloride, and low Br/Cl ratios imply halite dissolution has been important in the evolution of many groundwaters (Shand et al, 1999).

Groundwaters in the Penrith Sandstone are mainly of calcium-bicarbonate type. Arthurton and Wadge (1981) provide representative values for Penrith Sandstone groundwaters of total dissolved solids 105-225 mg/l, pH 6.15-7.9. total hardness 50-180 mg/l (as $CaCO_3$) and chloride 14-21 mg/l. Daily et al (2006) state that where the sandstones are close to the junction with, or overlain by, the Eden Shales, groundwaters can have higher calcium and sulphate derived from the dissolution of gypsum, although this is not always the case. Where boreholes encountered brockram, magnesium and bicarbonate are higher. Iron and manganese are generally low, as the aquifer is unconfined and oxidising; but one site (east of Penrith), has mean concentrations of iron of 4.33 mg/l and manganese of 0.198 mg/l, above their PCVs of 0.2 mg/l and 0.05 mg/l, respectively (Daily et al, 2006). Arsenic in isolated samples has exceeded the quality limit of 10 µg/l, in the area just to the north.

Groundwaters from the St Bees Sandstones are mainly of calcium-magnesium-bicarbonate type. Where calcium and chloride are higher in the St Bees Sandstone this mostly relates to dissolution of gypsum and halite from the aquifer matrix (Shand et al, 1999). Several waters show a decrease in calcium and bicarbonate, with an increase in sodium at high pH, probably as a consequence of calcite precipitation and ion exchange (Shand et al, 1997). Iron and manganese are not elevated in the Eden valley (but are at some sites further north in the Carlisle Basin) (Daily et al, 2006).

4.5.6 Agricultural pollution

In the Penrith Sandstone of the Eden valley nitrate is generally high and rising, with Daily et al (2006) stating that the site (north of Plumpton) with the highest concentrations had a mean of 28.9 mg/l and a maximum of 30.9 mg/l (as N); many others exceed 11.3 mg/l where the sandstone is at outcrop or below thin superficial cover and intensive pastoral farming occurs. Locally where nitrate is high, chloride is also elevated (and in one case potassium (and south of this area orthophosphate)), also from agricultural sources, spreading of artificial fertiliser and slurry (Daily et al, 2006). More recent data from the EA database gives nitrate concentrations of 25.3 mg/l (as N) in February 2010 at site NW-88020695 and 31.2 mg/l (as N) in July 2016 at site NW-88020034.

Daily et al (2006) detected the herbicides simazine, desisopropyl atrazine and desethyl atrazine above the DWI limit in a single sample from one borehole in the Penrith sandstone (just south of the study area) but expressed uncertainty if this represents a widespread issue. Atrazine was detected at 0.0047 µg/l in July 2013 at site NW-88020695 and 0.0279 µg/l in July 2015 at site NW-88010231 (EA database).

In the St Bees Sandstone nitrate can also be high and is generally rising with Daily et al (2006) quoting a mean of 15.6 mg/l (as N) at one site. Locally where nitrate is high, chloride is also elevated (and in one case potassium and nickel), again from agricultural sources. Orthophosphate exceeded the quality level of 0.153 mg/l in the north-east of the Eden valley, probably due to agricultural pollution. More recent data from the EA database gives a nitrate concentration of 15.4 mg/l (as N) in February 2010 at site NW-88021129 and orthophosphate of 1.53 mg/l (as P) in January 2010 at site NW-88010209.

4.5.7 Industrial pollution

There is some industrial pollution in the Penrith Sandstone east of Penrith where chloride, sodium, calcium and nitrate are high, and all but calcium having exceeded the drinking water limit; but nitrate is falling here (Daily et al, 2006).

The organic chemicals trichloroethene, 1, 1, 1 trichloroethene, tetrachloroethene, chloroform and phenol have been detected at a number of sites in the St Bees Sandstone, however all were below the drinking water quality limits (Daily et al, 2006).

4.5.8 Climate change data outputs

Figure 41 to Figure 45 summarises the impacts of climate change on temperature, rainfall, PET, groundwater recharge and groundwater levels based on the RCP8.5 emissions scenario as derived from UKCP18 and its application to the eFLaG models in the Eden. There is high confidence that temperature will rise throughout 21st century (Figure 41), with warmer temperatures on average for all months of year. Warming is greatest in summer (JJA) and Autumn (SON), and the hottest days will get hotter.

Long term average rainfall (Figure 42) is projected to be relatively stable over time on average across UKCP18 RCMs, but the direction of change is uncertain. On average across the RCMs wetter winters and drier summers are predicted, with the change enhanced over time. There is a high confidence of drier summers for the end of century. Autumn rainfall extremes are projected to become more extreme on average, but direction of change is uncertain

There is high confidence that PET will rise throughout 21st century (Figure 43), with mean month monthly PET increasing between April and October. PET increases are highest in summer and PET extremes (95th percentile) will get more extreme.

LTA recharge is projected to decrease on average across RCMs, but the changes are small and the direction of change is uncertain (Figure 44). Winter recharge is projected to increase on average across RCMs decrease for other months. Recharge extremes (95th percentile) are projected to increase in November and December.

There is high confidence that LTA groundwater levels will fall (across RCMs) throughout the 21st century (Figure 45). There is high confidence that the lowest groundwater levels (Autumn 5th percentile levels) will decrease, which may result increasing risk of groundwater droughts.

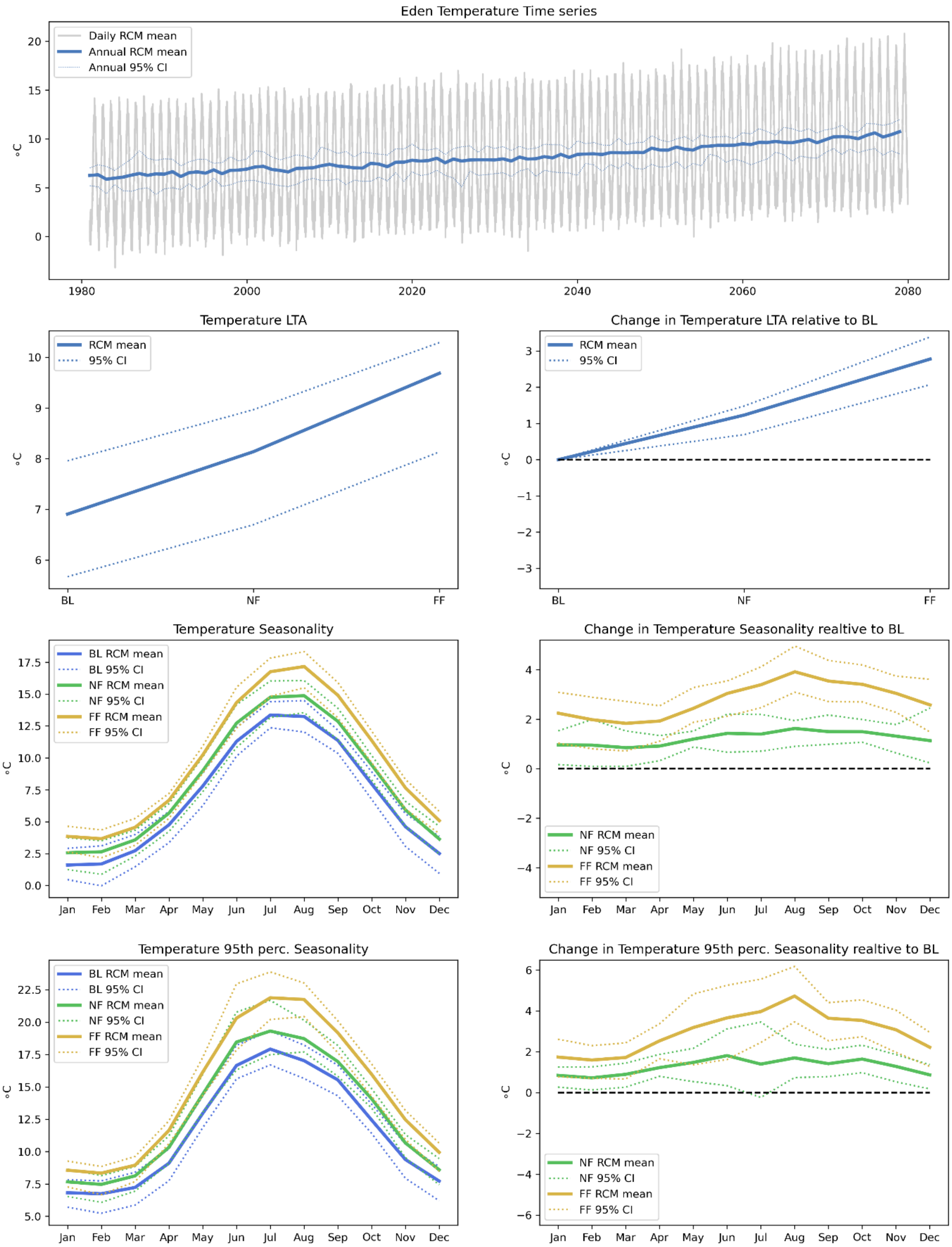

Figure 41 As Figure 15 but for temperature in the Eden

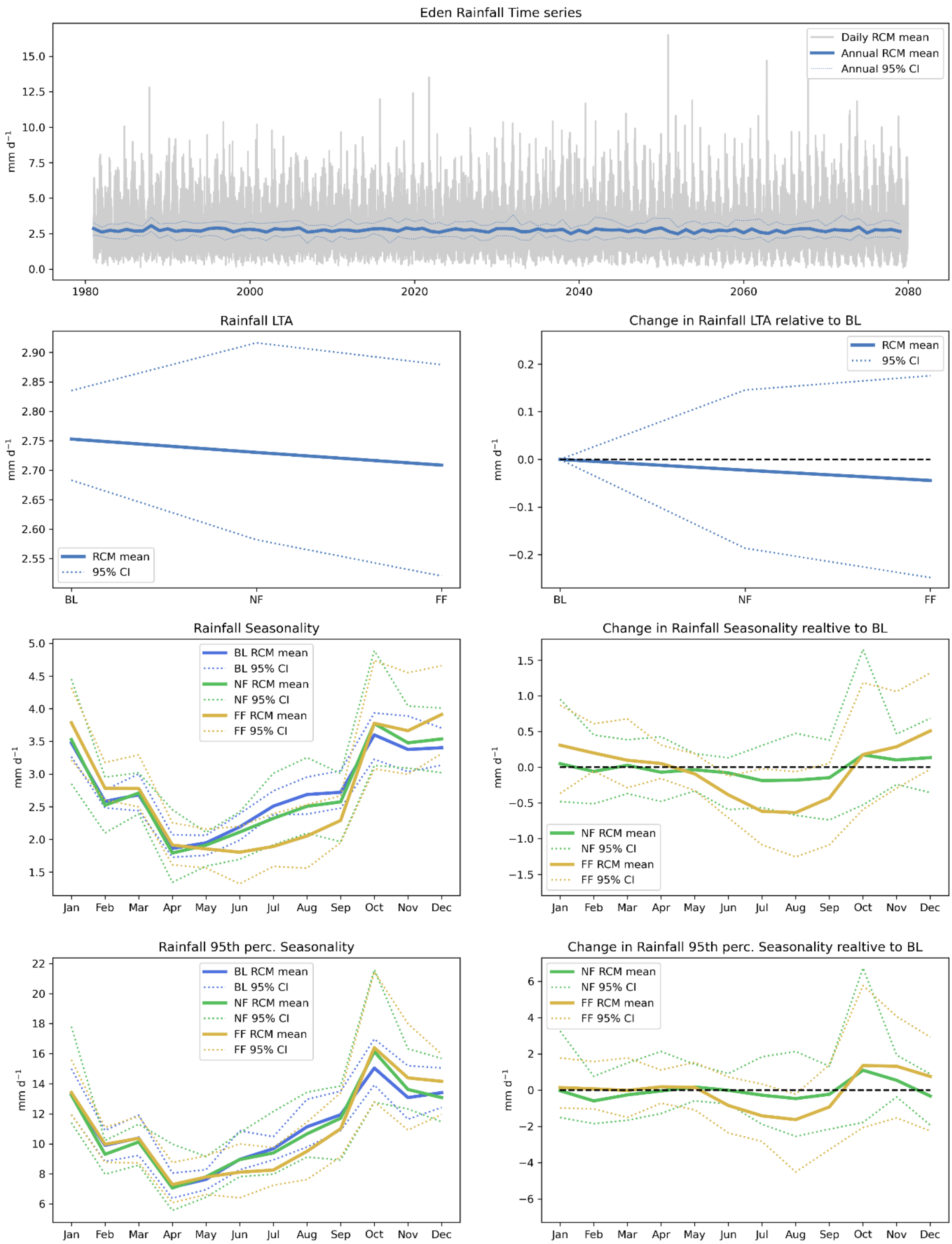

Figure 42 As Figure 15 but for rainfall in the Eden

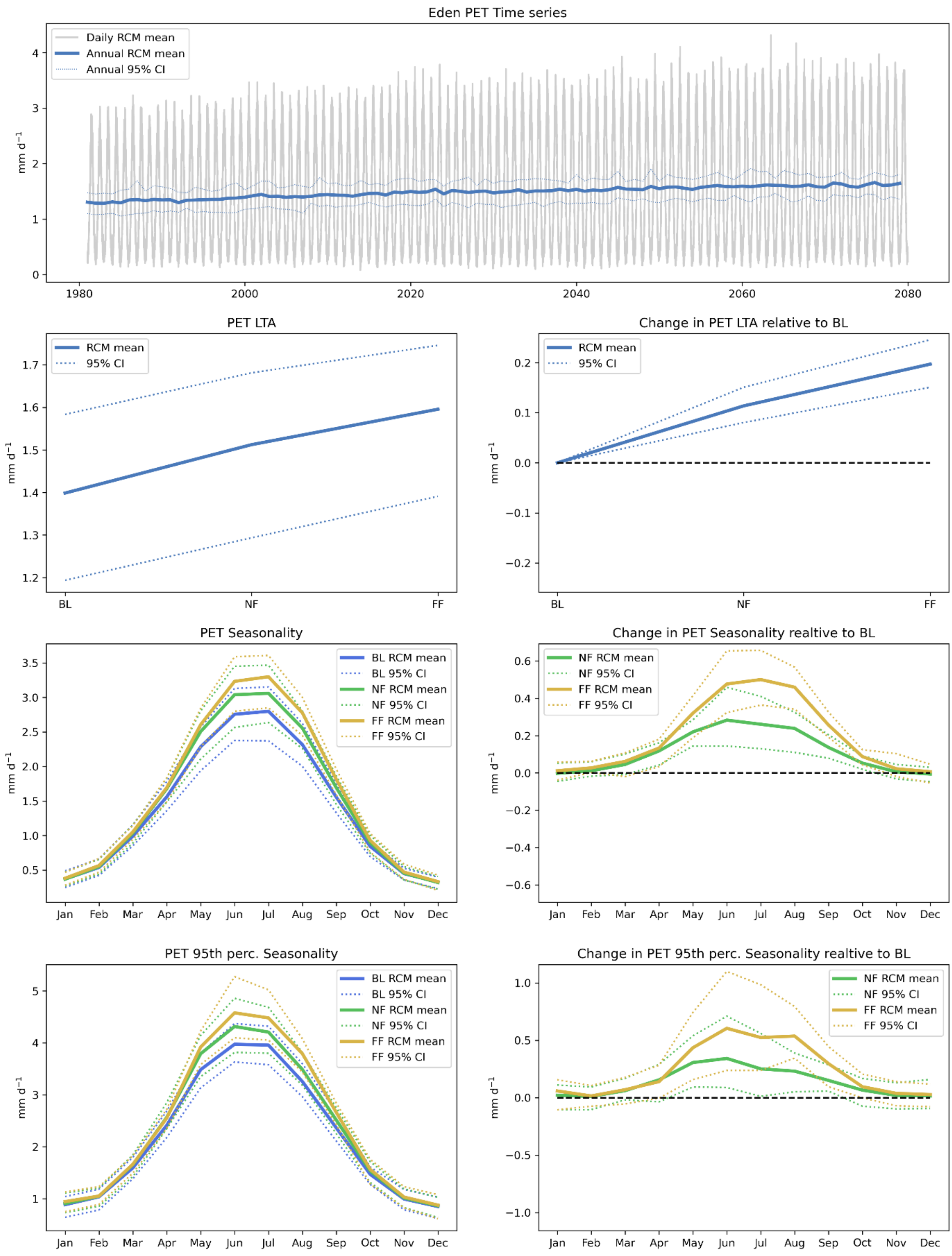

Figure 43 As Figure 15 but for PET in the Eden

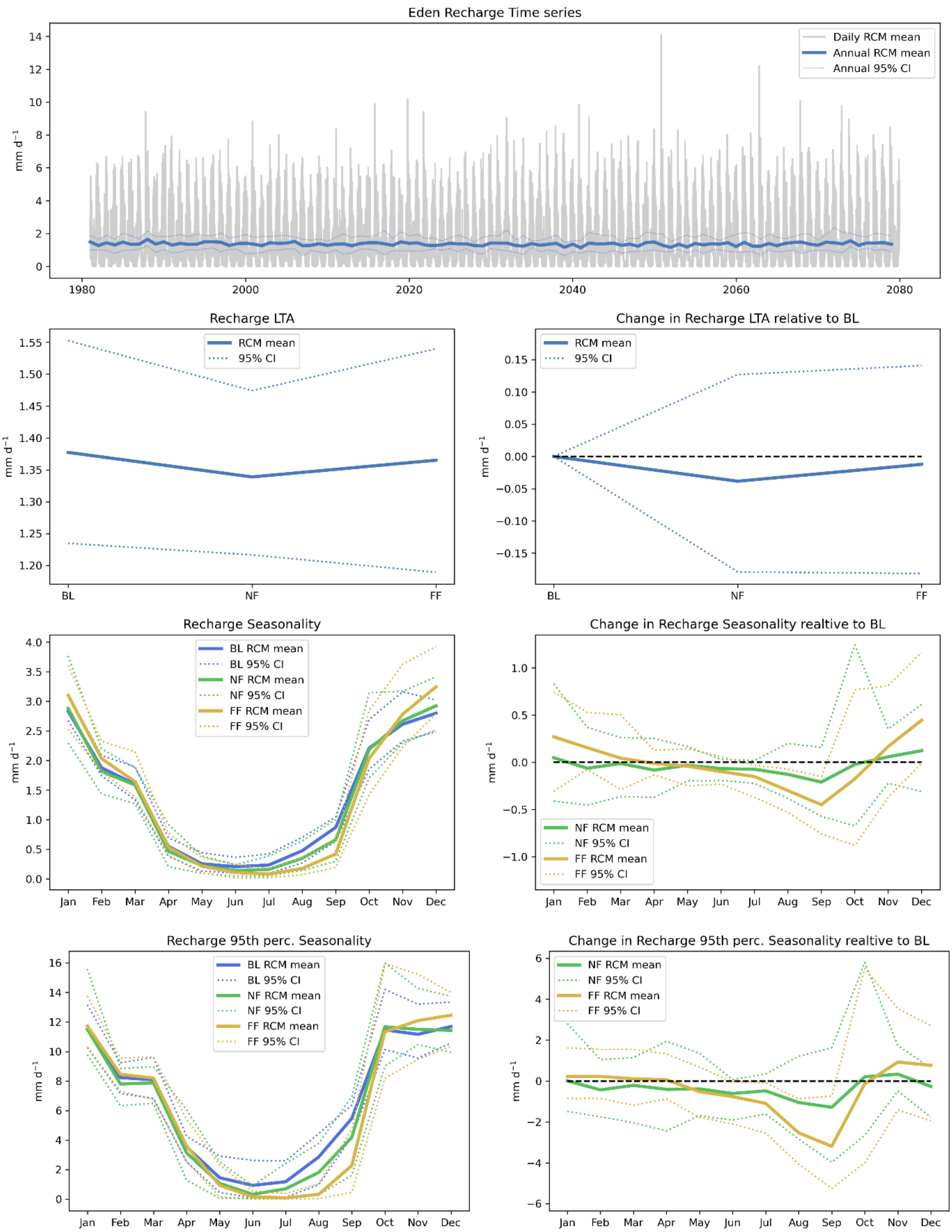

Figure 44 As Figure 15 but for groundwater recharge in the Eden

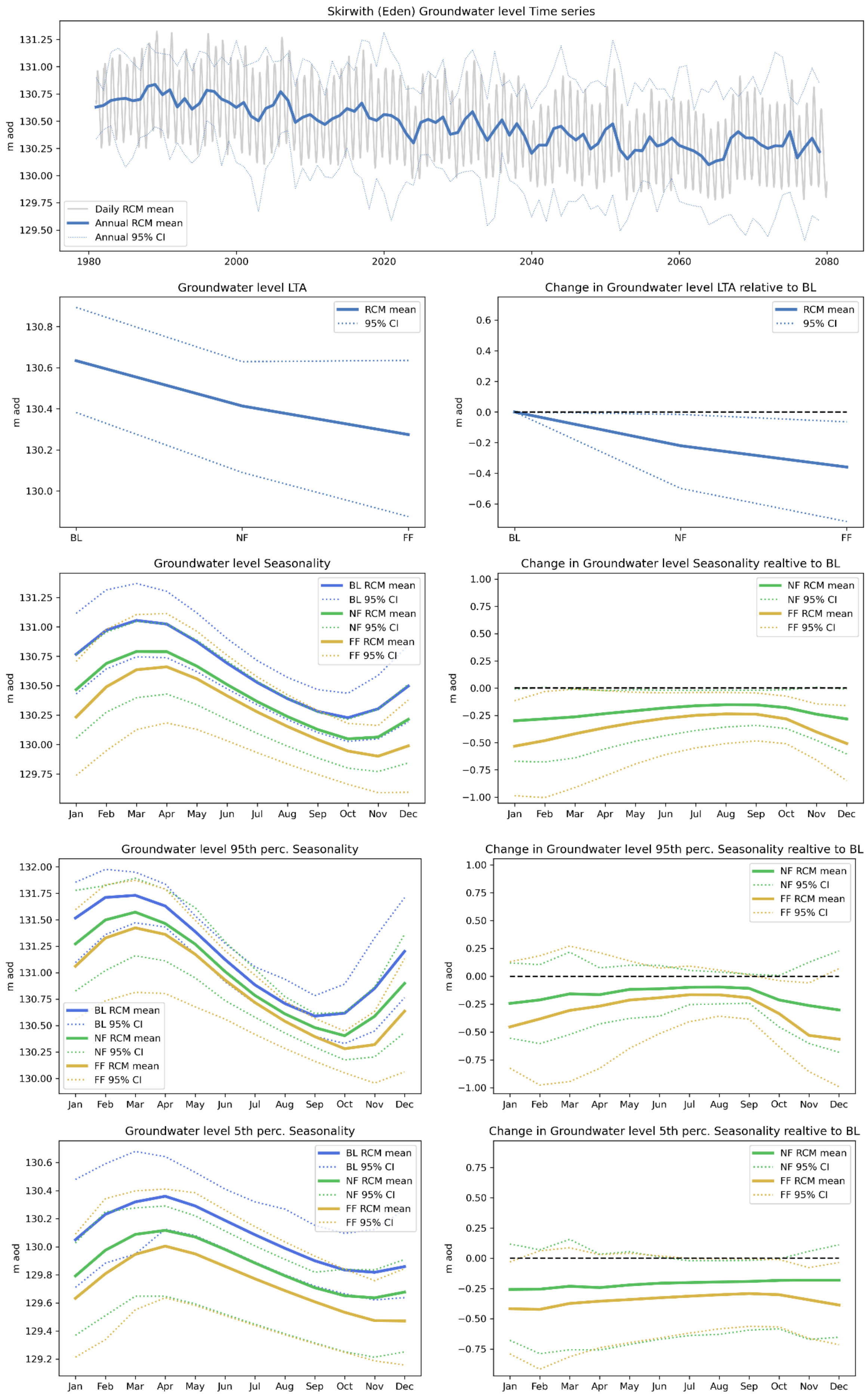

Figure 45 As Figure 15 but for groundwater levels in the Eden (Skirwith)

4.5.9 Potential impacts of climate change on groundwater quality

Based on the groundwater quality issues for this case study described in sections 4.5.5 to 4.5.7 and the UKCP18 projections described in section 4.5.8, the following direct impacts of climate change on groundwater quality are anticipated:

1. Increased temperatures of up to 3°C could increase reaction rates for degradation of nitrate and pesticides, but this may only be marginal.
2. Although long term average recharge is predicted to decrease, the direction of change in uncertain and wetter autumns and increased high intensity winter rainfall events increasing winter recharge, could lead to more recharge and produce more nitrate and pesticide spikes with increased mobilisation of these contaminants. However, this may be offset by greater dilution.
3. Drier summers could lead to less dilution and more concentrated recharge of nitrate and pesticides.
4. Long term average groundwater levels are predicted to fall, reducing groundwater flow into the predominantly effluent Eden which supports groundwater dependent ecosystems.

Potential changes of land use may have more influence on agricultural pollution than other changes in climate.

4.6 CASE STUDY 5 - CHICHESTER

4.6.1 Setting

This case study area concerns the Chichester Chalk Water Framework Directive Groundwater Body. The area is in two parts. The larger northern area covers from the coastal plain between Westbourne and Arundel, northwards inland to between the Hartings and Bury, west of the River Arun. In the west, the area also extends around the western side from the Weald, northwards to Selborne. The highest point is Butser Hill (4 km south-west of Petersfield) at 270 m. The northern extension and that north of the crest of the South Downs drain east to the River Rother, whilst the rest of the main Chalk outcrop is drained southwards by the Lavant and the Arun rivers. The Lavant has no perennnial head and dries out for long periods, but is very flashy, with flows of up to 685 Ml/d (Jones and Robins, 1999).

The southern area comprises the area from Emsworth and Thorney Island eastwards to Stocksbridge on the outskirts of Chichester. This area has a maximum elevation of 7 m and drains directly into Chichester Harbour via the Emsworth and Chichester Channels.

Land use in the northern area is mainly rural, with a mixture of arable farming, grassland and woodland, with pasture on the steeper scarp slope. Soils tend to be thin and stony. Whilst the southern area is predominantly arable and horticulture.

Rainfall varies from about 700 mm along the coast to over 1000 mm on the higher ground, with a mean of 904 mm/a and average recharge of 476 mm/a for the Chichester Chalk Block (this is not the same area as the Chichester Groundwater Body (Jones and Robins, 1999)). Modelling by Entec (2008b) indicated that the long term average recharge is in the range 418-432 mm/a, reducing to 186-203 mm/a during periods of extreme drought.

4.6.2 Geology

The area is in two parts separated by the outcrop of the low permeability Palaeogene deposits (comprising 30-45 m of Reading Formation and 80-115 m of London Clay Formation overlying the Chalk) of the Chichester syncline (Table 11, Figure 46). The larger northern area covers the Hampshire Chalk from the coastal plain northwards inland to the base of the Upper Greensand. In the west, the area also extends around the western side from the Weald, where it includes the Upper Greensand and mainly just the Grey Chalk outcrops. The southern area comprises the Bosham Chalk inlier.

Table 11 Stratigraphy of Chichester Chalk Groundwater Body

Group	Formation	Description
White Chalk Subgroup	Portsdown Chalk Formation	20 m relatively soft white chalk with common marl seams and some flints
	Culver Chalk Formation Spetisbury Chalk Member 40 m Tarrant Chalk Member 30-45 m	firm white flinty chalk
	Newhaven Chalk Formation	50-75 m soft to medium-hard, smooth, white chalk with numerous marl seams and flint bands
	Seaford Chalk Formation	55-80 m soft white chalk with seams of nodular and semi-tabular flint
	Lewes Nodular Chalk Formation	50-55 m interbedded, hard, nodular chalks and soft to medium-hard chalks and marls with regular seams of large nodular flints
	New Pit Chalk Formation	25-40 m white, massively bedded with regularly spaced marls and some flints in upper part
	Holywell Nodular Chalk Formation	15-35 m hard, white, nodular, very shelly chalk. Melbourn Rock Member (3-5 m hard, massive to nodular, shell-free chalk with marl partings) and Plenus Marls Member (1-3 m grey marls) at base
Grey Chalk Subgroup	Zig Zag Chalk Formation	40-60 m greyish blocky chalk with thin limestone/marl couplets at base
	West Melbury Marly Chalk Formation	5-35 m cycles of pale-grey, marly chalk with thin grey to brown limestones. Glauconitic Marl Member (1-3 m olive green, glauconitic sandstone) at base
Selborne Group (at depth)	Upper Greensand Formation	25-40 m calcareous, bedded, bioturbated and variously argillaceous siltstone with intermittent harder lenticular beds
	Gault Formation	About 92 m grey and bluish grey, silty mudstones with sporadic bands of phosphatic nodules

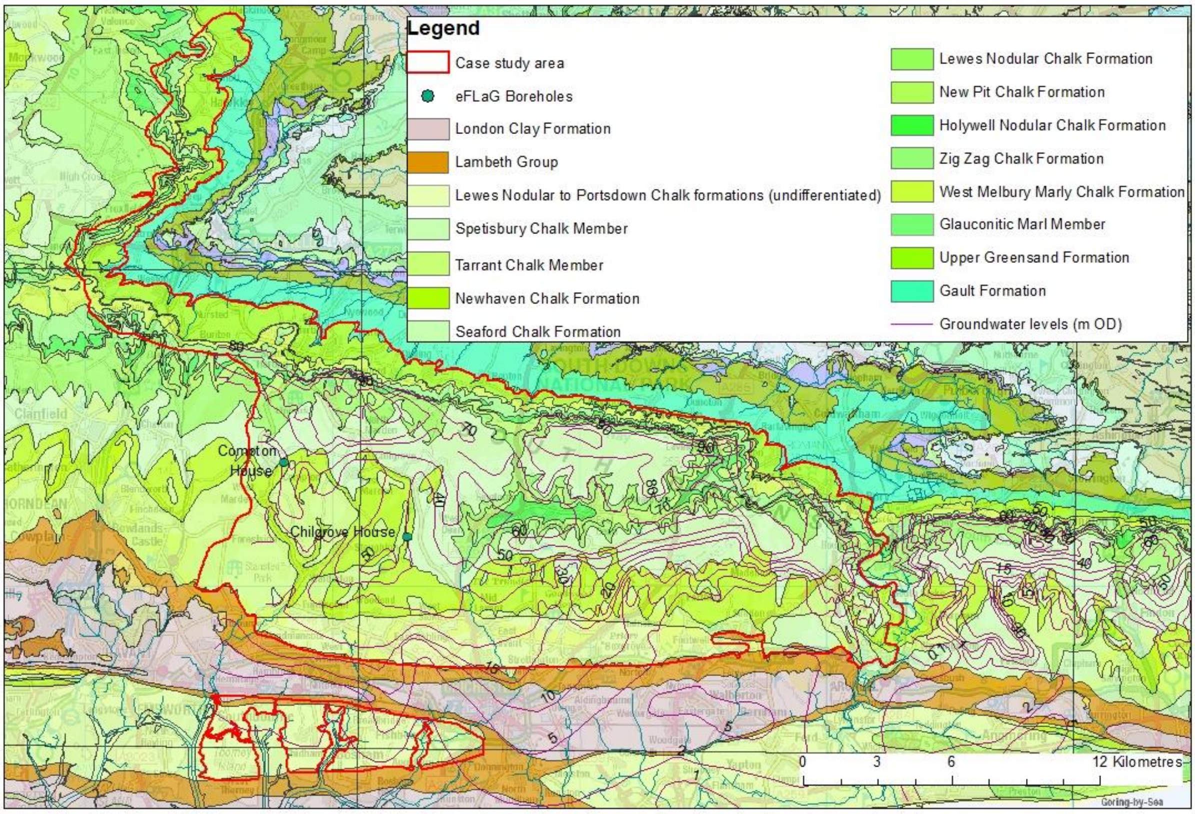

Figure 46 Bedrock and water levels (October 1993) in Chichester Chalk Water Framework Directive Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

In the main northern area, superficial deposits are generally absent (Figure 47), however alluvium (silty, sandy and pebbly clay with minor interbeds of gravel and peat) or head (gravelly, silty, sandy clay) are present along the river valleys and clay-with-flints (silty clay with angular and nodular flints) on the interfluves. Along the southern boundary, there is a 1-2.5 km wide zone of head gravel (angular flint gravel in red-brown silty clay or whitish chalky matrix) with areas of river terrace deposits (formerly called ‘brickearth’ and comprising silty clay).

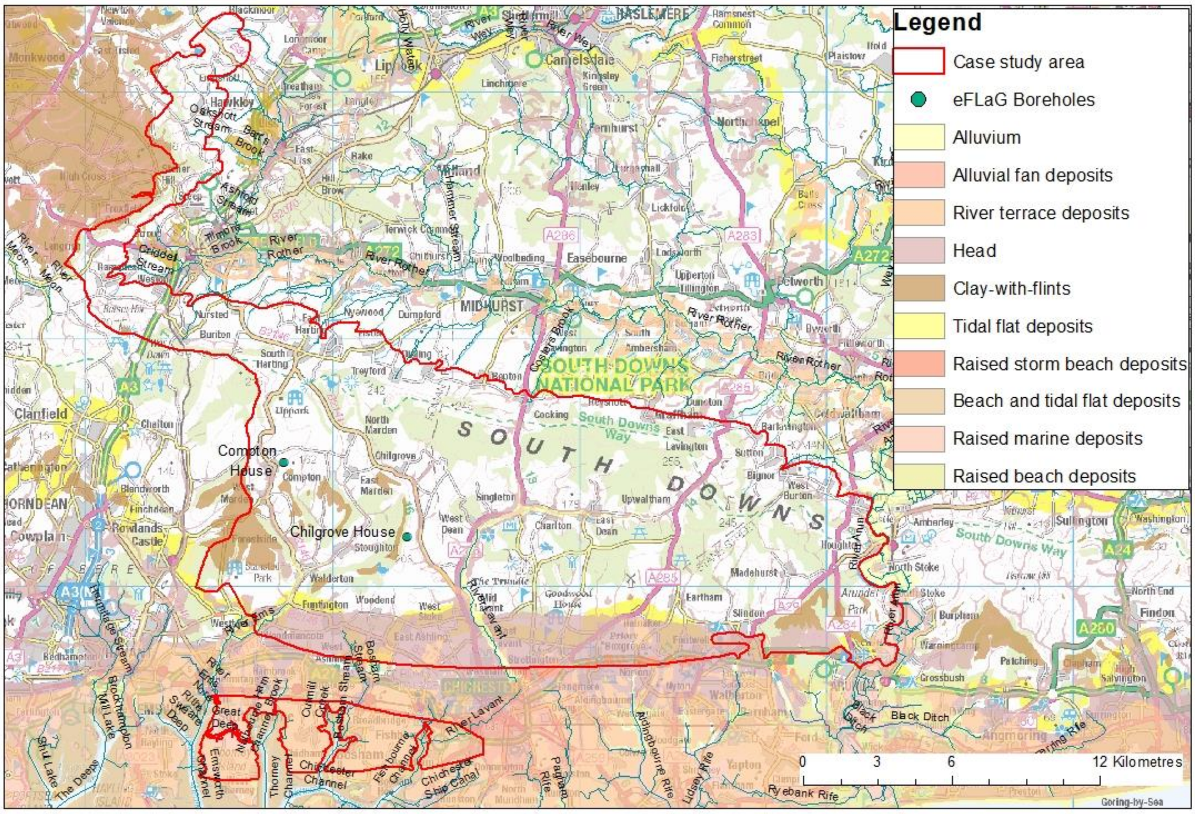

Figure 47 Superficial deposits in Chichester Chalk Water Framework Directive Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0.

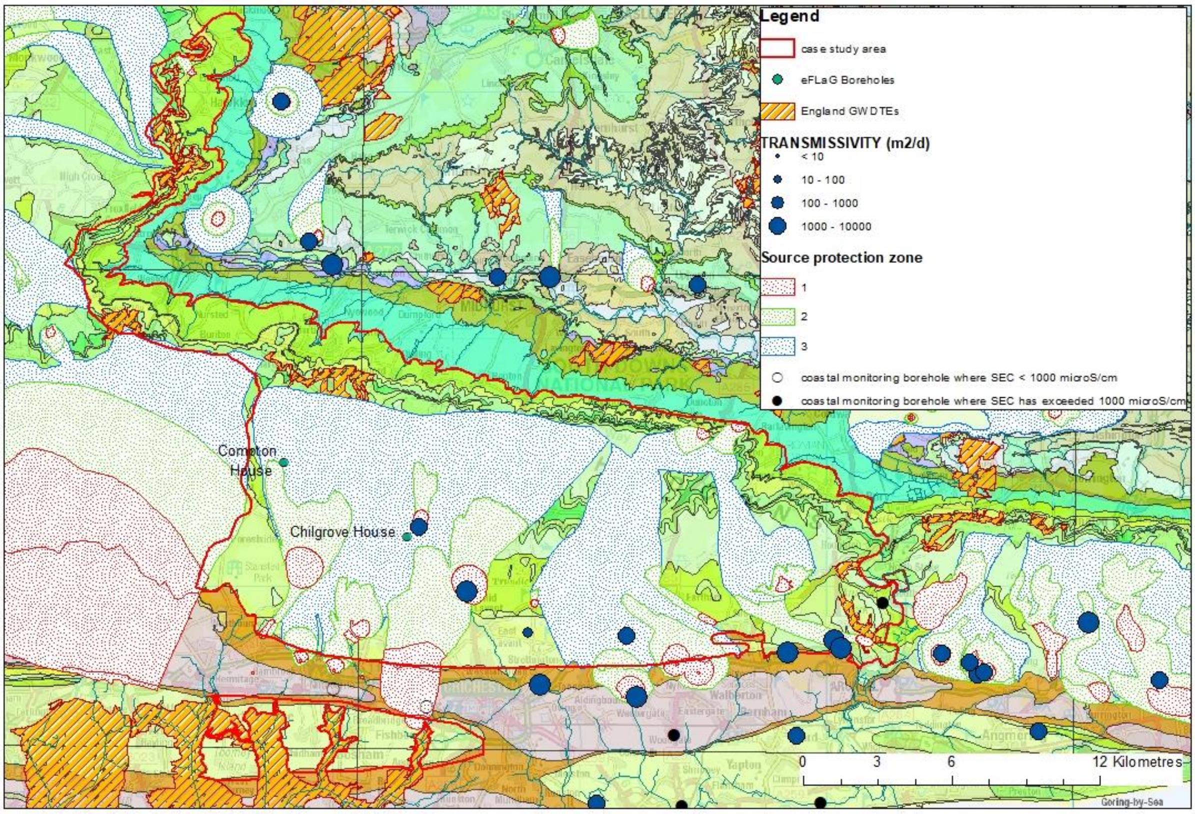

Figure 48 Various hydrogeological parameters in the Chichester Chalk Water Framework Directive Groundwater Body. Contains OS data © Crown copyright 2022, and public sector information licensed under the Open Government Licence v3.0. Source protection zones © Environment Agency copyright and/or database right 2016.

The Chalk outcrop of the southern area is overlain by river terrace deposits (formerly classified as 'brickearth' comprising clayey silts sometimes with fine sand) and raised marine deposits (laminated silty clay and fine sand).

4.6.3 Hydrogeology

The Upper Greensand in the north-west discharges via springs and seepages at its base into the River Rother headwaters. It is thought to have some hydraulic continuity with, and be recharged from, the Chalk, although the contact between the Grey Chalk and the top of the underlying Upper Greensand is also marked by a spring line.

The Chalk is a microporous limestone comprising coccolith debris and has dual porosity with a high intergranular porosity (with mean values of 38.8±5.8% for the upper part of the White Chalk, 28.4±4.2% for the lower part of White Chalk and 22.9±7.7% for the Grey Chalk (Allen et al, 1997)) but is not readily drained due to the small size of the pore throats. Hence it has a low matrix permeability and a high fracture permeability with water moving predominantly through a network of fractures that can be solution enhanced and are generally better developed in the zone of water table fluctuation (Allen et al, 1997). Groundwater flow is affected by the presence of flint bands, marl seams and hard bands which may act as inception horizons for dissolution. The Chalk transmits water less readily at interfluve locations than it does in valley localities due to the presence of fewer fractures. The Chalk also generally transmits water less readily with depth as fractures become smaller and less common although occasional hard limestone bands such as the Melbourn Rock can have well-developed fracture systems. Folding and fracturing may cause concentration of groundwater flow in areas such as the Chichester syncline around a large fault striking north-west from Arundel. Solution features such as sinkholes and dolines develop particularly at contacts between Chalk and less permeable formations, such as the

Palaeogene and clay-with flints, as the acidic runoff dissolves the Chalk, and occur along the Rivers Lavant and Ems where they cross from Chalk to Palaeogene. The Environment Agency has a numerical groundwater model of the East Hants Chichester Chalk that covers this area.

The Lavant flooded Chichester in winter 1993/94 as once groundwater levels had risen to a certain level that corresponded to a highly permeable zone of Chalk, this acted as an overflow system discharging water to springs within the valley (Jones and Robins, 1999).

Therefore transmissivity is related to topography (with the highest values in valleys and lowest under interfluves), lithology, structural features and proximity to Paleogene cover (with a concentration of chemically aggressive runoff). Transmissivity values from Allen at al (1997) vary from 57 to 9600 m^2/d; whilst Entec (2008b) quote values of 16-9500 m^2/d, with up to 25000 m^2/d at the Fishbourne springs (that have a long term average discharge of 27 Ml/d) probably representing karst development beneath the Chichester syncline (Figure 48). Modelling work indicates a specific yield of 1-2.4% (Entec, 2008b).

The thickness of the unsaturated zone reaches 150 m below the highest elevation areas of the South Downs, and annual fluctuations in water level can exceed 25 m. Groundwater contours (Figure 46) indicate a valley to the west of the Lavant and a ridge of higher water levels separating the Chalk Portsdown and Littlehampton anticlines.

Groundwater flow is generally north to the Rother or south towards the coast. The Singleton anticline acts as a barrier to southerly flow, effectively separating the catchments of the Arun and Lavant; as a result groundwater flows are directed eastwards towards Arundel and westwards towards Chichester. The Chichester syncline acts as a barrier preventing or restricting groundwater flow from north to south towards the sea. Flow is diverted and restricted to a few zones, giving rise to several discrete discharge points, such as those near the River Arun at Arundel (with an average discharge of 4.3 Ml/d), the Fishbourne springs (have flows between 13 and 35 Ml/d) and in the Havant area just west of the study area. The syncline is pierced by several tidal creeks near Chichester, located along the axis of the syncline and many springs. Flow under the Chichester syncline supports submarine springs in Chichester and Langstone harbours with groundwater flow much slower to the east of Chichester (Entec, 2008b).

The flow of the Forebridge, Elbridge and Lidsey rifes that rise on Palaeogene deposits are fed via the river terrace deposits ('brickearth') and head deposits from the Chalk to the north (Entec, 2008b). Chemical analyses indicate that the Bremere, Pagham and Oving rifes contain a significant chalk groundwater component (Jones and Robins, 1999).

There are a large number of licensed abstractions from the Chalk for public supply in the northern area: springs, as well as ten pumping stations. One source (in a tributary valley of the River Lavant) is operated only in summer to prevent abstraction from another source (in the main Lavant valley) having a detrimental effect on flows in the river during dry periods. There is also a private water supply and a supply for cressbeds/fish farm. There are abstractions from the confined Chalk for watercress and two public supplies with source protection zones that extend into the area, but the sources themselves are located within the adjacent Sussex Lambeth Group groundwater body. The total catchment area for twosources in the adjacent East Hampshire groundwater body also extends into this area.

From the southern area, there is a single licensed abstraction for public supply.

4.6.4 Groundwater dependent terrestrial ecosystems

Ashford Hangers Nature Reserve is a chalk woodland, biological SSSI. Butser Hill is a chalk massif with a discontinuous cap of clay-with-flints. The massif has been eroded leaving a series of deep coombes, with the modern spring line about 1 km from the coombe head. The Harting Downs SSSI comprises the scarp slope of the South Downs, and a series of parallel valleys across the gentler dip slope, that consist of chalk with clay-with-flints capping in places. It includes archaeological earthworks, cross-dykes and a fort, and a wide range of habitats. Arundel Park SSSI is an old deer park containing the artificial but old Swanbourne Lake that derives most of its flow from the Blue Spring; there have been impact studies that investigate the effects of the pumping regimes at two public supply sources. Chalk groundwater contributes spring flow (e.g. at the head of Fishbourne Channel) to Chichester Harbour SSSI during winter and spring when groundwater levels are highest.

4.6.5 Groundwater quality

The high intergranular porosity provides a very high surface area enhancing the potential for reaction of the chalk surfaces. In addition, the matrix, having a permeability several orders of magnitude lower than the fractures, acts as a reservoir for older water and may affect groundwater quality by slow exchange with the fracture water. The White Chalk Subgroup comprises very pure carbonates. The most important non-carbonate minerals are quartz, montmorillonite, white mica and apatite, with kaolinite only found in the Grey Chalk Subgroup. These have a disproportionate effect on water chemistry releasing magnesium, manganese, strontium and iron during congruent (rapid) and incongruent (slower) reactions. Rainfall acidity is quickly neutralised by reaction with chalk sediment and the reactivity of the soil water is greatly enhanced by the solubility of the carbon dioxide produced biogenically in the soil zone. Congruent dissolution of the chalk occurs during infiltration through the unsaturated zone and saturation with calcite is typically attained within a few metres of the surface (Edmunds et al, 1992); below this depth chalk dissolution is greatly diminished. However, a small percentage of by-pass flow through open fractures may result in calcite-undersaturated water being transported to deeper parts of the unsaturated zone with capacity for fissure enlargement, as demonstrated by tritium studies (Foster and Smith-Carrington, 1980).

The Chalk groundwater is generally of good quality with limited saline intrusion from the coast but the constant threat of pollution mainly from agrochemicals. The water is of calcium-bicarbonate type, with a specific electrical conductance typically of 400-750 µS/cm. Calcium concentrations are generally in the range 90-120 mg/l, magnesium less than 5 mg/l and sodium 10-20 mg/l. Bicarbonate is generally in the range 250-320 mg/l, sulphate 5-30 mg/l and chloride 18-35 mg/l (Jones and Robins, 1999; Entec, 2008b). Fluoride is reported as generally less than 0.1 mg/l (Jones and Robins, 1999; Edmunds and Brewerton, 1997), although Entec (2008b) quote concentrations up to 0.2 mg/l.

Magnesium, calcium, bicarbonate, nitrate and potassium increase from north to south, sulphate probably decreases in this direction. Chloride also increases eastwards towards the Arun. Sulphate appears to be lower following dry periods of low recharge and higher following wetter periods, similar to chloride and nitrate (Entec, 2008b). Chloride and conductivity are higher in late winter and early spring. In the Bosham inlier the mean chloride concentration is 25.6 mg/l (much less than from the Littlehampton inlier to the east) due to active discharge from the Chalk via phreatic and submarine springs reducing the potential for saline intrusion (Entec, 2008b).

Edmunds and Brewerton (1997) described these Chalk groundwaters as aerobic (maintaining low dissolved iron that precipitates as oxides on fracture surfaces, and persistence of nitrate) and of low mineralisation (less evolved than other Chalk groundwaters in southern England) reflecting atmospheric inputs as well as reactions at shallow depth between water and Chalk sediment. The waters can be described in terms of initial rapid water-rock interaction; slight modification of composition by incongruent reaction with increasing residence time and mixing with small amounts of saline and chemically-evolved water from matrix storage. Low and uniform strontium indicates short residence times (decades).

Within the confined Chalk aquifer of the Chichester syncline, there is a region of reducing groundwaters, with nitrate <1 mg/l (as N) due to dentification and dissolved iron up to 4.4 mg/l, as seen in the confined aquifer at Groves Farm (NGR 490300, 102900) on the northern edge of the Littlehampton Chalk block (Jones and Robins, 1999). Confined groundwaters of the Chichester block have high strontium relative to chloride indicating in excess of 1000 years residence times, similarly higher fluoride (>0.25 mg/l) related to water-rock interaction. A single sample from North Mundham (below Palaeogene deposits between the Bosham and Littlehampton inliers) has a lighter $\delta^{18}O$ isotopic composition of -7‰ indicating a probable palaeowater (Jones and Robins, 1999).

Entec (2008b) indicated that the main quality issue is diffuse pollution from predominantly agricultural land use; pesticides are transient and localised. Former gravel workings near

Chichester lying directly over the Chalk are used as landfills (1960s and 1970s) and may not be lined (Entec, 2008b).

4.6.6 Agricultural pollution

Nitrate concentrations are generally elevated above baseline and in the range 3-10 mg/l (as NO_3-N) and locally exceed the prescribed concentration or value (PCV) of 11.3 mg/l (as N) for public and private supplies. Concentrations increase along groundwater flow paths due to a reduction in the thickness of the unsaturated zone, differences in pumping rate and water level during pumping and differences in land use, with the highest values in sources near the edge of the outcrop. Jones and Robins (1999) reported that nitrate was increasing in about 40% of sources across the South Downs, predominantly in areas of more intensive arable agriculture and classified sources as having concentrations: 4-7 mg/l (as N); 4-7 mg/l and increasing; 4-7 mg/l with sharp recharge spikes; 7-11 mg/l and rising and 7-11 mg/l, increasing with sharp recharge spikes > 11.4 mg/l. Time series plots for nitrate show some seasonal variation (particularly following the drought in 1993), with peaks in winter and spring, coinciding with periods of highest recharge, soil leaching and water levels transporting nitrate from fertiliser or manure into the saturated zone of the aquifer (Jones and Robins, 1999). Seasonal variations can exceed 5 mg/l (as N).

Entec (2008b) reported that total pesticides exceeded the PCV of 0.5 µg/l at least once at 14 out of 24 sampling locations in the area. Bentazone (up to 0.25 µg/l) and atrazine (up to 0.27 µg/l) each exceeded the PCV of 0.1 µg/l at one site, but only the mean concentration of Bentazone of 0.16 µg/l (2 samples) was greater than the PCV. Edmunds and Brewerton (1997) detected atrazine at up to 0.047 µg/l.

4.6.7 Chlorinated solvents

Entec (2008b) stated that meta- and para-xylenes had been detected at concentrations exceeding the DWI PCV of 0.1 µg/l, at four sampling points and fluoranthene in excess of the DWI PCV of 0.001 µg/l at three different places. Other aromatic compounds such as toluene, phenols and ethylbenzene were detected but nowhere exceeded the PCV. Chlorinated solvent concentrations were typically low (median values < 0.25 µg/l) and nowhere exceeded their PCVs. There is no recent data for fluoranthene in the EA database, but there are meta- and para-xylenes concentrations of 0.45 µg/l (October 2005) at site SO-F0002591, 0.62 µg/l (April 2014) at site SO-G0017492 and 0.8 µg/l (April 2010) at site SO-F0002555.

4.6.8 Saline intrusion

The Chichester Chalk block is partly protected from saline intrusion by the Chichester syncline, which provides a low permeability barrier to groundwater flow. Flow gauging and chemical analysis of streams draining the central part of the block indicate that outflows are blocked by low permeability chalk. Consequently, none of the public supply sources is affected by salinity issues. However, monitoring boreholes south of the syncline to the east (in the adjacent Littlehampton inlier) show a high degree of saline intrusion with conductivities at Shripney (near coast) up to 20 000 µS/cm at 60 m increasing to 30 000 µS/cm at 145 m, and conductivities above 33 000 µS/cm below 110 m at Climping (< 1 km from coast). The high salinity appears to extend inland (at a reduced level) as far as the southern limb of the syncline with the Woodgate monitoring borehole (7 km from coast) having a conductivity of 2 000 µS/cm. The nearest PWS are 5 km to the north and are not affected by saline intrusion. The River Arun is a potential source of salinity north of the syncline (as it is tidal further inland than the base of the Chalk outcrop), however due to the concentration of groundwater flow caused by the syncline deflecting groundwater eastwards; and a source 500 m from river has average chloride concentrations of 25 mg/l. This source is however affected by filamentous algae requiring micro-filtration (Jones and Robins, 1999). Boreholes in the Worthing block (the other side of the River Arun) are affected by saline intrusion and groundwater at the South Stoke monitoring hole, 1.5 km north of Arundel (also 500 m west of the river) has a maximum conductivity greater than 10 000 µS/cm, and this occurs below a fractured zone at 70 m depth, with conductivity increasing from 70 m to the base of the hole at 113 m.

4.6.9 Climate change data outputs

Figure 49 to Figure 53 summarises the impacts of climate change on temperature, rainfall, PET, groundwater recharge and groundwater levels based on the RCP8.5 emissions scenario as derived from UKCP18 and it's application to the eFLaG models in Chichester. There is high confidence that temperature will rise throughout 21st century (Figure 49), with warmer temperatures on average for all months of year. Warming is greatest in summer (JJA) and Autumn (SON), and the hottest days will get hotter.

Long term average rainfall (Figure 50) is projected to decrease over time on average across UKCP18 RCMs, but the direction of change is uncertain. On average across the RCMs wetter winters and drier summers are predicted, with an increase in the magnitude of change over time. There is a high confidence of drier summers and wetter winters for the end of century. There is high confidence that rainfall extremes will become more extreme in winter by end of century and less extreme in summer. Note, however, that the latter results contradict the UKCP18 simulations driven by their convection permitting model (CPM) which indicate that summer rainfall extremes will become more extreme under the RCP8.5 emissions scenario. Simulations of summer rainfall extremes by the CPM are likely to be more reliable.

There is high confidence that PET will rise throughout 21st century (Figure 51), with mean month monthly PET increasing between April and October. PET increases are highest in summer and PET extremes (95th percentile) will get more extreme.

LTA recharge is projected to increase on average across RCMs, but the direction of change is uncertain (Figure 52). Winter recharge is projected to increase on average across RCMs decrease in Autumn. Recharge extremes (95th percentile) are projected to increase in Winter.

LTA groundwater levels are projected to be stable over 21st century although the direction of change is uncertain (Figure 53). Spring groundwater level maxima are projected to increase and Autumn/Winter groundwater level minima and during the start of the recharge period are projected to decrease on average across RCMs. There is a relatively high confidence that spring groundwater level extremes will increase indicating a potential increasing risk of groundwater flooding.

Sea level rise is predicted to occur along coast and along the tidal extent of major rivers and estuaries (Figure 54) which is likely to affect the extent of saline intrusion.

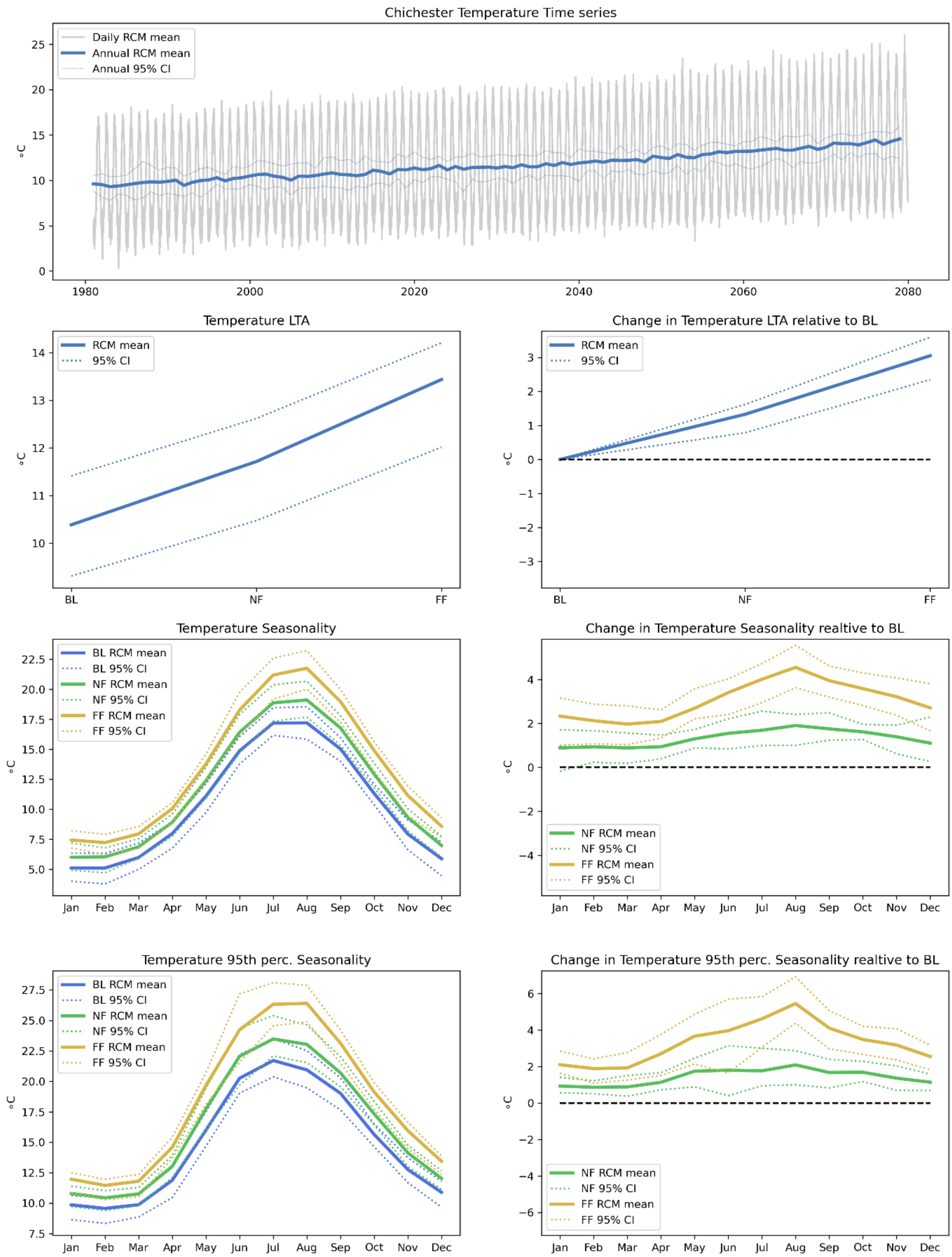

Figure 49 As Figure 15 but for temperature at Chichester

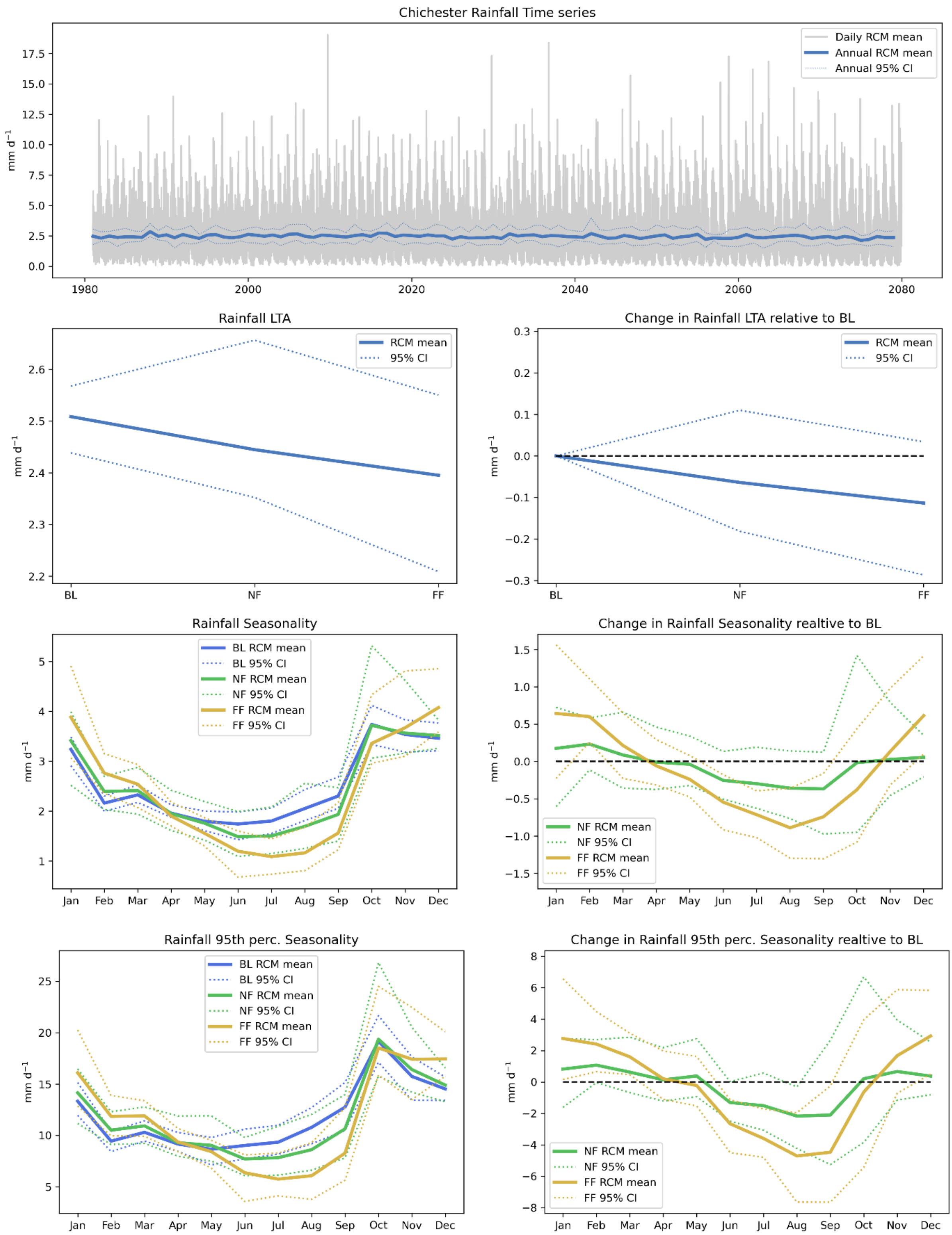

Figure 50 As Figure 15 but for rainfall at Chichester

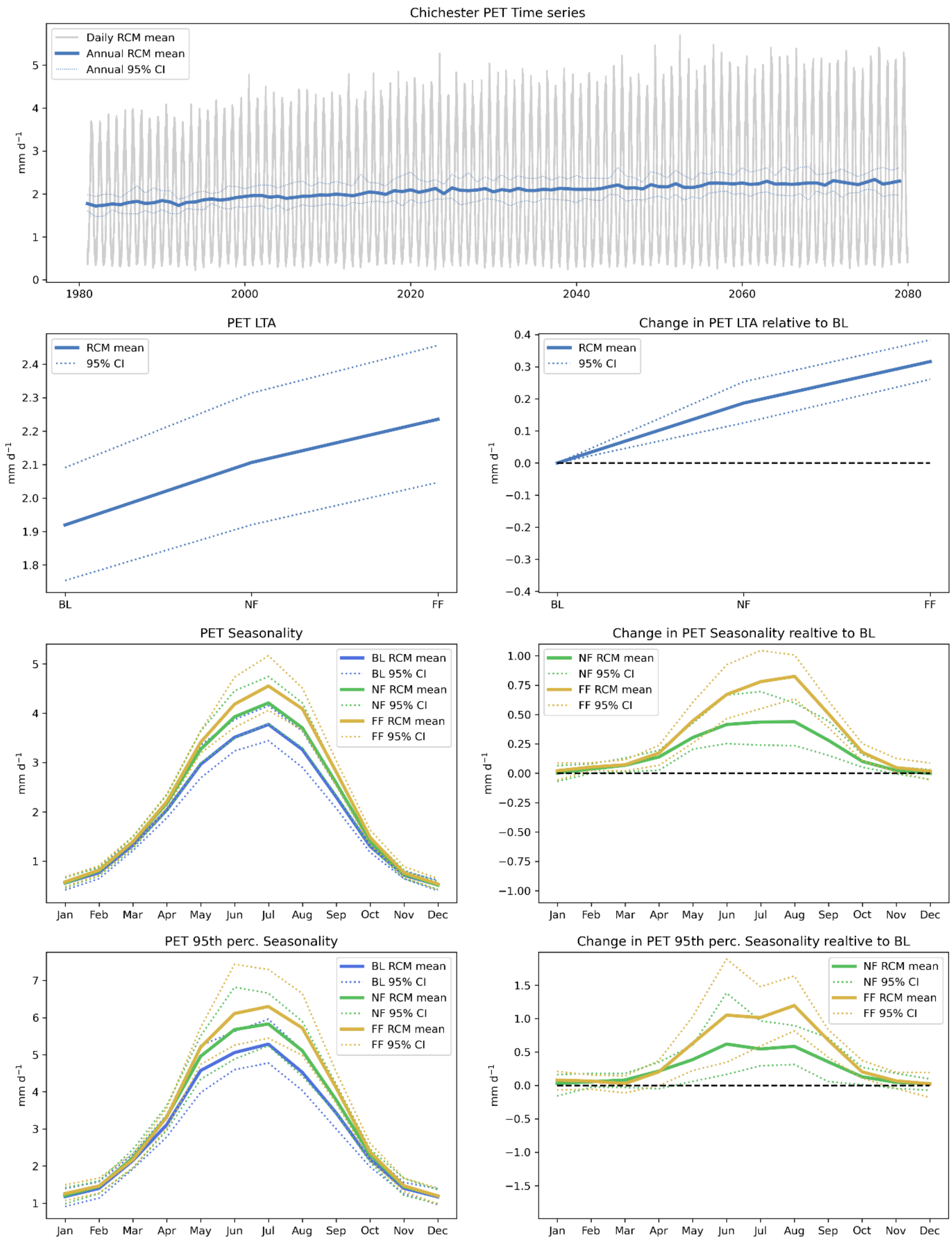

Figure 51 As Figure 15 but for PET at Chichester

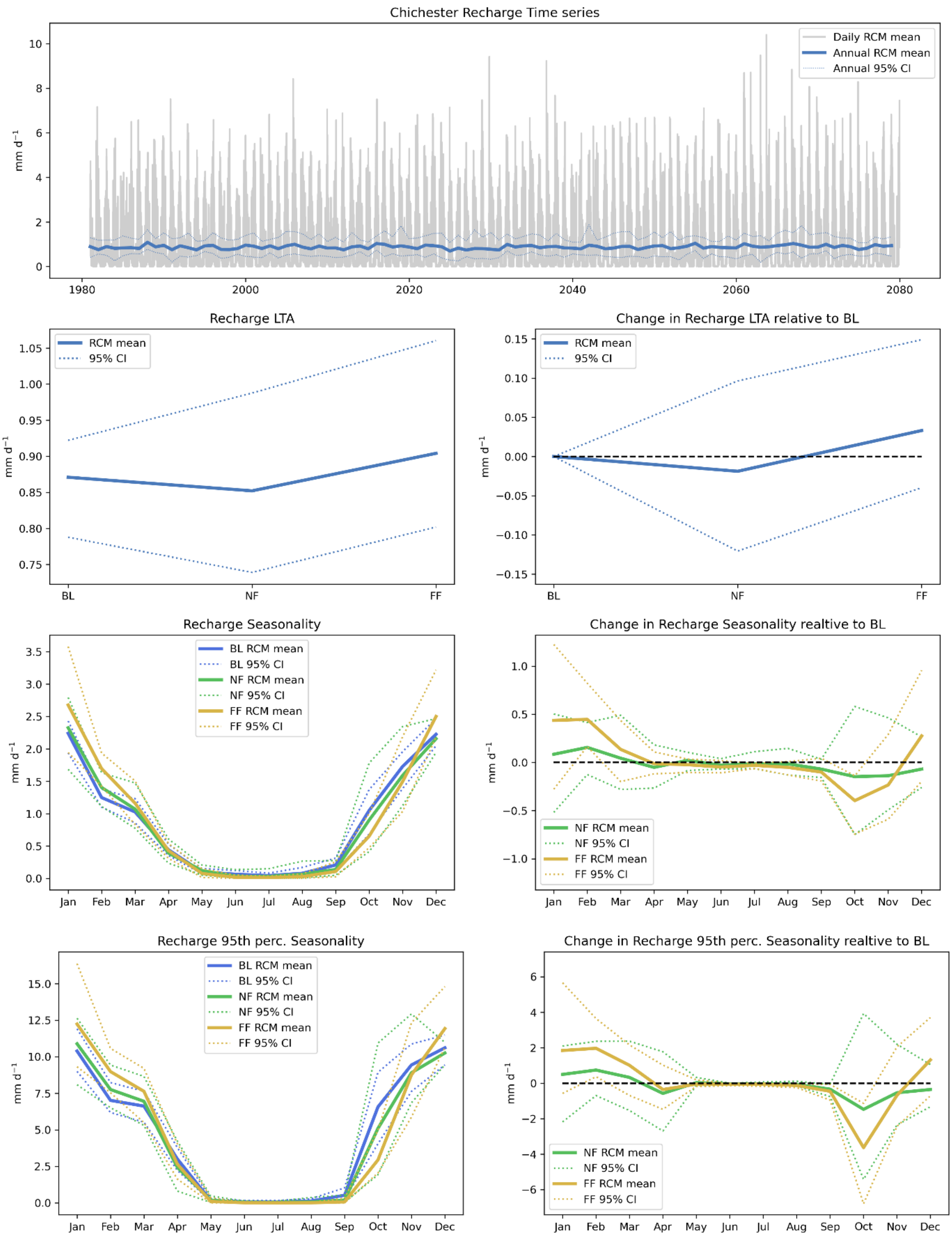

Figure 52 As Figure 15 but for groundwater recharge at Chichester

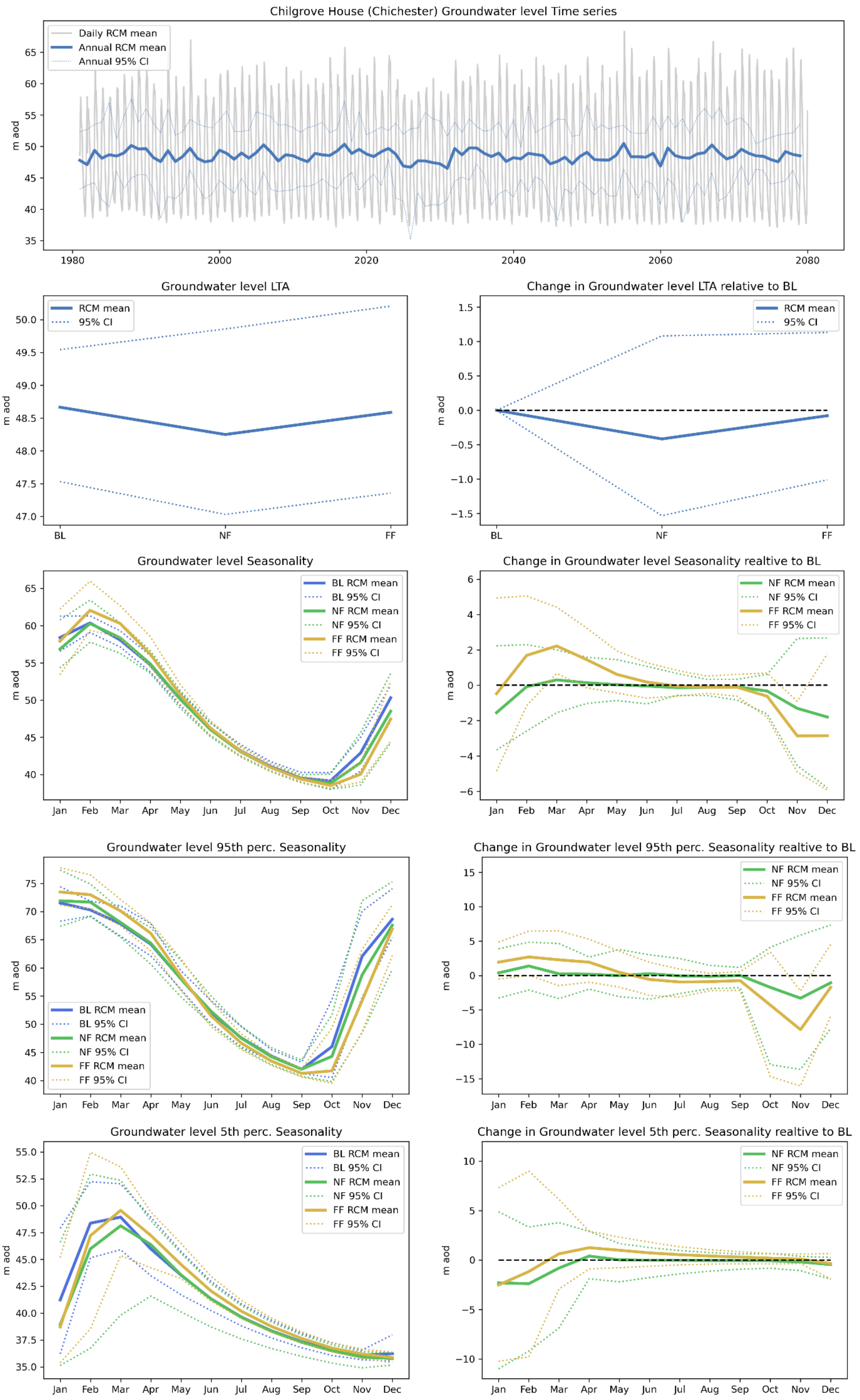

Figure 53 As Figure 15 but for groundwater levels at Chichester (Chilgrove House)

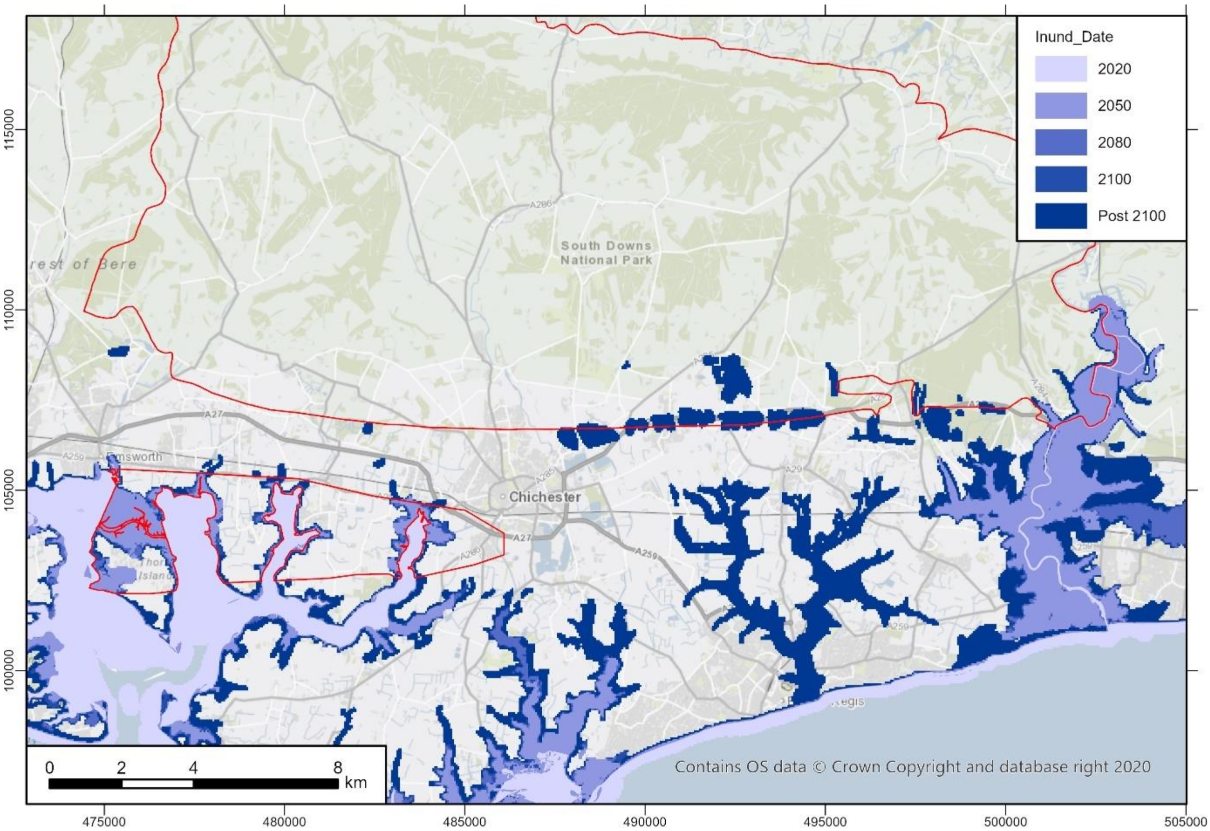

Figure 54 Chichester Study Area showing GeoCoast: Inundation UKCP18

4.6.10 Potential impacts of climate change on groundwater quality

Based on the groundwater quality issues for this case study described in sections 4.6.5 to 4.6.8 and the UKCP18 projections described in section 4.6.9, the following direct impacts of climate change on groundwater quality are anticipated:

1. Increased temperatures of up to 3°C could increase reaction rates for degradation of nitrate, pesticides and chlorinated solvents, but this may only be marginal.
2. Increased extreme winter rainfall events, could lead to more recharge and produce more nitrate, and pesticides spikes with increased mobilisation of these contaminants. However, this may be offset by greater dilution.
3. Drier summers could lead to less dilution and more concentrated recharge of nitrate and pesticides.
4. The effect of wetter winters and drier summers increasing the size of the seasonal fluctuations in water levels, would also decrease the thickness of the unsaturated zone in spring, potentially decreasing the timelag for nitrate to reach the water table.
5. Greater maximum groundwater levels could potentially increase groundwater flooding and mobilise more agricultural pollutants.
6. The rise in sea level due to increasing temperatures will lower head gradients at the coast and hence increase the potential for saline intrusion from both the sea and tidal River Arun. The southern Bosham inlier is particularly likely to be affected, potentially increasing chloride and sodium concentrations. However, the Chichester syncline should still operate as a barrier to groundwater flow, diverting outflows eastwards towards the Arun and preventing saline intrusion in the main part of the area.

The Environment Agency's East Hants Chichester Chalk numerical model (EHCC groundwater model) could be used to investigate some of these issues.

Potential changes of land use may have more influence on agricultural pollution than other changes in climate.

4.7 SUMMARY

Table 12 summarises the WFD groundwater classification status and objectives for each of the study areas; and Table 13 summarises the key groundwater quality issues for each of the case study areas, along with the outputs from UKCP18 data applied to the eFLaG and GeoCoast models, and the anticipated impacts on groundwater quality. The case study areas cover a karstic limestone with potential for very rapid recharge and groundwater flow (Dove), two Chalk catchments with dual porosity and permeability (Brighton and Chichester) and two sandstone areas with varying amounts of low permeability till cover (Birmingham and Eden). All except the Birmingham case study area are dominantly rural, with potential pollution from nitrate and also from pesticides (except the Dove).

The Birmingham case study area is highly urbanised and groundwater is affected by its long industrial history and particularly the legacy of metalworking. Large parts of the Brighton area are also urbanised.

Both the Brighton and Chichester areas are coastal, and also surrounded by tidal rivers. However, the potential for saline intrusion is very different in the two areas. The issue in the Brighton area is longstanding and well-managed by pumping from sources located at different positions in the aquifer at different times of the year. Whilst in the Chichester area, the majority of the Chalk aquifer is protected from saline ingress by the presence of the Chichester syncline.

In this research no quantitative assessment of the impacts of climate change on groundwater quality has been possible. However, a number of qualitative, heuristic statements can be made. All five areas are predicted to experience up to a 3°C increased in temperatures. This could increase reaction rates for degradation of contaminants, but such increases may only be marginal.

The direction of changes in long term average rainfall and recharge is uncertain. There is, however, generally a high level of confidence in increased rainfall and recharge seasonality (wetter winters and drier summers) and greater magnitude of extreme winter rainfall and recharge events. This has the potential to result in spikes of pollutants. This may be offset by dilution, although this is less likely in more karstified catchments such as the Dove. In all the areas, more extreme winter rainfall events could increase inputs of road salt into unconfined aquifers (less so in Eden and parts of Birmingham), however this may be offset by milder winters with less gritting required.

Water levels are predicted to be stable in the Brighton and Chichester areas and fall in the other areas. However, recovery from historic over-abstraction in the Birmingham area means that water levels are likely to continue rising in at least the medium term, mobilising pollutants stored in the soils and infill materials.

This assessment has not considered the impact of land use change associated with climate change on groundwater quality. Land use change may have a greater impact on groundwater quality than changes in climate.

Table 12 Water Framework Directive groundwater classification status and objectives (cycle 2) https://data.gov.uk/dataset/41cb73a1-91b7-4a36-80f4-b4c6e102651a/wfd-classification-status-cycle-2

Case study area		**Brighton coastal**	**Birmingham**	**Dove**	**Eden-Gt Crosby**	**Chichester**
WFD Groundwater Body		Part of Brighton Chalk block	Part of Tame-Anker-Meuse Permo-Triassic sandstones Birmingham-Lichfield	Dove-Carboniferous Limestone	Part of Eden valley and Carlisle Basin Permo-Trias	Chichester Chalk
Overall water body status	2013	Poor	Poor	Good	Poor	Poor
	2015	Poor	Poor	Good	Poor	Poor
	2109	Poor	Poor	Good	Poor	Poor
Quantitative status	2013	Poor	Poor	Good	Good	Poor
	2015	Poor	Poor	Good	Good	Poor
	2109	Poor	Poor	Good	Good	Poor
Chemical status	2013	Poor	Poor	Good	Poor	Poor
	2015	Poor	Poor	Good	Poor	Poor
	2109	Poor	Poor	Good	Poor	Poor
Quantitative saline intrusion for 2019		Good	Good	Good	Good	Good
Chemical saline intrusion for 2019		Good	Good	Good	Good	Good
Quantitative GWDTE status		Good	Good	Good	Good	Good
Chemical GWDTE status		Good	Good	Good	Poor	Good
Trend for 2019		Upward trend	Upward trend	Upward trend	Upward trend	Upward trend
Proposed overall water body objective		Poor by 2015	Good by 2027	Good by 2015	Good by 2027	Poor by 2015
Proposed quantity objective		Poor by 2015	Good by 2021	Good by 2015	Good by 2015	Poor by 2015
Proposed chemical objective		Good by 2027	Good by 2027	Good by 2015	Good by 2027	Good by 2027

Table 13 Summary of groundwater quality issues, UKCP18 projections and anticipated impacts on groundwater quality for each case study

Case study area	Quality issues	UKCP18 projections	Anticipated impacts
Brighton coastal	Saline intrusion, nitrate, pesticides, chlorinated solvents	Increased temperatures, increased winter rainfall and recharge, decreased summer rainfall, more extreme winter rainfall and recharge. Stable LTA groundwater levels but greater seasonality, though uncertain. Increased saline intrusion	Marginal increase in contaminant degradation from increased temperature. Possible winter spikes in nitrate and pesticides from flushing but may be offset by dilution. Possible increases in summer concentrations of contaminants from reduced dilution. Increased saline intrusion.
Birmingham	Sewers, organic contaminants, acidity, metals, nitrate, ash deposition, former landfill in sand pits	Increased temperatures, increased winter rainfall and recharge, decreased summer rainfall, more extreme winter rainfall and recharge. Decreasing groundwater levels though uncertain	Marginal increase in contaminant degradation from increased temperature. Possible winter spikes in nitrate, metals and solvents from mobilisation and leaching but may be offset by dilution. Possible increases in summer concentrations of contaminants from reduced dilution. Groundwater level recoveries more significant than climate change impacts on groundwater levels.
Dove limestone	Nitrate	Increased temperatures, increased winter rainfall and recharge, decreased summer rainfall, more extreme winter rainfall and recharge. Decreasing LTA groundwater levels but increased seasonality	Marginal increase in nitrate degradation from increased temperature. Possible winter spikes in nitrate and turbidity with little potential for dilution (karstic). Possible increases in summer concentrations of contaminants from reduced dilution. Groundwater level decreases may cause further upconing of thermal waters.
Eden valley	Nitrate and pesticides	Increased temperatures, increased winter rainfall and recharge, decreased summer rainfall, more extreme winter rainfall and recharge. Decreasing LTA groundwater levels but increased seasonality	Marginal increase in nitrate and pesticide degradation from increased temperature. Possible winter increases in nitrate and pesticides from flushing but may be offset by dilution. Possible increases in summer concentrations of contaminants from reduced dilution. Transient events may be less significant due to high storage of the aquifer. Groundwater level decreases may reduce baseflow to the Eden.
Chichester	Saline intrusion, nitrate, pesticides, chlorinated solvents	Increased temperatures, increased winter rainfall and recharge, decreased summer rainfall, more extreme winter rainfall and recharge. Stable LTA groundwater levels but greater seasonality including extremes. Increased saline intrusion	Marginal increase in contaminant degradation from increased temperature. Possible winter spikes in nitrate and pesticides from flushing but may be offset by dilution. Possible increases in summer concentrations of contaminants from reduced dilution. Increased saline intrusion. Higher groundwater level maxima increasing mobilisation of agricultural pollutants.
Everywhere	Road salting	-	More extreme rainfall may cause increased pollutant loadings, though potentially offset by increases in temperature meaning less gritting required

5 Implications for groundwater monitoring

5.1 CLIMATE AND WATER-QUALITY IMPACTS

The climate predictions outlined above for the case study areas across England suggest a range of outcomes and with very variable confidence in their probability of occurrence. Over the coming decades, high confidence is placed in increasing air temperature, with likely greatest increases in summer months, although absolute ranges are necessarily uncertain. There is also overall confidence that winters will be wetter and more extreme and summers drier. High confidence is placed too in increasing PET. Rather lower confidence is placed in recharge estimates, although these are projected on average to increase in winter and decrease in summer. Projections of groundwater levels vary between high to low confidence of falling levels to low confidence of little change.

The literature review has highlighted the limited numbers of studies of climate-change impacts on groundwater quality so far and the different countries and very varied climatic conditions under which they were investigated. The studies have also shown the strong intrinsic links between climate change and land-use change and the difficulties inherent in distinguishing the two. The uncertainties taken together make provision of recommendations for future groundwater monitoring as part of the GWQMN necessarily challenging and speculative, with a cautious approach needed in their development, and periodic review to ensure appropriate monitoring design.

5.2 MONITORING OBJECTIVES

As the 25 Year Environment Plan (HM Government, 2018) incorporates a drive for the UK to adapt to the effects of a changing climate, the Environment Agency needs to make provision for adequate monitoring of these effects within its national groundwater monitoring strategy. The Groundwater-Quality Monitoring Network (GWQMN) has been established over the last 20 years. In the March 2020 budget, the government's Infrastructure Funding included a four year programme of investment in groundwater monitoring, including the GWQMN.Recommendations for refining and future-proofing the network with respect to existing legislative drivers (WFD, Groundwater Directive, Nitrate Directive) were outlined by Ward et al. (2021). This included recommended improvements to the existing GWQMN as well as for new networks for the monitoring specifically of GWDTEs and groundwater temperature. The new networks have been taken up as part of the DEFRA National Capital Ecosystem Assessment programme. The objectives of the GWQMN are to:

- help characterise groundwater bodies and assess their pollution risks;
- establish groundwater body chemical status and identify anthropogenically-induced trends in pollutant concentrations;
- help design measures to protect groundwater and/or restore to good chemical status;
- evaluate the effectiveness of measures implemented on trend reversal;
- demonstrate compliance with protected areas objectives e.g. for drinking water and nitrate-vulnerable zones.

Many of these objectives overlap and are consistent with those for monitoring the impacts of climate change on groundwater quality and so many of the monitoring measures implemented already should capture observed impacts of climate change over time. Current robust monitoring of nitrate concentrations in groundwater for example, should help to define any future changing trends, including those related to changing climate. Moreover, the risk-based approach should identify pollutant risks as well as recognised risks arising from climate-change impacts, and the monitoring programme respond through increased monitoring accordingly. This section explores potential gaps in provision of data to monitor and assess the impacts of climate change from the existing GWQMN and makes some preliminary recommendations for addressing them.

One of the most useful principles for detecting impacts of climate change on groundwater quality is the ability to detect trends from a time series having established a robust baseline beforehand. This means that sites with long time series of groundwater-quality data will be extremely useful as change indicators. Paramount in the monitoring programme is to make best efforts to preserve long time-series records by securing access to the sites and maintaining their suitability and accessibility for monitoring. In establishing time series for groundwater quality in relation to climate change, the aim is to understand the changes caused by climate change rather than seek to reverse them as many of the changes may result from natural geochemical reactions caused by changes in input conditions (groundwater temperature, pCO_2 and volume of recharge).

5.3 WFD AND NVZ RISK ASSESSMENT

Assessment of risks to groundwater from a range of land-use pressures forms an integral part of the national GWQMN design in support of WFD objectives, as well as NVZ designation and a range of other uses. Assessing risks to GWDTEs are also an important WFD objective. The management unit for WFD assessment is usually the GWB and priority in the risk assessment is given to consideration of diffuse pollutants and widespread point sources. For current risks, vulnerable systems such as shallow flashy and unconfined aquifers are given priority over deeper or confined ones. In addition, for the assessment of groundwater-quality status, priority is given to monitoring of those with poor status with the result that good-status GWB/aquifers may be under-represented (Ward et al., 2021).

Risk assessment for NVZ designation considers a combination of risks from land-use pressures (i.e. nitrogen loading) taking into account local hydrogeological conditions, and risk observed from groundwater monitoring data (Defra, 2016).

Climate change could involve a change to local conditions and might therefore need an adjustment to monitoring priorities. For example, potential for contaminants to be transported to deeper parts of aquifers or to confined aquifers might be needed to be included as part of the risk assessment. Consideration of GWBs of currently good chemical status might also need an increased focus.

5.4 MONITORING RESPONSE TO POTENTIAL CLIMATE-CHANGE IMPACTS

5.4.1 WFD-related monitoring

Monitoring to assess the chemical status of GWBs is a key component of the monitoring design. The Ward et al. (2021) report highlighted the decline in spatial coverage of the sites in the GWQMN in the years since the network was set up and the need to replace sites to regain knowledge of the chemical status of under-represented bodies or aquifers, especially secondary aquifers. This is equally desirable for acquiring knowledge relevant to climate-change impacts.

Incremental increases in CO_2 in recharge due to atmospheric loading might impact on groundwater quality in different ways depending on geological and hydrogeological conditions, although in most conditions, impact of pCO_2 in the soil zone is likely to far outweigh that from atmospheric exchange with rainfall. Carbonate aquifers and sandstones with carbonate components are expected to have a strong pH-buffering capacity and hence strong resilience to any increased dissolved CO_2 concentrations. Lower resilience might be expected of carbonate-free silicate aquifers. These might be disproportionately defined as secondary aquifers. Increasing the spatial representation of monitoring sites in GWBs in secondary aquifers is therefore also relevant in a climate-change context.

Incremental increases in temperature (of a few °C) would not be expected to induce large changes in hydrogeochemical reactions but some changes in solute concentrations could occur, which could become increasingly important if water-quality thresholds are approached. Again, adequate spatial representation of monitoring sites in the English GWBs through strengthening the existing network would be an appropriate response.

The literature review also suggested some evidence for increasing concentrations of dissolved organic carbon in groundwater, partially due to temperature-induced biodegradation of organic matter but perhaps more related to land-use change. Organic carbon loading has implications for redox-sensitive solutes, though where this is likely to be most important would be difficult to predict, especially given the large number of redox-sensitive species in groundwater and aquifers.

The potential impacts on concentrations and distributions of organic compounds in groundwater are also difficult to predict and climate impacts equally difficult to unpick from land-use impacts. In the absence of clearer evidence of trends/impacts in given GWBs or aquifers, it is difficult to make recommendations on adaptation of monitoring design and a continuation of monitoring according to the current design is considered appropriate. The current design and Ward et al. (2021) recommendations are to reduce the numbers of measurements of analytes returning non-detects in order to reduce analytical and cost burden. This particularly applies to the organic suites but applies to a lesser extent to inorganic suites as well.

The Ward et al. (2021) review of monitoring advocated a change in emphasis towards introducing separate Surveillance and Operational monitoring modes to be more along the lines of guidance from the EC (European Commission, 2009) and UKTAG (UKTAG, 2012). Surveillance monitoring is designed to validate GWB risk assessments with respect to poor/good chemical status, identify and monitor management responses to trends and confirm meeting of environmental objectives. This is a more comprehensive monitoring to establish risk and inform subsequent Operational rounds. Operational monitoring in the intervening periods is restricted to GWBs at risk of failing to meet environmental objectives and is to determine status of these “at risk” GWBs and to monitor trends.

In the Surveillance rounds, this would involve analysis of ‘mandatory’ suites (I1 and I2/NUT) and other inorganic suites according to risk, coupled with exploratory screening of organic compounds with GC/LC-MS scans. In the Operational years, this would include continued monitoring of inorganic suites and quantitative certified analyses of organic compounds selected according to identified risk. This monitoring approach applies to any solutes or substances which have been first identified on the basis of a conceptual hydrogeological model and risk assessment and observed through monitoring, and this includes substances introduced or changed as a result of climate-change impacts. It should also see analysis continuing to be focussed on detectable organic compounds, maintaining the reduced analytical burden.

WFD objectives also require assessment of the impacts of groundwater quality on GWDTEs, including monitoring for pollutants such as nitrate. Monitoring of groundwater sites at GWDTEs can be achieved as for other sites in the GWQMN, but downhole sondes equipped with nitrate sensors are a potential option for evaluation where nitrate is the particular solute of concern.

5.4.2 Nitrate-vulnerable zones

The literature review revealed that more studies on climate-change impacts on groundwater quality discussed nitrate than other solutes. This might reflect inherent biases in the studies given the scale of existing nitrate problems in aquifers, but nonetheless indicates the emphasis that continues to be needed on evaluation of nitrate trends. Those published studies inferred general increases in nitrate concentrations on groundwater with time despite decreased loading, due to decrease in recharge and streamflow. It is unclear whether decreases in nitrate concentration follow from increased recharge due to dilution. The conclusions are caveated with large uncertainties.

The Ward et al. (2021) review identified the decline in numbers of monitoring sites for NVZ purposes across England despite the increases in designated NVZ areas. As for WFD evaluation, it highlighted the need to increase the spatial coverage of sites. The emphasis on monitoring is on aquifers/zones with high and increasing nitrate concentrations. This includes shallow and flashy unconfined aquifers. The published studies that suggest increasing nitrate concentrations in groundwater due to climate-induced changes in recharge also suggest increased nitrate loading and that aquifers not currently at risk could become so. This includes for instance, shallow confined aquifers that may be subject to denitrification. Increased nitrate loading could potentially see an advance of the nitrate front in a shallow confined aquifer through increased supply of the oxidising agent and change in the redox equilibria. This possibility brings more emphasis on the requirements for monitoring of nitrate (and other redox-sensitive solutes) at the edges of shallow confined aquifers, currently of low-nitrate status.

5.4.3 Urban floodplains

Urban flood plains subjected to frequent and increasing incidences of flooding could potentially be impacted by increases in concentrations of inorganic solutes and organic compounds from the wetted unsaturated zone and/or from urban contaminants such as sewage in the flood water or from landfills or other contaminated land (Visser et al., 2012). The literature review alluded to short-term impacts (e.g. weeks) of such water-quality changes (e.g. Ascott et al., 2016). Flooding impacts are difficult to monitor except by automated sensor installations because of the timing of the events. Sensors installed in boreholes can be used to monitor a basic suite of analytes, for example electrical conductivity and nitrate concentration, over the duration of (say hourly), and following, a flooding event. They cannot replace the comprehensive suites of analytes possible by physical sampling at non-flood periods, however. Security of the equipment during flood events is also a concern.

The groundwater-quality impacts of infiltration SuDs also merit further investigation in the climate-change context. Increased monitoring by both physical sampling and borehole-installed sensors could help provide a body of evidence to assess their impact (positive or negative) on the mobilisation of key analytes including urban organic contaminants (e.g. solvents, VOCs, PAHs), trace metals, nitrogen compounds and salinity. The bulk of such data would derive from sampling but with possibility of analysis of some solutes/volatiles by use of automated sensors, at least on a pilot scale.

5.4.4 Urban temperature impacts

It is important to recognise the difference between groundwater-quality changes as a direct consequence of climate change and those induced by urbanisation. The heat island effect is well-recognised in urban areas and is a response to a combination of use of groundwater for urban heating (or cooling), proximity of buildings with increased thermal storage, increased pavement surfaces and trapping of radiative heat loss, as well as leakage of sewage (Epting and Huggenberger, 2013; Saito et al., 2016; Yalcin and Yetemen, 2009; Zhu et al., 2010). The temperature effects of urbanisation are likely to be more pronounced than the effects caused by climate change directly. An estimated air-temperature rise of 2°C to 2050 was inferred for the Basel area of Switzerland on the basis of climate evidence. However, increasing urbanisation and influences of groundwater for thermal use resulted in simulated local groundwater temperatures rising by up to 8°C in the Basel urban area (Epting and Huggenberger, 2013). Saito et al. (2016) observed a groundwater temperature increase of up 7°C above baseline temperatures in proximity to a well installed as a ground-source heat pump system in Japan. Similarly, Zhu et al. (2010) inferred a temperature increase of up to 5°C in groundwater over the last century as a result of urbanisation in Cologne, Germany, and Winnipeg, Canada.

The Saito et al. (2016) study described changes in the inorganic chemistry of the heated groundwater. These were mostly increases in major-ion concentrations resulting from enhanced geochemical reactions at increased temperature. The changes were due to the urban heat-island effect and though perhaps an exaggeration of what might happen with climate-change impacts, cannot be attributed to climate change directly.

5.4.5 Industrial settings

The potential impacts of climate change on industrial settings including mine workings are highly uncertain. The water-quality effects of rising water levels on closed mine workings are well-established, although the previous sections describe low confidence in the impact of climate change on groundwater levels in the case studies across England. In a study of a Netherlands Zn smelter environment, models of climate time series projected increased precipitation in winter, reduced precipitation in summer, and higher air temperatures (between 2 °C and 5 °C) throughout the year (Visser et al., 2012). Future climate scenarios projected higher evapotranspiration rates, more irrigation, less drainage, lower discharge rates and lower groundwater levels, due to increased evapotranspiration and a slowing down of the groundwater system. As a result, lower concentrations of Cd and Zn in surface water were projected. The reduced leaching of heavy metals, due to drying of the catchment, showed a positive impact on a limited aspect of surface water quality. It is therefore conceivable that climate-change impacts could lead to improvements in water quality in some situations despite deterioration in others. Adequate monitoring of such situations for inorganic solutes as defined by the risk assessment should highlight any changes that occur over time.

5.4.6 Saline intrusion

Confidence in rising sea levels was found to be high in the climate scenario assessment with resultant impacts of saline intrusion in coastal areas, from both the sea and tidal rivers. Here, the impacts and potential future impacts on water quality are well-recognised and in terms of regulatory monitoring, concern chiefly the major ions (but also boron and fluoride). As with other environmental settings, the impacts of land-use have associated and possibly larger impacts, with knock-on consequences for groundwater quality due to relocation of pumping sites and changing pumping regimes as a result of saline influxes. Monitoring in areas impacted by saline intrusion involves ensuring adequate spatial coverage of coastal GWBs, and adequate frequency (suggested annual, for inorganic analytes).

5.5 SUMMARY

Although involving much higher uncertainties than trying to monitor for established pollution scenarios and known impacts, the principles of monitoring for the impacts of climate change are similar. They involve similar evaluations of the hydrogeological conceptual model and risk assessments. They also involve the same sets of analyte suites, similar frequencies of monitoring and similar provision for adequate spatial coverage. Similar to the objectives of other aspects of the GWQMN, the objectives are assessing groundwater chemical status and detecting change.

One aspect difficult to rationalise and monitor for is the distinction between climate-change impacts and land-use change responses. Given the distinctions, the objectives and desired outcomes from the monitoring must be clear.

6 Recommendations

6.1 PRIORITISED RISKS TO GROUNDWATER QUALITY FROM CLIMATE CHANGE

Based on the literature reviews (section 2) and case studies (section 4), Table 14 summarises the potential risks to groundwater quality from different components of climate change. For each component of climate change an initial subjective prioritisation has been made based on the potential impact on groundwater quality and the level of uncertainty.

Future increases in temperature associated with climate change have a high degree of confidence. These rises in temperature are likely to result in increases in microbial and chemical degradation of contaminants and may result in changes to groundwater ecology and microbial communities. The impact of groundwater temperature on groundwater invertebrates (stygobites) is uncertain given the limited number of studies. Brielmann et al. (2009) considered that stygobites are unlikely to survive above temperatures of about 16oC, but the low abundance and diversity observed in groundwater may be more related to factors such as resource availability than a temperature effect. The impact of changing temperature on microorganisms in groundwater is also hard to predict and likely dependent on many factors, including the absolute temperature change. A detailed review of pathogenic organism survival and inactivation rates in groundwater by John and Rose (2005) found increased inactivation rates with increasing temperatures for investigated viruses but no clear temperature association for investigated bacteria. The authors cited interplay of controls such as ambient microbial populations and water chemistry as well as temperature on bacterial reproduction and inactivation rates. It is therefore difficult to make generalisations about impacts of temperature change on groundwater ecology (invertebrate and microbial) as so many interrelated factors are likely to be involved. However, the scale of rises predicted (c. 1 – 2 °C) mean that such changes may be small and thus of low impact.

Long term average rainfall is projected to decrease by the end of the century, but the magnitude of change is typically small across the catchments (~5%) and the RCMs typically span both positive and negative values of change indicating some uncertainty in this result. The direction of change for groundwater recharge is less certain and, on average across the RCMs, the magnitude of change in long term average recharge by the end of the century is typically <5%. Given the uncertainty and the magnitude of change, the potential impacts on groundwater quality are largely unknown but possibly small.

There is high confidence that climate change will result in wetter winters and drier summers and greater magnitude extreme events, with associated impacts on groundwater recharge. This has the potential to increase winter leaching and mobilisation of contaminants, with potential increases in contaminant spikes. Conversely, there is also the potential for increased dilution in winter and decreased dilution during summer.

There is high confidence that increases in sea level will occur. This is likely to result in increased saline intrusion, but impacts are uncertain due to the role of aquifer heterogeneity (e.g presence of karst) in propagation of saline intrusion and the role of changing driving aquifer heads from changing rainfall and recharge. It should also be noted that this is clearly only a local issue confined to coastal aquifer settings.

The impact of land use change on groundwater quality has the potential to substantially affect contaminant sources and recharge pathways. This includes land use change directly caused by climate change, as well as land use change caused by other factors which in turn may be controlled by climate change. This includes, for example, increasing urbanisation; socioeconomic changes resulting in changes in food requirements and associated changes in cropping; implementation of land use changes and use of new chemicals to meet Net Zero. There is very low confidence in the trajectory of land use change associated with these different competing pressures, but a potentially highly significant impact that is more important than the direct impacts of climate change on groundwater quality as detailed above.

Table 14 Summary of prioritised risks to groundwater quality from climate change

Climate change component	Confidence	Description of impact on groundwater quality	Impact	Prioritisation
Land use change (climate induced)	Low	Changing contaminant sources and recharge pathways	High	1
Increasing rainfall and recharge seasonality and extremes	High	Increased leaching and spike risk, potentially increased dilution in winter and decreased in summer	Medium	2
Increases in sea level	High	Increased saline intrusion	Medium (local to coastal aquifers only)	3
Changes in LTA rainfall and recharge	Low (direction of change uncertain) but limited change	Unknown but likely small	Low	4
Increasing temperatures	High	Increased reaction rates and contaminant degradation, possible changes to groundwater ecology and microbial communities	Low (small temperature rise)	5

6.2 EVIDENCE GAPS

Based on the workshop (section 3 and Ascott (2022)) there are a number of key evidence gaps that need to be addressed in future work. These can be broadly divided into work to improve our understanding of changes in drivers and pressures and associated groundwater system responses, and work to improve understanding of impacts on receptors and potential management approaches. Whilst some of literature reviewed in section 2.2 has attempted to address these evidence gaps, the vast majority of studies are from outside England and work to date has focussed on local scale impacts. Within in England (and elsewhere), very limited work has been undertaken to systematically address these evidence gaps at the national scale.

6.2.1 Understanding current and future groundwater quality drivers and pressures and hydrogeological system response

A fundamental theme identified at the workshop was the high level of uncertainty associated with how drivers and pressures on groundwater are going to change, and how for even a single change in pressure the groundwater system response is uncertain, let alone when considering multiple, competing pressures. Consequently, detection and attribution of changes in future groundwater quality associated with individual drivers and pressures is likely to be highly challenging.

There is a need for further research to build the evidence base to support decision making regarding appropriate mitigation and adaptation measures. This is potentially a vast area of work, and therefore whilst acknowledging the interconnected nature of these issues, some prioritisation is required. Based on the outcomes of the workshop, some potential research projects could aim to:

1. Understand how nutrient sources and pathways may change in the future associated with both land use and climate change
2. Understand how changes in rainfall and recharge seasonality and magnitude of extremes will affect contaminant mobilisation, spikes and/or potential dilution.
3. Understand how groundwater levels will recover due to abstraction reductions and what are the associated water quality implications

4. Identification of the next generation of emerging contaminants considering changes in chemical use associated with climate change
5. Understand the current and future form and function of groundwater ecosystems and how these may be affected by climate change.

6.2.2 Understanding impacts on receptors and management strategies

The research outlined in 6.2.1 will improve our understanding of how groundwater quality is likely to be affected by climate and land use change. Further research is subsequently required to consider what the impacts of these changes will be on receptors, and what management strategies should be adopted. At this stage it is challenging to specify what such work would be, but based on the outcomes of the workshop some possible projects could include:

1. Quantification of impacts of future changes in groundwater quality on groundwater dependent terrestrial ecosystems and other receptors;
2. Understanding the human and environmental toxicology of both existing (e.g nitrate) and emerging substances;
3. Developing novel approaches to management of saline waters, to reduce the cost and environmental impact of desalination.
4. Modelling to evaluate the impact of management interventions (e.g rural land use change and nature-based solutions) on groundwater quality

6.3 MONITORING

Given the large uncertainties in future impacts of climate change on groundwater quality, any recommendations must be speculative and exploratory at this stage. Some key recommendations include:

1. Prioritise monitoring of sites with long time records of water-quality data to ensure continuity and preserve data of value for establishing baseline and long-term chemical changes associated with changes in climate, land use and abstraction;
2. Improve spatial coverage of the national GWQMN in terms of WFD and NVZ requirements in order to similarly improve coverage for monitoring for climate-change impacts;
3. Development of national groundwater temperature monitoring network to support research to evaluate the impact of anthropogenic warming and ground source heating and cooling scheme development on groundwater temperatures.
4. Consider increased use of automated sensors for measurement of basic water-quality analytes (e.g. electrical conductivity, groundwater temperature, nitrate) in flood-prone urban areas, GWDTEs and areas where rapid changes may be anticipated (e.g. karst).These need to be tested on a pilot scale for fitness for purpose before deployment as part of a network;
5. Consider further development of monitoring in urbanised settings, and piloting of use of automated sensors (e.g. for BTEX, PAHs) for infiltration SuDs schemes;
6. Assess the need to consider more good-status aquifers/GWBs to cater for potential but not yet actual water-quality impacts;
7. Consider use of more innovative monitoring techniques, such as application of citizen science and remote sensing data products
8. Building on the recommendations of Ward et al. (2021), consider the development of national monitoring of groundwater quality at groundwater dependent terrestrial ecosystems.
9. Review the groundwater quality data acquired periodically (e.g. on 6–10 year cycles) to assess any evidence for changes that could be climate-change-related and review risk assessments and monitoring design accordingly.

6.4 REGULATION AND ADAPTATION MEASURES

A key recommendation from the workshop (Ascott, 2022) was for better integration of groundwater resources and quality in regulation, as well as better integration of groundwater as a whole within the terrestrial water cycle and urban planning. An exemplar of this is the fact that groundwater is not considered within the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) programme to evaluate the potential impacts of new chemicals on the environment. Better integration of groundwater is a programme of work in its own right and beyond the scope of this project.

Given the high levels of uncertainty associated with potential impacts of future climate and land use change, development of “no regrets” adaptation measures are the most appropriate at this time. These are adaptation measures which will be of benefit under current climate and land use and in any climate and land use future. Such measures are likely to be addressing water quality issues that are occurring now, and may include:

1. Development of nutrient inventories at the catchment level to identify whether current nutrient loadings are acceptable.
2. Implementation of best management practices to reduce nutrient and pesticide losses from agriculture
3. Rural land-use change and nature-based solutions measures to increase water storage and slow runoff.
4. Development of flexible abstraction regimes (e.g. increasing abstraction when groundwater levels are high to increase storage).
5. Stronger enforcement of existing regulations and guidance related to groundwater protection and urban development
6. Deployment of real-time monitoring of water quality parameters at rapidly responding receptors (e.g public water supply boreholes in karstified aquifers, boreholes affected by saline intrusion) to assess contaminant spikes associated with extreme rainfall events under current climate

These measures are unlikely to address groundwater quality issues that are not already a concern. This highlights the importance of addressing the evidence gaps highlighted in section 6.2 to support the development of adaptation measures that go beyond “no regrets” measures.

6.5 NEXT STEPS

This report is the output of a project which was an initial scoping study undertaken in FY2021/22 to evaluate impacts of climate change on groundwater quality. To address the evidence gaps identified in section 6.2, focused research projects are required. Detailed development of project proposals is beyond the scope of this report, and should be undertaken as a co-production exercise between the Environment Agency, BGS and other key stakeholders. Such projects also need to be mindful of ongoing related work in the field. Some potential ideas for future work that could leverage existing projects could include the following:

- National mapping of potential risks to groundwater quality from climate change, combining land use projections (e.g the SPEED project https://uk-scape.ceh.ac.uk/our-science/projects/SPEED/land-use-change-projections), maps of changes in rainfall/recharge seasonality and extremes (UKCP18, Murphy et al. (2018)), maps of intrinsic vulnerability of different aquifer settings to changes in groundwater quality. This approach could be used to determine relative risks in certain areas, and therefore where to focus future regional and local scale work.
- National scale assessments of whether seasonality and extremes of rainfall and recharge result in increases/spikes or dilution of certain contaminants (e.g. nitrate), combined with use of UKCP18 data to evaluate potential future changes.
- Building on existing nitrate trend evaluation for NVZ and WFD reviews, determine whether current observed nitrate trends agree or disagree with national scale models

(e.g Wang et al. 2012) currently used to determine whether "peak nitrate" has occurred, to inform predictions of future nitrate concentrations in groundwater.

- Use of existing EA regional scale groundwater models or the national scale HydroJULES model with particle tracking to evaluate the impacts of climate and land use change on different conservative and potentially unconservative tracers and saline intrusion risk.

In addition to conventional scientific outputs of such work (e.g. models, datasets, reports and peer-reviewed papers), efforts should be made to disseminate results to stakeholders without hydrogeological expertise (e.g. developers, Local Planning Authorities and the general public).

7 Conclusions

This project has explored the potential impact of climate and land-use change on groundwater quality in England through a literature review, stakeholder workshop and five detailed case studies. Key conclusions of these tasks are detailed below.

Air temperature, evapotranspiration and sea level are all predicted to increase throughout the 21st century. Whilst the direction of change in annual precipitation is unclear, wetter winters and drier summers are predicted, with greater-magnitude extreme winter rainfall events. There is limited consistency in the direction of change in long-term average groundwater recharge and levels in England. There is some consistency in changes to seasonality in groundwater recharge and levels, with increased recharge and levels in winter, decreased recharge and levels in summer. There is limited evidence for changes in extremes (increasing high winter groundwater levels). The international literature suggests an overall worsening of groundwater quality over the next 50 – 80 years, although the trajectory of change for individual parameters is highly uncertain. Some parameters have a high level of confidence in a relationship with climate variables (e.g. shallow groundwater temperature and air temperature, sea level rise and salinity in coastal aquifers). However, for many components of climate change and water-quality parameters our understanding of relationships is near non-existent and speculative.

This lack of understanding was also highlighted in the stakeholder workshop. The workshop also identified the need for holistic approaches to management of groundwater in the terrestrial water cycle, and the need for continued monitoring. A number of focus areas were also identified: nutrients, emerging substances, changing rainfall characteristics, changing temperature, groundwater rebound, urban development and construction, changing salinity and groundwater ecosystems.

Across all five case study areas, air temperatures are predicted to increase by up to 3°C. This could increase reaction rates for degradation of contaminants, but such increases may only be marginal. Increased sea levels are predicted to increase salinity in coastal aquifers. The direction of changes in long term average rainfall and recharge is uncertain, but the magnitude of changes is predicted to be small. There is generally a high confidence of increased rainfall and recharge seasonality and greater magnitude of extreme events in winter. This has the potential to result in spikes of pollutants, but this could also be offset by increased dilution (although less likely in karstified catchments). Land use change and groundwater level recovery from historic overabstraction may have a greater impact on groundwater quality than changes in climate.

On the basis of the literature review, stakeholder workshop and case studies, an initial prioritisation of the potential risks to groundwater quality associated with climate change has been made. The relatively small increases in temperatures and changes in long term average rainfall and recharge make these a low priority. The local nature of increases in sea level affecting coastal aquifers make these a medium priority. The high confidence in changes in rainfall and recharge seasonality and extremes and potential impact through changes to leaching, spikes and dilution make these a relatively high priority. The highest priority risk is land use change, whether induced by climate change or otherwise. Land use change has the potential to change contaminant sources and pathways, and is both highly uncertain and has a potentially high impact.

Building on the previous project tasks, a number of recommendations have been made regarding evidence gaps, monitoring approaches, regulation and adaptation measures. Further research is required to address the significant evidence gap related to how drivers of groundwater quality are likely to change in the future, and what the hydrogeological system response to changes in multiple, competing drivers may be. This is a potentially large area of work. Prioritising research based on stakeholder needs raised at the workshop, potential projects could aim to understand future changes in nutrient sources and pathways, the impacts of abstraction reductions on groundwater quality, the impacts of changing rainfall and recharge characteristics, future emerging contaminants and the form and function of groundwater ecosystems. Additional subsequent work is required to consider the impacts of future changes in groundwater quality on downstream receptors, and what management strategies should be adopted. Given the uncertainty in potential future groundwater quality it is challenging to specify what this may be, but some possible areas raised at the workshop include assessment of the toxicology of existing and emerging contaminants, and development of new approaches to management of saline waters.

Recommendations for changes in groundwater quality monitoring are speculative at this stage given the high level of uncertainty associated with the impacts of climate change on groundwater quality. Existing long-term monitoring should be prioritised to assess baseline conditions and chemical changes associated with changes in climate, land use and abstraction. Development of new monitoring networks using novel techniques (e.g. remote sensing, automated sensors) should be considered. The spatial coverage of the existing GWQMN should be improved, including the potential to bring in currently good status groundwater bodies where groundwater quality may deteriorate in the future. Groundwater quality data should be reviewed periodically (c. 6 – 10-year cycles) to assess for evidence for changes that could be climate-change-related.

A key recommendation from the workshop was for better integration of groundwater resources and quality in regulation, as well as better integration of groundwater as a whole within the terrestrial water cycle and urban planning. These recommendations are aspirational and a programme of work in its own right and beyond the scope of this project. Given the level of uncertainty regarding the impacts of climate and land use change on groundwater quality, “no regrets” adaptation measures are most appropriate at this time. These measures will be of benefit under current climate and land use and in any future climate and land use, and can be delivered through collaborative working between the Environment Agency and external stakeholders (e,g. water companies, NFU). Some examples include: development of catchment nutrient budgets and implementation of best management practices to reduce nutrient losses from agriculture; land use change to increase water storage and runoff; development of flexible abstraction regimes; greater enforcement of existing groundwater protection guidance and regulation; development of real time monitoring to assess contaminant spikes. However, as “no regrets” measures address current groundwater quality issues, future issues which are not currently a known concern (e.g. the next generation of emerging contaminants) will not be impacted by these approaches. This highlights the importance of addressing the evidence gaps above through targeted research projects. Detailed project proposals to address these gaps are beyond the scope of this report and should be co-produced between the Environment Agency, BGS and other stakeholders.

References

British Geological Survey holds most of the references listed below, and copies may be obtained via the library service subject to copyright legislation (contact libuser@bgs.ac.uk for details). The library catalogue is available at: https://of-ukrinerc.olib.oclc.org/folio/.

ABESSER C & SMEDLEY P L 2008. Baseline groundwater chemistry: the Carboniferous Limestone aquifer of the Derbyshire Dome, British Geological Survey open report OR/08/028, 64pp.

AITKENHEAD N, BARCLAY W J, BRANDON A, CHADWICK R A, CHISHOLM J I, COOPER A H & JOHNSON E W 2002. *British Regional Geology: The Pennines and adjacent areas* (4th edition). (Nottingham: British Geological Survey)

AKBARI M, ALAMDARLO H N & MOSAVI S H 2020. The effects of climate change and groundwater salinity on farmers' income risk. Ecological Indicators, 110, 105893.

AKBARIYEH S, GOMEZ PENA C A, WANG T, MOHEBBI A, BARELT-HUNT S, ZHANG J & LI Y 2019 Prediction of nitrate accumulation and leaching beneath groundwater irrigated corn fields in the Upper Platte basin under a future climate scenario. Science of the Total Environment, 685, 514-526.

AKHTAR N, SYAKIR ISHAK M I, BHAWANI S A & UMAR K 2021. Various natural and anthropogenic factors responsible for water quality degradation: A review. Water, 13, 2660.

ALADEJANA J A, KALIN R M, SENTENAC P & HASSAN I 2020. Assessing the impact of climate change on groundwater quality of the shallow coastal aquifer of eastern Dahomey basin southwestern Nigeria. Water, 12, 224.

ALLEN D J, BREWERTON L J, COLEBY L M, GIBBS B R, LEWIS M A, MACDONALD A M, WAGSTAFF S J & WILLIAMS, A T 1997. *The physical properties of major aquifers in England and Wales.* British Geological Survey, 333pp. (WD/97/034).

ANAWAR H MD 2013. Impact of climate change on acid mine drainage generation and contaminant transport in water ecosystems of semi-arid and arid mining areas. Physics and Chemistry of the Earth, 58-60, 13-21.

ARNELL N W & FREEMAN A 2021. The impact of climate change on policy-relevant indicators of temperature extremes in the United Kingdom. Climate Resilience and Sustainability, n/a(n/a). DOI:https://doi.org/10.1002/cli2.12

ARTHURTON R S & WADGE A J 1981. Geology of the area around Penrith, Institute of Geological Sciences, memoir for 1:50 000 geological sheet 24.

ASCOTT M J, LAPWORTH D J, GOODDY D C, SAGE R C & KARAPANOS I 2016. Impacts of extreme flooding on riverbank filtration water quality. Science of the Total Environment, 554-555, 89-101.

ASCOTT M J, MANSOUR M M, BLOOMFIELD J P & UPTON K A 2019. Analysis of the impact of hydraulic properties and climate change on estimations of borehole yields. Journal of Hydrology, 577: 123998. DOI:https://doi.org/10.1016/j.jhydrol.2019.123998

ASCOTT M J, 2022. *Groundwater Quality into the Future: Workshop Summary*, British Geological Survey, 14pp. (OR/22/010)

ASCOTT M J, BLOOMFIELD J P, KARAPANOS I, JACKSON C R, WARD R S, MCBRIDE A B, DOBSON B, KIEBOOM N, HOLMAN I P, VAN LOON A F & CRANE E J 2021. Managing groundwater supplies subject to drought: perspectives on current status and future priorities from England (UK). Hydrogeology Journal May, 29 (3), 921-4.

ATAWNEH D A, CARTWRIGHT N & BERTONE E 2021. Climate change and its impact on the projected values of groundwater recharge: A review. J. Hydrol., 601: 126602. DOI:https://doi.org/10.1016/j.jhydrol.2021.126602

BANKS D 1997. Spas of England, 235-280 in Mineral and Thermal Groundwater Resources, Albu M, Banks D and Nash H (editors). London: Chapman and Hall

BARBIERI M, BARBERIO M D, BANZATO F, BILLI A, BOSCHETTI T, FRANCHINI S, GORI F & PELITTA M 2021. Climate change and its effect on groundwater quality. Environmental Geochemistry and Health. DOI 10.1007/s10653-021-01140-5

BLOOMFIELD J P, WILLIAMS R J, GOODDY D C, CAPE J N & GUHA P 2006. Review of the impacts of climate change on the fate and behaviour of pesticides in surface and groundwater – A UK perspective. Science of the Total Environment, 369, 163-177.

BLOOMFIELD J P, GAUS I & WADE S D 2003. A method for investigating the potential impacts of climate-change scenarios on annual minimum groundwater levels. Water and Environment Journal, 17(2): 86-91. DOI:https://doi.org/10.1111/j.1747-6593.2003.tb00439.x

BLOOMFIELD J P, MARCHANT B P & MCKENZIE A A 2019. Changes in groundwater drought associated with anthropogenic warming. Hydrol. Earth Syst. Sci., 23(3): 1393-1408. DOI:10.5194/hess-23-1393-2019

BRASSINGTON FC 2007. A proposed conceptual model for the genesis of the Derbyshire thermal springs, Quaterly Journal of Engineering Geology and Hydrogeology, 40, 35-46.

BRERETON N R & DOWNING R A 1975, Some applications of thermal infra-red linescan in water resources studies, Water Services, 79, 91-96.

BRIELMANN H, GRIEBLER C, SCHMIDT S I, MICHEL R & LUEDERS T 2009. Effects of thermal energy discharge on shallow groundwater ecosystems. *FEMS Microbiology Ecology*, 68 (3), 273–286.

BURRI N M, WEATHERL R, MOECK C & SCHIRMER M 2019. A review of threats to groundwater quality in the Anthropocene, Science of the Total Environment, 684, 36-154

BUSS S R 1999, Attenuation of strong acids in the Birmingham Sherwood Sandstone aquifer, PhD thesis, University of Birmingham

BUSS S R, TELLAM J H, LLOYD J W & HARRIS B C 1997. Natural retardation of industrial fluids in the Birmingham urban Triassic Sandstone aquifer, p97-100 in Groundwater in the Urban Environment: Problems, Processes and Management, Chilton et al (eds), Balkema, Rotterdam.

CAVELAN A, GOLFIER F, COLOMBANO S, DAVARZANI H, DEPARIS J & FAURE P 2022. A critical review of the influence of groundwater level fluctuations and temperature on LNAPL contaminations in the context of climate change. Science of the Total Environment, 806, 150412.

CHEN M, QIN X, ZENG G & LI J 2016. Impacts of human activity modes and climate on heavy metal “spread” in groundwater are biased. Chemosphere 152, 439-445.

CHISHOLM J I, CHARSLEY T J & AITKENHEAD N 1998. Geology of the country around Ashbourne and Cheadle, *Memoir British Geological Survey,* sheet 124.

DAILY P, INGRAM J & JOHNSON R 2006. Groundwater Quality review: the Permo-Triassic sandstone aquifers of the Eden valley and Carlisle Basin Groundwater Quality Monitoring Unit, Environment Agency report GWQR04 (ESI report 6438A R1)

RICHARD JOHN HARVEY DALLISON, A. PRYSOR WILLIAMS, IAN M HARRIS & SOPAN D PATIL 2022. Modelling the impact of future climate change on streamflow and water quality in Wales, UK, Hydrological Sciences Journal DOI: 10.1080/02626667.2022.2044045

DAILY PJJ & BUSS SR, 2013, Birmingham Permo-Triassic Sandstone Aquifer Investigation, Conceptual Model Report, ESI Ltd for Environment Agency, Midlands Region

DARLING W G & BOWES M J 2016. A long-term study of stable isotopes as tracers of processes governing water flow and quality in a lowland river basin: the upper Thames UK. Hydrological Processes, 30, 2178-2195.

DARLING W G, GOODDY D C, MORRIS B L & PEACH D W 2012. The hydrochemistry of a Chalk aquifer during recovery from drought, Quarterly Journal of Engineering Geology & Hydrogeology, 45, 47-486.

DAVIES M C 1973. TA Thermal infrared linescan along the Sussex coast, Water and Water Engineering 77, 392-296.

DEFRA 2016. Implementation of the Nitrate Pollution Prevention Regulations 2015 in England. Method for designating Nitrate Vulnerable Zones for groundwaters, 27 pp. Defra. London. https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/573533/groundwater-nvz-methodology-2017-2020.pdf.

DRAGONI W & SUKHIJA B S 2008. Climate change and groundwater: a short review, Climate Change and Groundwater. Geological Society London Special Publications, 288, 1-12.

EARMAN S & DETTINGER M 2011. Potential impacts of climate change on groundwater resources – a global review. Journal of Water and Climate Change, 2(4): 213-229. DOI:10.2166/wcc.2011.034

EDMUNDS W M & BREWERTON L J 1997. The Chalk of the South Downs, The natural (baseline) quality of groundwaters in England and Wales, Environment Agency R&D technical report W6/i722/4/A.

EDMUNDS W M 1971. Hydrochemistry of groundwaters in the Derbyshire Dome with special reference to trace constituents, Institute of Geological Sciences report 71/7, 52pp.

EDMUNDS W M, DARLING W G, KINNIBURGH D G, DEVER L & VACHER P 1992. Chalk groundwater in England and France: hydrogeochemistry and water quality, BGS research report SD/92/2.

ELLIS PA 2002. The impact of urban groundwater upon surface water quality: Birmingham- River Tame study, UK, PhD thesis, University of Birmingham.

ENTEC UK LTD 2008a. Environment Agency - Southern Region, Groundwater Body Groundwater Quality Reports: Brighton Chalk Groundwater Body Final report

ENTEC UK LTD 2008b. Environment Agency - Southern Region, Groundwater Body Groundwater Quality Reports: Chichester Chalk Groundwater Body Final report

EPTING J & HUGGENBERGER P 2013. Unraveling the heat island effect observed in urban groundwater bodies – Definition of a potential natural state. Journal of Hydrology, Vol. 501, 193-204.

EUROPEAN COMMISSION 2009. Common implementation strategy for the Water Framework Directive (2000/60/EC). Guidance Document No. 18: Guidance on groundwater status and trend assessment. European Commission. Luxembourg.

FORD M & TELLAM J H 1993. Source, type and extent of inorganic contamination within the Birmingham urban aquifer system, UK. J. Hydrology, 156, 101-135.

FORD M 1989. Extent, type and source of inorganic pollution below the Birmingham conurbation, PhD thesis, University of Birmingham

FORD M, TELLAM J H & HUGHES M 1992. Pollution-related acidification in the urban aquifer, Birmingham, UK, J. Hydrology 140, 297-312.

FORD T D 1999. The growth of geological knowledge in the Peak District, Mercian Geologist, 14, 161-190.

FOSTER S S D & SMITH-CARRINGTON A 1980. The interpretation of tritium in the Chalk unsaturated zone, J Hydrology, 46, 343-354.

GINTAMO T T, MENGISTU H & KANYERERE T 2021. GIS-based modelling of climate variability impacts on groundwater quality: Cape Flats aquifer Cape Town South Africa. Groundwater for SUSTAINABLE DEVELOPMENT, 15, 100663.

GREEN T R, TANIGUCHI M, KOOI H, GURDAK J J, ALLEN D M, HISCOCK K M, TREIDEL H & AURELI A 2011. Beneath the surface of global change: Impacts of climate change on groundwater. Journal of Hydrology, 405, 532-560.

GREEN T R 2016. Linking climate change and groundwater, Integrated Groundwater Management. Jakeman A J et al (eds), DOI 10.1007/978-3-319-23576-9_5

GREEN T R, TANIGUCHI M, KOOI H, GURDAK J J, ALLEN D M., HISCOCK K M, TREIDEL H, AURELI A 2011. Beneath the surface of global change: Impacts of climate change on groundwater. Journal of Hydrology, 405(3): 532-560. DOI:https://doi.org/10.1016/j.jhydrol.2011.05.002

GUNN J 1992, Hydrogeological contrasts between British Carboniferous Limestone Aquifers, International Contributions to Hydrogeology, 13, 25-41.

GUNN J, BOTTRELL S H, LOWE D J, & WORTHINGTON S R H 2006. Deep groundwater flow and geochemical processes in limestone aquifers: evidence from thermal waters in Derbyshire, England, UK. Hydrogeology Journal, 14, 868-881.

HALCROW 1994. National Rivers Authority, Severn Trent Region, Birmingham Water Resources Contract No.1. Sir William Halcrow & Partners Ltd.

HANLON H M, BERNIE D, CARIGI G & LOWE J A 2021. Future changes to high impact weather in the UK. Climatic Change, 166(3): 50. DOI:10.1007/s10584-021-03100-5

HANNAFORD J, MACKAY J, ASCOTT M, BELL V, CHITSON T, COLE S, COUNSELL C, DURANT M, JACKSON C R, KAY A, LANE R, MANSOUR M, MOORE R, PARRY S, RUDD A, SIMPSON M, FACER-CHILDS K, TURNER S, WALLBANK J, WELLS S & WILCOX A 2022. eFLaG: enhanced future FLows and Groundwater. A national dataset of hydrological projections based on UKCP18. Earth Syst. Sci. Data Discuss. [preprint], https://doi.org/10.5194/essd-2022-40, in review.

HERRERA-PANTOJA M, HISCOCK K M 2008. The effects of climate change on potential groundwater recharge in Great Britain. Hydrol. Processes, 22(1): 73-86. DOI:10.1002/hyp.6620

HERRERA-PANTOJA M, HISCOCK K M & BOAR R R 2012. The potential impact of climate change on groundwater-fed wetlands in eastern England. Ecohydrology, 5(4): 401-413. DOI:https://doi.org/10.1002/eco.231

HM GOVERNMENT 2018. A Green Future: Our 25 Year Plan to Improve the Environment, 151 pp pp. HM Government. https://www.gov.uk/government/publications/25-year-environment-plan.

HUGHES A., MANSOUR M, WARD R, KIEBOOM N, ALLEN S, SECCOMBE D, CHARLTON M, PRUDHOMME C 2021. The impact of climate change on groundwater recharge: National-scale assessment for the British mainland. Journal of Hydrology, 598: 126336. DOI:https://doi.org/10.1016/j.jhydrol.2021.126336

INSTITUTE OF GEOLOGICAL SCIENCES & SOUTHERN WATER AUTHORITY 1978. Hydrogeological map of the South Downs and adjacent parts of the Weald, 1:100,000.

IYALOMHE F, RIZZI J, PASINI S, TORRESAN S, CRITTON A & MARCOMINI A 2015. Regional risk assessment for climate change impacts on coastal aquifers. Science of the total Environment 537, 100-114.

JACKSON D & LLOYD J W 1983. Groundwater chemistry of the Birmingham Triassic Sandstone aquifer and its relation to structure, Quart. J Engineering Geology, 16, 135-142.

JACKSON D 1981. Hydrochemical aspects of the Triassic Sandstone aquifers of the West Midlands, PhD thesis, University of Birmingham

JACKSON C R, BLOOMFIELD J P & MACKAY J D 2015. Evidence for changes in historic and future groundwater levels in the UK. Progress in Physical Geography: Earth and Environment, 39(1): 49-67. DOI:10.1177/0309133314550668

JACKSON C R, MEISTER R & PRUDHOMME C 2011. Modelling the effects of climate change and its uncertainty on UK Chalk groundwater resources from an ensemble of global climate model projections. J. Hydrol., 399(1): 12-28. DOI:https://doi.org/10.1016/j.jhydrol.2010.12.028

JIMENEZ-MARTINEZ J, SMITH M & POPE D 2016. Prediction of groundwater-induced flooding in a chalk aquifer for future climate change scenarios. Hydrol. Processes, 30(4): 573-587. DOI:10.1002/hyp.10619

JOHN DE & ROSE J B 2005. Review of Factors Affecting Microbial Survival in Groundwater. ES&T 39 (19), 7345-7356.

JONES H K & ROBINS N S (editors) 1999. The Chalk aquifer of the South Downs. Hydrogeological Report Series of the British Geological Survey, 111pp.

KENDON E, FOSSER G, MURPHY J, CHAN S, CLARK R, HARRIS G, LOCK A, LOWE J, MARTIN G & PIRRET J 2019. UKCP Convection-permitting model projections: Science report, Met Office, UK.

KENNEDY-ASSER A T, ANDREWS O, MITCHELL D M & WARREN R F 2021. Evaluating heat extremes in the UK Climate Projections (UKCP18). Environmental Research Letters, 16(1): 014039. DOI:10.1088/1748-9326/abc4ad

KHALAJI M, KHOLGHI M, SAGHAFIAN B & BAZRAFSHAN J 2019. Impact of climate variation and human activities on groundwater quality in northwest of Iran, Journal of Water Supply: Research and Technology – AQUA 68.2.

KHAN A A, KHAN S U, SURFAN AI M A, JAVED T & KHAN A 2021. An impact of climate change and groundwater salinity on shadow price of water farmer's revenue and socioeconomic and environmental indicators in district Kohat-Pakistan. Environmental Science and Pollution Research, DOI 10.1007/s11356-021-16179-1.

KNIPE C V, LLOYD J W, LERNER D N & GRESWELL R 1993, Rising Groundwater Levels in Birmingham and the Engineering Implications, CIRIA Special Publication 92

LAND D H 1966. Hydrogeology of the Triassic Sandstones in the Birmingham-Lichfield district, Water Supply Papers of the Geological Survey of Great Britain, Hydrogeological Report No.2

LASAGNA M, DUCCI D, SELLERINO M, MANCINI S & DE LUCA D A 2020. Meteorological variability and groundwater quality: Examples in different hydrogeological settings. Water, 12, 1297.

LI R & MERCHANT J W 2013. Modelling vulnerability of groundwater to pollution under future scenarios of climate change and biofuels-related land use change: A case study in North Dakota, USA. Science of the Total Environment, 447,32-45.

LIPCZYNSKA-KOCHANY E 2018. Effect of climate change on humic substances and associated impacts on the quality of surface water and groundwater: A review, Science of the Total Environment, 640-641, 1548-1565.

MACDONALD A M ET AL 2016. Groundwater quality and depletion in the Indo-Gangetic Basin mapped from in suit observations. Nature Geoscience DOI: 10.1038/NGEO2791.

MACKAY J D, JACKSON C R & WANG L 2014. A lumped conceptual model to simulate groundwater level time-series. Environmental Modelling & Software, 61: 229-245. DOI:https://doi.org/10.1016/j.envsoft.2014.06.003

MANSOUR M M, WANG L, WHITEMAN M & HUGHES A G 2018. Estimation of spatially distributed groundwater potential recharge for the United Kingdom. Quarterly Journal of Engineering Geology and Hydrogeology, 51(2): 247-263. DOI:10.1144/qjegh2017-051

MAS-PLA J & MENCIO A 2019. Groundwater nitrate pollution and climate change: learnings from a water balance-based analysis of several aquifers in a western Mediterranean region (Catalonia). Environmental Science and Pollution research, 26, 2184-2202.

MCDONOUGH L K, SANTOS I R, ANDERSEN M S, O'CARROLL D M, RUTLIDGE H, KEREDITH K, OUDONE P, BRIDGEMAN J, GOODDY D C, SORENSEN J P R, LAPWORTH D J, MACDONALD A M, WARD J & BAKER A 2020. Changes in global groundwater organic carbon driven by climate change and urbanization. Nature Communications, DOI 10.1038/s41467-020-14946-1.

MCGILL B M, ALTCHENKO Y, HAMILTON S K, SYLVESTER S R & VILLHOLTH K G 2019. Complex interactions between climate change sanitation and groundwater quality: a case study from Ramotswa, Botswana. Hydrogeology Journal, 27, 997-1015.

MET OFFICE 2018. About UKCP18. https://www.metoffice.gov.uk/research/approach/collaboration/ukcp/about

MONKHOUSE R M & FLEET M 1975. A geophysical investigation of saline water in the Chalk of the south coast of England, Quart. J. Engineering Geology, **8**, 291-302

MORTAZAVI-NAEINI M, BUSSI G, ELLIOTT J A, HALL J W, & WHITEHEAD P G 2019. Assessment of risks to public water supply from low flows and harmful water quality in a changing climate. Water Resources Research 55, 10386– 10404. https://doi.org/10.1029/2018WR022865

MURPHY J, HARRIS G, SEXTON D, KENDON E, BETT P, CLARK, R, EAGLE K, FOSSER G, FUNG F, LOWE J 2018. UKCP18 land projections: Science report, Met Office, UK.

O'DWYER J,CHIQUE C, WEATHERILL J & HYNDS P 2021. Impact of the 2018 European drought on microbial groundwater quality in private domestic wells: A case study from a temperate maritime climate. Journal of Hydrology, 601, 126669.

PALMER M, HOWARD T, TINKER J, LOWE J, BRICHENO L, CALVERT D, EDWARDS T, GREGORY J, HARRIS, G, KRIJNEN J 2018. UKCP18 marine report, Met Office, UK.

PARADIS D, VIGNEAULT H, LEFEBVRE R, SAVARD M M, BALLARD J-M & QIAN 2016. Groundwater nitrate concentration evolution under climate change and agricultural adaptation scenarios: Prince Edward Island, Canada. Earth System Dynamics, 7, 183-202.

PIRRET J S R, FUNG F, LOWE J A, MCINNES R N, MITCHELL J F B & MURPHY J M 2020. UKCP Factsheet: Soil Moisture and the Water Balance, Met Office, UK.

POLEMIO M 2016. Monitoring and management of karstic coastal groundwater in a changing environment (Southern Italy): A review of a regional experience. Water, 8, 148

POWELL J H, GLOVER B W & WATERS C N 2000. Geology of the country around Birmingham, Memoir of British Geological Survey, sheet 168 (England and Wales).

PRUDHOMME C, HAXTON T, CROOKS S, JACKSON C, BARKWITH A, WILLIAMSON J, KELVIN J, MACKAY J, WANG L, YOUNG A & WATTS G 2013. Future Flows Hydrology: an ensemble of daily river flow and monthly groundwater levels for use for climate change impact assessment across Great Britain. Earth Syst. Sci. Data, 5(1): 101-107. DOI:10.5194/essd-5-101-2013

PULIDO-VELAZQUEZ M, PENA-HARO S, GARCIA-PRATS A, MOCHOLI-ALMUDEVER A F, HENRQUEZ-DOLE L, MARCIAN-SORRIBES H & LOPEZ-NICHOLAS A 2015. Integrated assessment of the impact of climate and land us changes on groundwater quantity and quality in the Mancha Oriental system (Spain). Hydrology and Earth System Science, 19, 1677-1693.

RANI M, JOSHI H, KUMAR K, BHATT D K & KUMAR P 2021. Climate change scenario of hydro-chemical analysis and mapping spatio-temporal changes in water chemistry of water springs in Kumaun Himalaya. Environment, Development and Sustainability, 23, 4659-4674.

RETTER A, KARWAUTZ C & GRIEBLER C 2021. Groundwater microbial communities in times of climate change. Current Issues in Molecular Biology 41, 509-537.

RIEDEL T 2019. Temperature-associated changes in groundwater quality. Journal of Hydrology, 572, 206-212.

RIVETT M O 1988. The organic contamination of the Birmingham aquifer, PhD thesis, University of Birmingham

RIVETT M O, LERNER D N, LLOYD J W & CLARK L 1990. Organic contamination of the Birmingham aquifer, UK, J. Hydrology, 113, 307-323.

RIVETT M O, TURNER R J, GLIBBERY P & CUTHBERT M O 2012. The legacy of chlorinated solvents in the Birmingham aquifer, UK: Observations spanning three decades and the challenge of future urban groundwater development, J Contaminant Hydrology, 140-141, 107-123.

ROBINSON E L, KAY A L, BROWN M, CHAPMAN R, BELL V A & BLYTH E M 2021. Potential evapotranspiration derived from the UK Climate Projections 2018 Regional Climate Model ensemble 1980-2080 (Hydro-PE UKCP18 RCM). NERC EDS Environmental Information Data Centre.

ROZEMEIJER J, NOORDHUIS R, OUWERKERK K, PIRES M D, BLAUW A, HOOIJBOER A & VAN OLDENBORGH G J 2021. Climate variability effects on eutrophication of groundwater, lakes, rivers and coastal waters in the Netherlands. Science of the Total Environment 771, 145366.

SAITO T, HAMAMOTO S, UEKI T, OHKUBO S, MOLDRUP P, KAWAMOTO K & KOMATSU T 2016. Temperature change affected groundwater quality in a confined marine aquifer during long-term heating and cooling. Water research, Vol. 94, 120-127.

SALEEM S, LEVISON J, PARKER B, MARTIN R & PERSAUD E 2020. Impacts of climate change and different crop rotation scenarios on groundwater nitrate concentrations in a sandy aquifer. Sustainability 12, 1153.

SHAND P, HARGREAVES R & BREWERTON L J 1997. The natural (baseline) quality of groundwaters in England and Wales, part 4: the Triassic sandstones of Cumbria, BGS technical report WD/97/54.

SIDIROPOULOS P, TZIATZIOS G, VASILIADES L, MYLOPOULOS N & LOUKAS A 2019. Groundwater nitrate contamination integrated modelling for climate and water resources scenarios: The case of Lake Karla over-exploited aquifer, Water, 11, 1201.

SMERDON B D 2017. A synopsis of climate change effects on groundwater recharge. J. Hydrol., 555: 125-128. DOI:https://doi.org/10.1016/j.jhydrol.2017.09.047

SORENSEN J P R, ALDOUS P, BUNTING S Y, MCNALLY S, TOWNSEND B R, BARNETT M J, HARDING T, LA GARIONE E M, STUART M E, TIPPER H J & PEDLEY S 2021. Seasonality of enteric viruses in groundwater-derived public water sources., Water Research 207, 117813.

SPRENGER C, LORENZEN G, HŰLSHOFF I, GRŰTZMACHER G, RONGHANG M & PEKDEGER A 2011. Vulnerability of bank filtration systems to climate change. Science of the Total Environment 409, 655-663.

STEIAKAKIS E, VAVADAKIS D, KRITSOTAKIS M, VOUDOURIS K & ANAGNOSTOPOULOU C 2016. Drought impacts on the fresh water potential of a karst aquifer in Crete, Greece. Environmental Earth Sciences, 75, 507.

STUART M E, GOODDY D C, BLOOMFIELD J P & WILLIAMS A T 2011. A review of the impact of climate change on future nitrate concentrations in groundwater of the UK, Science of the Total Environment, 409, 15, 2859-2873.

TAYLOR R G, SCANLON B, DÖLL P, RODELL M, VAN BEEK R, WADA Y, LONGUEVERGNE L, LEBLANC M, FAMIGLIETTI J S, EDMUNDS M, KONIKOW L, GREEN T R, CHEN J, TANIGUCHI M, BIERKENS M F P, MACDONALD A, FAN Y, MAXWELL R M, YECHIELI Y, GURDAK J J, ALLEN D M, SHAMSUDDUHA M, HISCOCK K, YEH P J F, HOLMAN I & TREIDEL H 2013. Ground water and climate change. Nature Climate Change, 3(4): 322-329. DOI:10.1038/nclimate1744

THIVYA C, CHIDAMBARAM S, THILAGAVATHI R, VENKATRAMAN GANESH N, PANDA BANAJARANI & PRASANNA M V 2018. Short-term periodic observation of the relationship of climate variables to groundwater quality along the KT Boundary. Journal of Climate Change, 4, 2, 77-86.

TYLER-WHITTLE R, BROWN S & SHAND P 2002. Baseline report series 3: The Permo-Triassic Sandstones of South Staffordshire and North Worcestershire British Geological Survey commissioned report CR/02/119N, Environment Agency National Groundwater and Contaminated Land Centre Technical Report NC/99/74/4

UKTAG 2012. Groundwater trend assessment. UK Technical Advisory Group on the Water Framework Directive.

VISSER A, KROES J, VAN VLIET M T H, BLENKINSOP S, FOWLER H J & BROERS H-P 2012. Climate change impacts on the leaching of a heavy metal contamination in a small lowland catchment. Journal of Contaminant Hydrology, 127, 47-64.

WARD R S, SMEDLEY P L, FARR G J, ROBERTSON A, BOWES M J, LEWIS M A, LAPWORTH D J & MAURICE L 2021. Developing a Groundwater Monitoring Network Fit for the Future: Recommendations. Open Report, OR/21/025, British Geological Survey. Keyworth.

WARREN S C 1962. Some notes on an investigation into sea water infiltration. Proceedings of the Society of Water Treatment and Examination, 11**,** 38-42.

WU Y, LIU S & GALLANT A I 2012. Predicting impacts of increased CO_2 and climate change on the water cycle and water quality in the semiarid James River Basin of the midwestern USA Science of the Total Environment, 430, 150-160.

XIAO H, TANG Y, LI H, ZHANG L, NGO-DUC T, CHEN D & TANG Q 2021. Saltwater intrusion into groundwater systems in the Mekong Delta and links to global change. Advances in Climate Change Research, 12, 342-352.

YALCIN T & YETEMEN O 2009. Local warming of groundwaters caused by the urban heat island effect in Istanbul, Turkey. Hydrogeology Journal, Vol. 17, 1247-1255.

YAWSON D O, ADU M O, MULHOLLAND B, BALL T, FRIMPONG K A, MOHAN S & WHITE P J 2019. Regional variations in potential groundwater recharge from spring barley crop fields in the UK under projected climate change. Groundwater for Sustainable Development, 8: 332-345. DOI:https://doi.org/10.1016/j.gsd.2018.12.005

YOUNG B & LAKE R D 1988. Geology of the country around Brighton and Worthing. Memoir of the British Geological Survey, sheets 318 and 333 (England and Wales)

YUSOFF I, HISCOCK K M & CONWAY D 2002. Simulation of the impacts of climate change on groundwater resources in eastern England. Geological Society, London, Special Publications, 193(1): 325-344. DOI:10.1144/gsl.Sp.2002.193.01.24

ZHU K, BLUM P, FERGUSON G, BALKE K-D & BAYER P 2010. The geothermal potential of urban heat islands. Environmental Research Letters, Vol. 5, 044002.